PREDICTING AND PREVENTING LOSSES OF IMPERILED FISH SPECIES IN AN URBANIZING ENVIRONMENT

by

SETH J. WENGER

(Under the Direction of Mary C. Freeman)

ABSTRACT

Increasing urban land cover is a threat to many freshwater fish species. To effectively manage this threat we need to understand the nature of the stressors, develop relationships between stressors and measures of population viability, create a policy for limiting stressors and use predictive modeling to fine tune the policy and forecast the outcome of management. In this dissertation, I use the development of a Habitat Conservation Plan (HCP) for the imperiled fish species of the Etowah River basin, Georgia, USA, as a case study in applying this kind of conservation approach. I review the literature on urban effects on fishes, concluding that stormwater runoff from impervious surfaces is most likely to be the major threat to imperiled species of the Etowah. I then develop models relating the presence/absence of five species to effective impervious area (EIA) after accounting for historic land use, hydrogeomorphic variables and the confounding factors of incomplete detection and spatial autocorrelation. For a species (the Cherokee darter, *Etheostoma etowahae*) that shows a relationship between EIA and abundance but not presence, I propose an extension of existing methods to simultaneously model species presence, abundance and detection. I then explain the policy we have developed under the Etowah HCP for limiting the total volume of runoff from developed sites. This policy has the potential to limit EIA to sufficiently low levels to permit species persistence in the Etowah while accommodating future growth. Finally, I illustrate how predictive modeling can be used to guide application of this policy and forecast the outcomes, even with limited data, and propose this as a general approach to managing for imperiled species threatened by land use change.

INDEX WORDS: Fish, streams, freshwater, urban, urbanization, Cherokee darter, Etowah darter, amber darter, predictive modeling, stormwater runoff, effective impervious area, impervious cover, EIA, Etowah Habitat Conservation Plan, HCP, land use, policy, management, Endangered Species Act

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Urbanization is a worldwide phenomenon with little sign of abatement (Cohen 2003). As a consequence, an increasing number of freshwater ecosystems can be expected to suffer from the "urban stream syndrome," a condition characterized by a flashy hydrograph, altered channel morphology, increased contaminant levels and degraded biotic assemblage (Walsh et al. 2005). An expected result of urbanization is a loss of sensitive fish species, as indicated by numerous studies that have demonstrated statistical relationships between fish communities and urban cover (e.g., Klein 1979, Meador et al. 2005, Walters et al. 2003, Wang et al. 2001). The risk is especially serious for narrowly-distributed endemic species. This is the situation facing the Etowah River basin in northern Georgia, USA, where an exceptionally diverse fish fauna (76 extant native species; Burkhead et al. 1997) is confronting rapid urbanization as the Atlanta metropolitan region expands northward. Four of these fish species are endemic to the system, and three species are federally listed under the US Endangered Species Act (ESA; Figure 1). The US Fish and Wildlife Service (FWS), which administers the ESA for terrestrial and freshwater species, has become increasingly active in reviewing development plans and requiring measures to minimize impacts to species. Nevertheless, most projects historically have escaped review (Robin Goodloe, FWS, pers. com.). The current trajectory is for the gradual degradation of aquatic resources in the Etowah, leading to increasing pressure on the FWS to take more significant action—setting the stage for a conflict that pits the growth and economic development of a region against the persistence of a suite of fish species. It is to avert this impending train wreck that the Etowah Habitat Conservation Plan (HCP) is being developed.

Under the ESA, “take” (killing, harassment, habitat destruction, etc.) of protected species is prohibited on both public and private lands, although an exception can be made if the take is incidental to otherwise lawful activities and a HCP has been written and approved.
Most HCPs authorize activities by private property owners on lands with populations of threatened or endangered species. Under the proposed Etowah HCP, local governments are the focus. The idea is that since the 11 counties and dozens of municipalities of the Etowah have regulatory authority over development activities, participating jurisdictions will implement a set of policies that manage the effects of development so that the imperiled fish species are minimally impacted. Developers in participating jurisdictions will benefit because the lengthy consulting process many currently go through with the US Fish and Wildlife Service (FWS) will be greatly streamlined. In addition, both local governments and developers will be covered under permits authorizing incidental take. All three federally listed species and six other imperiled species are covered by the plan (Table 1).

Creation of the Etowah HCP has been a nearly five-year process. Mike Harris, Chief of Non-Game Wildlife for the Georgia Department of Natural Resources, first proposed the development of an HCP for the imperiled aquatic species of the Etowah in 2001. At his suggestion, Laurie Fowler, Bud Freeman, Mary Freeman and I successfully wrote a US Fish and Wildlife Service (FWS) Habitat Conservation Planning grant. This funded research, planning and (perhaps most importantly), an extensive outreach program to involve developers, local officials and other stakeholders in the creation of workable policies for managing development impacts. A steering committee of local government officials was created to oversee and approve proposed policies.

We initially conceived of these policies as falling into two categories, which we called "how to develop" and "where to develop." The "how to develop" policies included such things as erosion and sedimentation regulations, riparian buffer ordinances, road crossing requirements to maximize fish passage, and a post-development stormwater management ordinance. These policies were selected to prevent and minimize the impacts of identified urban stressors. The "where to develop" policies were intended to be changes to comprehensive plans and zoning codes designed to steer development away from watersheds that supported major populations of the protected fish species.

However, while we made steady progress with stakeholders in developing workable "how to develop" policies, the "where to develop" component remained stuck in neutral through 2004. By that time we had conducted preliminary analyses of the relationship between impervious cover and the occurrence of Etowah darters, and the results were sobering. Etowah darters simply did not occur in streams with watershed impervious cover greater than 1-2%. While we couldn't establish causation, the potential implication was that the species was extraordinarily sensitive to changes associated with increasing impervious cover. Although we expected the "where to develop" policies to reduce the effects of urbanization, the available evidence was that the policies proposed thus far could not reduce effects to the equivalent of only 1-2% impervious cover, unless they were coupled with considerable constraints on land use. We estimated that to meet the goal, densities in many watersheds would need to be restricted to one home per 10 acres (or less), or subject to extremely low impervious surface limits. Otherwise vast tracts would need to be acquired and preserved. The first two options appeared politically impossible, and the third option would require hundreds of millions of dollars.

This problem—how to manage the impacts of urbanization to prevent the loss of sensitive fish species in the Etowah—motivated the research work of this dissertation. The solution we proposed was straightforward: since we could not practically limit total impervious cover to low levels, then we needed to limit *effective* impervious cover to low levels. We could do this with a volume-based stormwater performance standard (a "how to develop" policy) that was strictest in the most watersheds inhabited by the most sensitive species. This approach was justifiable because we had previously identified stormwater runoff as the premier threat to sensitive species in the Etowah (Chapter 2). To allow for high-density development such as commercial and industrial uses, we proposed allowing development "nodes" where less strict standards applied. But to ensure that these nodes would not result in excessive "take" of species, we proposed employing the models relating fish species with impervious cover to forecast species presence or abundance under future estimated effective impervious cover, assuming watershed buildout under the volume-based performance standard. This forecasting would allow us to test different node locations and would also incidentally provide a means of estimating species take, which was necessary for the HCP application.

This approach ultimately proved acceptable to the Etowah HCP Steering Committee and has been incorporated into the Etowah HCP itself. In this dissertation I present five papers that provide the background and framework for this approach.

Chapter 2 is an analysis of stressors to the sensitive fish species of the Etowah basin, with focus on the mechanisms by which urbanization leads to the extirpation of species. It provides the justification for the management policies of the Etowah HCP. It concludes that, based on our current understanding, stormwater runoff from impervious surfaces is currently the greatest threat to fish species in the Etowah, setting the stage for management of that runoff. This chapter also serves as a review of the literature addressing urban effects on stream fishes.

In Chapter 3, I model the presence/absence of five presumed sensitive Etowah fish species as a function of impervious cover and other land use indicators, after accounting for historic land use, hydrogeomorphic variables, and the confounding factors of incomplete detectability and spatial dependence. The model for the Etowah darter, one of the federally listed species in the Etowah, is a result of this study.

Chapter 4 introduces a new method for simultaneously modeling presence, abundance, and incomplete detectability of species. It is a simple extension of existing methods and has proven useful for modeling the Cherokee darter. This species is another federally listed fish and, although apparently absent from some urbanized watersheds (Figure 1), the models constructed in Chapter 3 did not show a strong relation between Cherokee darter occurrence and impervious cover. This left FWS and the HCP without a basis for estimating Cherokee darter "take" in relation to increasing urban development. I used the approach evaluated in Chapter 4, which builds on recently developed extensions of occupancy models, to analyze the response of Cherokee darter abundance to effective impervious cover (and other predictor variables) and to develop a model that could be used to forecast effects of increasing effective impervious cover on the species. The modeling approach may be generally useful for species in which patterns of occurrence and abundance arise from separate processes.

In Chapter 5, I present the volume control performance standard, known as "Runoff Limits." The stormwater regulations employed by most communities in the US limit stormwater discharge rates, and sometimes contaminant concentrations. Although these standards can reduce hydrologic alteration and contaminants, they are unlikely to reduce impacts to levels sufficiently low to permit survival of very sensitive species. However, a standard that limits volumes to pre-development levels does have this potential. One of the challenges for the Etowah HCP entailed understanding what policies were either unlikely to be sufficiently protective of the targeted fish species, or politically unacceptable, and to develop a workable alternative. Chapter 5 describes the resulting policy and contrasts it with other approaches.

Chapter 6 presents the overall approach I used for combining predictive modeling with the runoff limits policy to manage for the imperiled species of the Etowah. It uses the Etowah HCP as a case study and suggests that the approach is a general one for use in guiding management policy for imperiled species, especially when data are limited and uncertainty is high.

Because Chapter 6 relies upon and integrates the previous four chapters, it also serves the traditional role of a concluding chapter in a dissertation. However, I append a brief Chapter 7 that discusses how the material in the previous chapters has been incorporated into the Etowah HCP and provides a few general recommendations for future HCPs in other localities.

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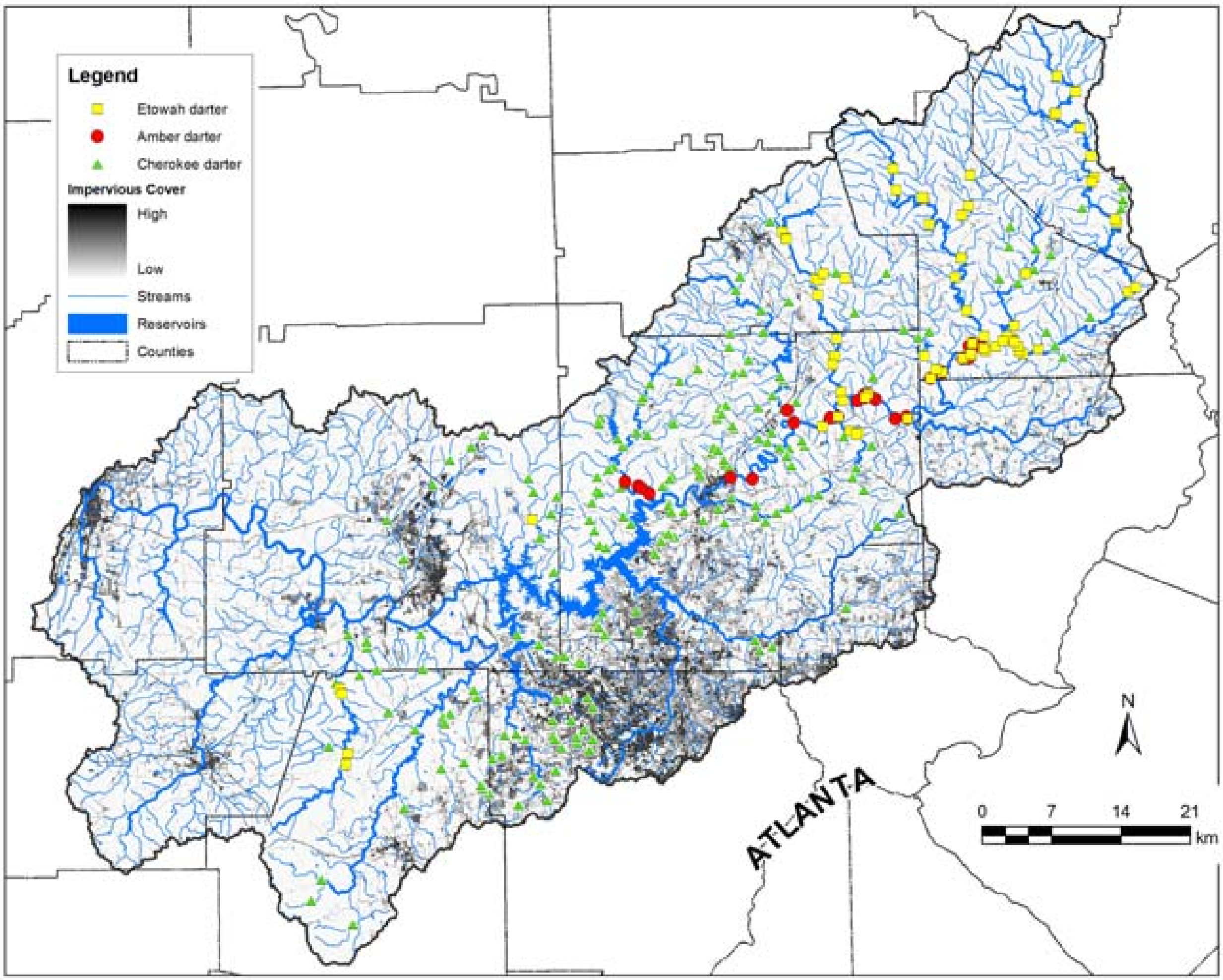
Table 1. Fish species covered under the Etowah HCP. Status refers to federal (Fed.) or state (GA) listing as endangered (E) or threatened (T). Federally listed species are the subjects of this study. Source: GDNR 1999, updated in 2006.

Scientific Name	Common Name	Family	Status
Macrhybopsis sp. cf. *aestivalis*	Coosa chub	Cyprinidae	GA E
Noturus sp. cf. *munitus*	Coosa madtom	Ictaluridae	GA E
Percina antesella (Williams and Etnier)	amber darter	Percidae	Fed. E / GA E
Percina lenticula (Richards and Knapp)	freckled darter	Percidae	GA E
Percina sp. cf. *macrocephala*	bridled darter	Percidae	GA E
Etheostoma etowahae (Wood and Mayden)	Etowah darter	Percidae	Fed. E / GA E
Etheostoma scotti (Bauer, Etnier and Burkhead)	Cherokee darter	Percidae	Fed. T / GA E
Etheostoma sp. cf. *brevirostrum* A	holiday darter	Percidae	GA E
Etheostoma sp. cf. *brevirostrum* B	holiday darter	Percidae	GA E

Figure 1. Known occurrences (through 2004) of Etowah, amber and Cherokee darters in the Etowah basin, with impervious cover as of 2001.



CHAPTER 2

STRESSORS TO IMPERILED FISHES IN THE ETOWAH BASIN:

MECHANISMS, SOURCES AND MANAGEMENT UNDER THE ETOWAH HCP[1]

[1] Wenger, S.J. and M.C. Freeman. To be published as a report on the Internet at *www.etowahhcp.org*.

Abstract

The Etowah River basin in Georgia, USA, supports nine imperiled fish species that are the object of protection under the proposed Etowah Habitat Conservation Plan (HCP). With urban land cover steadily increasing in the basin at the expense of forest and agricultural land cover, development-related activities and their consequences appear, as a group, to be the major threat to the species. However, urbanization is a complex phenomenon that involves numerous intermediate stressors. The purpose of this study is to review the scientific literature on urban stressors with the goal of identifying the major threats to the survival of fishes, so that management strategies may be implemented to avoid or minimize these threats as part of the Etowah HCP. We identify ten potential stressors: sedimentation, hydrologic alteration, extensive riparian buffer loss, contaminants (heavy metals, pesticides, etc.), movement barriers, channelization /piping of streams, invasive species, temperature alteration, loss of woody debris and eutrophication. For each we review the mechanisms by which the stressors may affect fish, the likely sources of the stressors within the Etowah, and the management strategies to be implemented under the Etowah HCP to address the stressors. We conclude that the first six stressors listed above are likely to be significant threats that must be managed by the Etowah HCP. We identify the most significant *source* of stressors as stormwater runoff from impervious surfaces, and the most critical policy as a stormwater management ordinance.

Introduction

The Etowah and its Aquatic Fauna

The Etowah River is a major headwater tributary of the Coosa River system in northern Georgia, USA. The basin is exceptional for its aquatic biodiversity, with 76 extant native fish species (Burkhead et al. 1997), including three species listed under the Endangered Species Act and six others that are considered imperiled but not currently listed (GDNR 1999). Five federally listed mussel species were once found in the Etowah (Burkhead et al. 1997), although all but one are now considered extirpated. A species of brachycentrid caddisfly (*Brachycentrus etowahensis*) also is considered imperiled because it is believed to exist only in the Etowah and Hiawassee Rivers. All of the Etowah's imperiled species are listed in Table 2.1.

Due largely to its proximity to Atlanta, the Etowah River basin is undergoing rapid development. During the 1990s, the Atlanta metropolitan area added more people than any other region in the U.S. except Los Angeles (McCosh 2000); in the last decade, counties in the southern portion of the basin have consistently ranked among the most rapidly developing in the nation. Accordingly, urban land cover in the Etowah Basin has increased steadily (Figure 2.1) (Kramer 2004), and the pace appears to be accelerating in recent years. This growth has raised concerns within the U.S. Fish and Wildlife that sedimentation, chemical contaminants and other stressors may threaten the survival of imperiled aquatic species.

These concerns are the impetus behind the development of the Etowah HCP, which calls for participating local governments to implement a set of growth management policies and ordinances to minimize the impact of future development on aquatic fauna, thus permitting additional growth without impairing survival and recovery of federally protected species. Development of policies is overseen by the Etowah HCP Steering Committee, the voting members of which are representatives of the participating local governments. The steering committee voted to focus on urbanization because other sources of stressors (e.g., agriculture and forestry) are declining as urbanization increases (Table 2.2, Figure 2.1), and the impacts of urbanization on streams are frequently more extreme than those of agriculture and forestry (Lenat and Crawford 1994, Wang et al. 2000). The Steering Committee also chose to write the Etowah HCP to cover the nine fish species listed in Table 2.1, but not the Etowah caddisfly and mussel species.

This document reviews the scientific literature and recent research on the effects of urbanization and suburbanization on sensitive fish species. It examines both the mechanisms and the sources of stressors, with a focus on the sources found within the Etowah basin itself. The purpose is to identify the key stressors to fish species in the Etowah and the management strategies available to mitigate those threats. As such, this review provides a major part of the scientific basis for the avoidance, minimization and mitigation policies of the Etowah HCP.

Overview of Stressors

Many studies have demonstrated that fish assemblages respond to a gradient of urbanization, with sensitive fishes disappearing as urbanization increases (Helms et al. 2005, Klein 1979, Meador et al. 2005, Morgan and Cushman 2005, Onorato et al. 2000, Roy et al. 2005b, Walters et al. 2005, Walters et al. 2003a, Wang et al. 2001, Wang et al. 2000)[2]. The mechanisms for

[2] Because effects can occur at relatively low levels of development, "urbanization" is used here to refer to any increase in development, including construction of low density suburban housing.

these changes are not simple. The conversion of a forested or agricultural landscape into parking lots, buildings and lawns produces a cascade of impacts to stream systems, including changes to hydrology, geomorphology, water temperature and stream chemistry, as well as inputs of various toxins (for recent reviews, see Allan 2004, Paul and Meyer 2001, Walsh et al. 2005b). Here, we organize these effects into ten categories of stressors (Table 2.3): sedimentation, altered flows, extensive loss of riparian buffers, movement barriers, contaminants, channelization and piping, loss of woody debris, eutrophication, invasive species and temperature alteration. This list is based in part on a previous review of stressors in the Etowah (Freeman et al. 2002) and the reviews cited above.

In creating this list of stressors, we have taken into consideration certain traits of imperiled fish species in the Etowah:

- Most are riffle-dwelling species, and tend to be found in association with coarse particles (gravel and cobble).
- Most are lotic specialists, and tend *not* to be found in lentic conditions.
- All are either narrowly distributed (e.g., several are endemic to the Etowah) or are very rare.

We have assumed, for example, that sedimentation of riffles is a threat because so many of the species are found in riffles. Loss of access to lotic habitat is likewise a concern. Conversely, degradation of lentic habitat is given less weight in the review. Some stressors are likely to be most acute at certain life history stages of species; for example, larval fish may be especially sensitive to physical displacement from excessive storm flows caused by habitat alteration.

Note that some stressors are best described as direct or proximate stressors, while others are indirect or ultimate stressors. Loss of riparian buffers, for example, generally acts via other stressors (i.e., it is a source of other stressors such as temperature alteration). Some stressors have both direct and indirect effects: for example, altered flows may lead to sedimentation, and general degradation from multiple stressors may facilitate species invasions. For simplicity we treat all stressors in a similar fashion.

Table 2.2 lists the categories of stressors with their potential sources and the HCP policies designed to avoid, minimize or mitigate the stressors. The list of potential sources includes those associated with urbanization as well as those associated with agriculture and forestry, although the HCP management policies only address urbanization. This does not mean that agriculture and forestry are not bound by the provisions of the Endangered Species Act; rather, it means that they will not enjoy the benefits of coverage under the Etowah HCP. In addition, there are some other aspects of urbanization that are also not covered by the Etowah HCP. Construction of roads by local, state and federal governments is not covered, and water and sewer construction and operations are not covered. These were deliberate omissions by the Etowah HCP Steering Committee designed to keep the HCP manageable by limiting its scope.

The next ten sections discuss each of the categories of stressors, including the mechanisms by which they affect fish, their sources, and the HCP management policy designed to address them.

Sedimentation

Studies have shown that fish richness, density and species composition in the Etowah Basin are well predicted by stream geomorphic variables, including those reflecting sedimentation (Walters et al. 2003a). Streams draining highly urbanized portions of the Etowah Basin have finer bed texture and higher turbidity, and fewer endemic or sensitive fishes, than those draining less urbanized areas, even after accounting for the effect of slope (Walters et al. 2003b). This is significant evidence that Etowah fish species are affected by sedimentation.

Increased sediment in streams can impact fish in two major ways: (1) bed sediment may degrade physical habitat and reduce productivity, and (2) suspended sediment may cause behavioral, sublethal health effects and mortality. These pathways can be further broken down into five mechanisms (Figure 2.2):

- Bed sediment can reduce primary and secondary production (Wood and Armitage 1997), or otherwise modify food webs (Schofield et al. 2004).
- Bed sediment can degrade spawning habitat for crevice and gravel-spawning fishes. Fine sediments can clog the interstices of larger particles, reducing spawning habitat (Berkman and Rabeni 1987); it can also reduce egg survival.
- Suspended sediment can reduce spawning success. Studies have shown that increasing levels of suspended sediment reduce spawning success of both salmonids and minnows, many of which depend on clear water for visual reproductive cues (Burkhead and Jelks 2001, Sutherland 2005).
- Suspended sediment can reduce feeding effectiveness for sight-feeding fishes (Sweka and Hartman 2003).

- Suspended sediment can cause stress, reduced growth, and physical abrasion to gills and other body parts (Newcombe and MacDonald 1991, Sutherland 2005). In a recent study, Sutherland (2005) showed that sediment levels sufficient to cause significant physical and physiological effects can occur in Southern Appalachian rivers more than fifty percent of the time.

Sources of sedimentation associated with urbanization

- Construction sites. Failure to properly install and maintain appropriate best management practices is a highly visible source of sediment to aquatic systems in the Etowah.
- Channel erosion. Runoff from impervious surfaces can lead to increased frequency and magnitude of storm flows in urbanizing streams. This can cause erosion of the stream banks and bed, leading to downstream sedimentation (Arnold et al. 1982, Hammer 1972, Trimble 1997). See Hydrologic Alteration.
- Utility and road crossings. Open trenching of utility lines across streams can lead to short-term but severe sedimentation (Reid et al. 2004). Road crossing construction can also lead to short-term sedimentation (Taylor et al. 1999), although literature on the topic appears almost entirely focused on logging roads.

Other sources of sedimentation

- Dredging and instream mining. A sand and gravel dredging operation in the Etowah near Canton has the potential to produce sedimentation, especially if adequate settling does not occur; however, there is little known habitat for covered species downstream of the operation, so the effects may not be severe. Amateur gold mining is practiced in the

Etowah as well; the impacts of this have not been evaluated, but the extremely small scale of these operations suggests that effects may not be major.

- Agriculture. In the Etowah, sedimentation from modern row crop agriculture appears to be a minor threat, because little row crop agriculture is practiced. However, bank erosion at cattle access points can be readily observed in many areas of the basin.
- Forestry. Forestry operations can result in substantial erosion, especially if best management practices are improperly applied. Reports from the Georgia Forestry Commission say that the most frequently violated BMPs are those for stream crossings (Green 2003). As a general rule, however, forestry activities produce less sediment than agriculture (Wood and Armitage 1997).
- Historic land use. Historic agriculture and gold mining deposited large amounts of sediment in stream and river valleys (Leigh 1994, Trimble 1970). Some channels may still be readjusting to this massive change, and may be slowly degrading as they cut down through the sediment back to their original channel level.

Depending on extent of urbanization, the dominant source of sediment may shift. Pre-development, agriculturally-derived sediment and historical sediment remobilized in the stream are often dominant sediment sources. As a watershed begins to urbanize, much sediment comes from construction sites. As development progresses, construction sites are replaced with impervious cover and there is a decrease in sediment delivery to streams; however, scouring flows associated with increased runoff increase the amount of sediment eroding from the bed and banks (Arnold et al. 1982, Doyle et al. 2000, Wolman 1967). In urbanizing watersheds, this stream channel erosion can be the major source of sediment (Trimble 1997), and researchers have found a significant sediment supply in streams even in heavily urbanized watersheds (Pizzuto et al. 2000). Streams may reach a new equilibrium after one to two decades, although some may take longer and others have not been found to stabilize in measured time frames (Henshaw and Booth 2000, Pizzuto et al. 2000).

Management Strategies

There is substantial evidence that sedimentation is a major threat to imperiled fishes of the Etowah, so the Etowah HCP policies address all three of the sources of sedimentation associated with urbanization. Sedimentation from construction sites is managed via a “standard operating procedure” (SOP) for enforcement of existing erosion and sedimentation ordinances by local governments. The Steering Committee approved this approach based on the argument that existing regulations are an adequate basis for an effective program, but the rules are unevenly enforced. An audit of the state Erosion and Sedimentation Control Program by the Georgia Department of Audits and Accounts came to this conclusion in 2001 (Georgia Department of Audits and Accounts 2001) and local officials confirm that it is still the case in many areas. The purpose of the SOP is to achieve a uniformly high level of enforcement across the basin. The SOP is supplemented by a grading ordinance which encourages developers to minimize the amount of exposed soil during site preparation, since the larger the area of exposed soil, the greater the possibility of erosion.

Sedimentation generated during construction of utility crossings is managed with a utility crossing policy that specifies that directional boring be used in preference to other stream crossing methods. Directional boring is a non-invasive alternative to open trenching that is increasingly common in the Etowah. Other approaches are permitted if it can be shown that directional boring is infeasible, except during spawning periods when directional boring is the only permissible option for crossing streams with populations of species covered by the Etowah HCP. The road crossings policy requires that appropriate best management practices be employed to minimize sedimentation during the construction of crossings.

Hydrologic Alteration

We focus on two aspects of hydrologic alteration: (1) an increase in storm flow frequency and intensity and (2) a decrease in base flows, which together create a "flashy" hydrologic regime. There are other potential types of hydrologic alteration, such as daily pulsing of flows below peaking hydroelectric dams, but we focus mainly on flashy stream flows because they are associated with urban runoff, which is arguably the most common source of hydrologic alteration in the Etowah Basin, as well as the one under potential management of the Etowah HCP. There are numerous mechanisms by which altered flows can affect sensitive fish (Figure 2.3):

- Reduced base flows can reduce lotic habitat, which especially affects high-flow specialists (Armstrong et al. 2001, Freeman and Marcinek 2004, Freeman and Marcinek 2006, Power et al. 1996, Walsh et al. 2004a).
- Increased storm flows will result in channel widening or deepening to accommodate the additional discharge, unless the channel is physically constrained (Arnold et al. 1982, Booth 1990, Doyle et al. 2000, Trimble 1997, Wolman 1967). During this process, which may take years or decades (if hydrologic alteration continues to increase), the bed is likely to be physically unstable at many locations (Booth 1990, Doyle et al. 2000). This instability may

significantly degrade habitat for spawning, feeding, and refugia, especially for riffle-dwelling species.

- The sediment from channel widening and deepening will move through the system, leading to sedimentation of downstream habitat. This may be ephemeral or long-term. A higher frequency of storm flows will also increase the amount of time that organisms are exposed to high levels of suspended sediment.
- Increased storm flows can cause physical washout of eggs and larval fishes, and stresses on adults as well (Freeman et al. 2001, Power et al. 1996).
- In addition to direct effects on fish, hydrologic alteration may also act via the four mechanisms described above to alter the quantity and quality of primary and secondary production in a stream (Bunn and Arthington 2002), indirectly affecting many fish species.
- For species that rely on annual hydrologic cycles for spawning or other life history patterns, disruption of the natural flow regime can reduce recruitment or cause other negative impacts (Bunn and Arthington 2002, Poff et al. 1997).
- Alteration of the natural hydrologic regime can also facilitate invasion by exotic species (Bunn and Arthington 2002, Fausch et al. 2001).

Sources

- Stormwater runoff from impervious surfaces. With the possible exception of Allatoona Dam operations (described below), runoff from impervious surfaces is the most significant source of hydrologic alteration in the Etowah basin. Impervious surfaces—roads, parking lots, rooftops, etc.—alter the natural hydrologic cycle. In a natural forested system, much of the stormwater infiltrates into the soil and is carried to the stream via shallow or deep subsurface flow paths. A significant amount evaporates or transpires, and a relatively small amount becomes surface runoff. In an urbanized system with high levels of impervious cover, most stormwater hits impervious surfaces and becomes runoff, which is then channeled quickly to streams via stormwater drain pipes. Relatively little infiltrates into the soil. As a result, storm flows in the stream are higher and more frequent, although briefer in duration, and base flows are lower (Ferguson and Suckling 1990) (Figure 2.4). Studies have shown that the storm discharge of urban streams can be twice that of rural streams draining watersheds of similar size (Pizzuto et al. 2000, Rose and Peters 2000), and the frequency of channel-forming events can be ten times that of the pre-development conditions (Booth and Jackson 1997).

Research in the Etowah basin conducted as part of the Etowah HCP demonstrated that watersheds with high imperviousness are flashier and have more frequent high-discharge events than watersheds with low imperviousness (Roy et al. 2005b). Variables describing hydrologic alteration explained 22-66% of the variation in fish assemblage richness and abundance, demonstrating that hydrologic alteration is indeed a potential mechanism of impacts to fish communities. Flow alteration was most significant during summer and autumn (Roy et al. 2005b).

Many researchers have made the case that the most problematic impervious surfaces are those that are directly connected to streams via drainage and conveyance systems (Alley and Veenhuis 1983, Booth and Jackson 1997, Walsh et al. 2004a, Walsh et al. 2005b). Studies have demonstrated that this effective impervious area (EIA) is a better predictor of stream biological and chemical response than total impervious area (TIA) (e.g., Hatt et al. 2004, Walsh et al. 2004b, Wang et al. 2001). A recent study in the Etowah found that EIA was a better predictor of sensitive fish occurrence than TIA (Chapter 3). The implication is that if EIA can be maintained at low levels—by using stormwater infiltration in place of conventional stormwater management systems that pipe runoff to streams– it is possible to maintain healthy aquatic systems while permitting further development of the watershed (Roy et al. 2005b, Walsh et al. 2005a). Through infiltration, EIA can stay nearly constant even while TIA increases.

As part of the Etowah HCP, researchers conducted a study to determine the levels at which sensitive fish species in the Etowah respond to increases in impervious cover (Chapter 3) The researchers tested the possibility that other factors, particularly historic land use, could also explain current fish distributions, as they have elsewhere (Harding et al. 1998). A total of 357 fish collections from the Etowah from 1999-2003 were used in the analyses. Five species of fish thought to be sensitive to urban or other stressors were evaluated. Two of these species, the Etowah darter and the Cherokee darter, are species covered by the Etowah HCP. The results showed that the Etowah darter and several other species were sensitive to increasing EIA, even when historic land use and other variables were taken into consideration (Figure 2.5).

- Reservoirs. Reservoirs can significantly alter hydrology downstream, especially when dams are operated for hydroelectric power generation (Freeman et al. 2001, Power et al. 1996). Hydropeaking dams, such as Allatoona Dam, release high flows only when additional power is needed. This can produce a daily pulsing cycle that is very different from the natural flow regime. Farm ponds and small water supply reservoirs also may substantially alter hydrologic regimes. Even if water is consistently released from a reservoir (e.g., as a minimum flow), the storage created by a reservoir may delay the return of normal or high flows to the stream following drought periods. Water supply reservoirs typically are operated to store water captured during higher flow periods for offstream use during low flow periods, with the effect of dampening moderate to high flows and in some cases augmenting low flows.

The operation of Allatoona Dam as a hydropeaking facility may be a factor explaining the absence of the imperiled fish species of the Etowah in the mainstem below the impoundment. There are several other water supply reservoirs either existing (e.g., Yellow Creek Reservoir) or under construction (e.g., Hickory Log Creek Reservoir) that are large relative to their watersheds and can significantly impact downstream flows.

- Water Withdrawals. Water withdrawals lower downstream water levels, and recent studies in the Georgia Piedmont show that fish assemblage integrity levels decline as water withdrawal levels increase (Freeman and Marcinek 2006). In the Etowah Basin, there are 21 water withdrawals, with maximum daily withdrawal levels ranging from 0.2 to 86 million gallons per day (mgd) (not counting Georgia Power's Plant Bowen) (Freeman et al. 2005). At present, no one of these appears to be at a level to cause major downstream problems, but further growth in the area will continue to increase pressure for additional water withdrawals.

Management Strategies

There is substantial evidence that hydrologic alteration is a significant threat to imperiled fishes in the Etowah. Management is focused on controlling stormwater runoff from impervious surfaces, which is both the most common source of hydrologic alteration and the one most amenable to management. The principal tool is a stormwater ordinance based on the model ordinance of the Metropolitan North Georgia Water Planning District (the "Metro District") (Metropolitan North Georgia Water Planning District 2004). The HCP ordinance includes five performance standards, four of which are based directly on the Metro District ordinance:

- Water quality protection: capture and treat runoff from all storm events of 1.2" or less, as well as the first 1.2" of runoff for all larger storm events.
- Channel protection: provide 24 hours of extended detention for runoff generated by the one-year, 24-hour storm event.
- Overbank flood protection: reduce the post-development 25-year, 24-hour storm event peak discharge rate to no more than the pre-development discharge rate.
- Extreme flood protection: design all stormwater management facilities to safely convey the runoff from the 100-year, 24-hour storm event.

The first standard is intended to reduce contaminants (discussed in a subsequent section), while the second standard is designed to manage hydrologic alteration, although its effectiveness is unproven. The third and fourth standards are intended to protect property from flood damage. These standards are retained in the model stormwater ordinance in part to ensure compliance with Metro District requirements. In addition, however, the Etowah HCP model stormwater ordinance includes a fifth requirement: a limit on the total volume of water that can leave a site as surface runoff. This "runoff limit" performance standard requires that excess runoff from small storms be infiltrated back into the soil as close as possible to where it is generated. Essentially, this should limit EIA to levels that are both low and predictable, providing near-natural hydrologic function as well as highly effective pollutant removal. The "runoff limit" standard applies only to watersheds that support populations of the most sensitive imperiled species ("Priority 1" and "Priority 2" areas). The ordinance allows local governments to designate development nodes where less strict runoff limits apply. However, the number and locations of these nodes are limited so that they will not threaten the survival of any of the species covered by the Etowah HCP (see Chapters 5 and 6).

Hydrologic alteration due to the management of Allatoona Dam for hydropeaking power production may be an important factor in making the Lower Etowah uninhabitable for many species. However, operation of the dam is outside the scope of the Etowah HCP. However, construction of new water supply reservoirs is addressed in the Etowah HCP in limited form. A protocol has been developed to evaluate potential impacts of competing reservoir locations, to ensure that reservoirs are built where they will have minimal impact on the imperiled species of the Etowah (see Movement Barriers for more information).

Extensive Riparian Buffer Loss

Removal of riparian buffers can have a number of effects on streams, including exacerbating several other stressors. Removal can (Figure 2.6):

- Destabilize stream banks, increasing stream sedimentation (Barling and Moore 1994, Beeson and Doyle 1995).

- Reduce capacity for trapping and removing contaminants from runoff (Dillaha et al. 1988, Groffman et al. 1991, Herson-Jones et al. 1995, Lowrance et al. 1997)
- Increase transport of nutrients to streams (Osborne and Kovacic 1993, Peterjohn and Correll 1984, Vought et al. 1994)
- Increase water temperature (Barton et al. 1985, Brazier and Brown 1973, Jones et al. 2006, Meyer et al. 2005a, Pusey and Arthington 2003).
- Increase light penetration to streams, increasing primary production (Noel et al. 1986, Pusey and Arthington 2003);
- Reduce woody debris inputs, removing a source of aquatic habitat (Karr and Schlosser 1978);
- Reduce leaf litter and terrestrial invertebrate inputs, decreasing production (Nakano et al. 1999, Pusey and Arthington 2003, Wallace et al. 1999).
- Decrease stream width, reducing the overall amount of stream habitat (Sweeney et al. 2004).

Many of these effects can lead to increased productivity of the stream system, which is not necessarily harmful. However, if loss of riparian buffers is extensive, then the stream can become inhospitable to fish species that depend on natural forested conditions. To better understand the effect of riparian buffer loss in an urban setting, Allison Roy and collaborators conducted a series of studies in the Etowah basin from 2002-2004 in association with the development of the Etowah HCP. They compared paired open and forested reaches along five small streams in suburban catchments (Roy et al. 2005a). They found no differences in overall habitat diversity between the reaches, although open reaches had higher amounts of woody debris and increased algal biomass. Open reaches had correspondingly higher densities of fish, especially the algivorous *Campostoma oligolepis*, but assemblages in all reaches appeared to be impaired due to urbanization. They concluded that small gaps in riparian buffers had little effect on biological integrity, and that the negative effects of urbanization on streams are primarily due to watershed-scale effects, not local loss of riparian forest (Roy et al. 2005a). Similarly, in a study of 30 small streams along a gradient of impervious cover, they found that land cover at the watershed scale was a filter for sensitive species, although loss of riparian cover could lead to higher abundances of some tolerant species (Roy et al. 2006). They concluded that riparian buffers alone are insufficient to maintain healthy fish assemblages in an urban setting where much stormwater runoff is transported to the stream in pipes, bypassing the buffer; nevertheless, preventing extensive buffer loss is a necessary component of an overall program of stream ecosystem protection.

Sources

Riparian forests were previously removed on many streams to increase the land available for crop agriculture and to provide cattle with water access. Current pressures to remove riparian forests are likely to be related to new development. Some of the most extensive riparian buffer losses are associated with golf courses, which historically have been able to secure variances from local and state buffer protection regulations to heavily modify streams. Other losses of riparian buffers are associated with piping of small streams for commercial and industrial development (see Channelization and Piping). This is an extreme form of buffer loss where the riparian zone is obliterated and the stream is completely disconnected from the terrestrial system.

Management Strategies

Preservation of riparian buffers is essential to protecting the imperiled species covered by the Etowah HCP. The chief management strategy for protecting riparian forests under the Etowah HCP is a riparian buffer ordinance. The regulations are based on a model ordinance of the Metropolitan North Georgia Water Planning District (Metropolitan North Georgia Water Planning District 2004) and require, at a minimum, protection of 50 ft naturally vegetated riparian buffers with an additional 25 ft setback for impervious surfaces along all perennial streams. A slightly less restrictive option (without the 25 ft setback) is recommended for Lumpkin County, Pickens County, Dawson County and Dawsonville, which are outside of the Metro District. A 50 ft buffer is the minimum necessary to maintain basic buffer performance for nutrient and contaminant removal (Wenger and Fowler 2000). The ordinance does not apply to agriculture and forestry lands, although appropriate best management practices are strongly encouraged on lands used for those activities. Variances are available, but mainly limited to cases where they are necessary to allow use of property and prevent a regulatory "takings."

Contaminants

Aquatic contaminants, including metals, hydrocarbons, pesticides, and other potentially harmful organic and inorganic compounds, are common in urban streams and may be partially responsible for the absence of sensitive fish in those system. Because of the expense of monitoring and experimental study, however, they have not received the attention they deserve. In the past, some studies have dismissed the role of water quality on aquatic species in urbanizing landscapes, but more recently scientists have challenged this view and suggest that contaminants may play a major role (Walsh et al. 2004a). There are a number of mechanisms by which contaminants can affect fish:

- Contaminants can cause direct mortality. Laboratory studies have shown that high levels of metals, pesticides and other contaminants can cause lesions, deformities and even mortality in fish (e.g., Meyers and Hendricks 1982, Woodling et al. 2002). However, most of the acute toxicity studies have been conducted on fish of commercial importance, although these may not be good predictors of nongame species responses (Woodling et al. 2002).
- Contaminant can have sublethal effects. Heavy metals such as mercury, lead, arsenic, selenium, cadmium and copper have been found to impair physiological functions of the liver, heart and kidneys, as well as impair growth rate, metabolic capacities and reduce respiration rates (e.g., (Rajotte and Couture 2002, Rowe et al. 2002) and cause morphological and morphometric changes to organs (Jagoe et al. 1996). Organic compounds such as surfactants, PCBs, insecticides (e.g., dioxins, malathion) and fungicides (e.g., imidazole, triazole) have been found to cause morphological alterations, increased instances of sores, lesions and fin erosion, impaired reproductive function and reduced reproductive fitness (e.g., Monod et al. 2004). Endocrine disrupting chemicals can cause subtle changes in fish physiology and sexual behavior or more permanent damage such as sexual differentiation and impairment of reproductive fitness (Carlisle and Clements 2003, Jobling and Tyler 2003, Noaksson et al. 2003, Van Der Kraak et al. 2001).
- Contaminants can reduce primary or secondary productivity. Contaminants can impair production and degrade the quality of food sources. Rosi-Marshall (2004) found that the quality of fine particulate matter as a food source was lower in the Chattahoochee River

below Atlanta than in a control, although she was unable to attribute the reduction to a specific cause. Studies have shown that aquatic invertebrate density, production and diversity is lower in streams with metal contamination (Maret et al. 2003).

Sources

- *Urban Point Sources.* The most recent database of point sources permitted under the National Pollution Discharge Elimination System lists 96 wastewater discharges in the Etowah. These include wastewater treatment plants, mines, and industrial facilities. The largest discharge is the cooling water for Georgia Power's Plant Bowen; the next largest discharges are the wastewater treatment facilities for Cobb County, Cartersville and Rockmart.

  Organic chemical compounds such as polychlorinated biphenyls (PCBs) are found in urban streams, sometimes as a result of point sources. Fish tissue samples from the Coosa River at Rome found levels of PCBs many times greater than the maximum recommended by the National Academy of Science/National Academy of Engineering (Zappia 2002). This is believed to be a legacy of a General Electric transformer plant in Rome. Because PCBs bioaccumulate and continue to cycle through biota, they can be transported both upstream (into the Etowah) and downstream by the movement of fishes, especially large migratory fish such as striped bass.

- *Urban Nonpoint Sources.* Pesticides are heavily used in urban and suburban areas, and many of these find their way to streams and groundwater (Schueler 1995). The highest levels of the pesticides 2,4,D, imazaquin and malathion recorded nationally in the National Water Quality Assessment program were found in an urban stream in Montgomery, Alabama (McPherson et al. 2003). A comparison of agricultural and urban groundwater quality in the Mobile Basin found a greater variety and frequency of pesticide compounds in the urban groundwater (Robinson 2003). Chlordane and other now-banned organochlorine pesticides are still common in urban streams, including those in the Mobile Basin (Zappia 2002). Although most pesticides applied to lawns remain bound to soils or thatch, a significant amount runs off during storm events, or infiltrates into shallow groundwater, and can be transported to streams (Schueler 1995).

Streets and parking lots can contribute large quantities of heavy metals (zinc, cadmium, chromium, nickel, manganese, copper and others) that are largely derived from automobiles (Bannerman et al. 1993, Muschak 1990, Van Hassel et al. 1980). Runoff from rooftops is relatively clean, although galvanized roofing can contribute large amounts of zinc (Bannerman et al. 1993). Oil and other hydrocarbons are also common constituents in runoff, and the amounts washed into streams and rivers may be massive (Paul and Meyer 2001). It is generally accepted that most of the contaminants in stormwater are washed off in a "first flush," although there is evidence that in highly urbanized watersheds, significant contaminants continue to be delivered after the first flush (Goonetilleke et al. 2005, Schueler 1994).

- *Agriculture.* Pesticides are frequently found in streams draining agricultural land uses, with herbicides being the most commonly detected (McPherson et al. 2003). Many agricultural streams still contain DDT and its degradation products (Zappia 2002).

Management Strategies

Although not well characterized, contaminants may be a major threat to the imperiled species covered by the Etowah HCP. Fortunately, the most significant source of contaminants—stormwater runoff—can be managed with the same stormwater ordinance that also controls hydrologic alteration. The ordinance requires that all new development must meet a standard of 80% removal of total suspended solids in the first 1.2” of runoff. This is intended to treat small storms and the first flush of large storms. This is based on requirements of the Metro District ordinance, but it is unknown whether this level is sufficiently protective. In addition, under the runoff limits program, new development in Priority Areas 1 and 2 will need to use infiltration practices to meet the volume control performance standard under most circumstances. Pollutant removal performances of infiltration practices are among the highest of any stormwater treatment BMPs (Walsh et al. 2004a). Studies have found nearly 100% removal of metals within bioretention areas (Davis et al. 2003). Studies of infiltration areas in Switzerland and France found that soils effectively trapped heavy metals and other pollutants; concentrations of pollutants decreased rapidly within a short distance in soils, indicating that even after decades of use there was effective treatment and little risk to groundwater (Barraud et al. 2005, Barraud et al. 1999, Mikkelsen et al. 1997). Infiltration areas may be less effective at removing nutrients, however; see the section on eutrophication, below. Management of point sources and agricultural sources are outside the jurisdiction of the Etowah Regional HCP.

Movement Barriers

Many fish species need to move upstream and downstream as part of their natural life cycles. A number of species release larvae in upstream areas, allowing them to drift to favorable downstream habitats (Robinson et al. 1998, White and Harvey 2003). This is then balanced by upstream movement of adults (Hall 1972). Movement barriers interrupt this process, fragmenting populations and making them more vulnerable to local extinction.

In addition, connectivity is essential for allowing a species to recover from small-scale disturbances: a local population may be wiped out by a pulse of sediment from a construction site or a chemical spill, but as long as recolonization routes are available, such periodic events may not have long-term impacts. Several authors have reported rapid recovery of defaunated streams (Bayley and Osborne 1993, Lonzarich et al. 1998, Peterson and Bayley 1993, Sheldon and Meffe 1995), suggesting that many species have a natural ability to recover from such impacts, provided that they have an unblocked route for recoloniation. In fact, many fish populations may be best termed metapopulations. According to classical metapopulation theory, a population can persist in numerous patches that are alternately extirpated and recolonized, allowing the overall persistence of the metapopulation even when local patches are inhospitable (Hanski and Simberloff 1997, Levins 1969). Metapopulation dynamics of freshwater fish have received only a modest amount of study to date (but see Dunham and Reiman 1999, Gotelli and Taylor 1999, Koizumi and Maekawa 2004), although it is widely thought that metapopulation dynamics do operate on many stream fishes in some fashion (Fagan 2002, Rieman and Dunham 2000). If this is so, then it is essential to maintain open pathways connecting population patches to allow recolonization. Because fish movement pathways are confined to the streams themselves (unlike those of amphibians and most aquatic arthropods, for example), fish are highly susceptible to the effects of movement barriers (Charles et al. 1998, Joy and Death 2001, Koizumi and Maekawa 2004). Movement barriers play a critical role in determining the likelihood of extinction or persistence of the imperiled fish species in the Etowah.

Sources

Because streams are linear systems, any obstacle or reach of inhospitable habitat can act as a significant barrier to fish movement. Movement barriers can be natural or man-made, partial or complete, one-way or two-way. Natural barriers include waterfalls, riffles, areas of bedrock and dry stream segments; man-made barriers include culverts and other road crossings, channelized stream segments, dewatered stream segments and dams.

- Natural Barriers. Movement studies have found evidence that even natural partial barriers such as riffles can inhibit movement, although the effect is most severe at low flows. A study of leopard darter (*Percina pantherina*) movement found very little movement across riffles and areas of bedrock (Schaefer et al. 2003), while a pair of short-term movement studies in Arkansas found that five species of cyprinids and centrarchids were three times more likely to cross short riffles (average 8m) than long riffles (average 50m) (Lonzarich et al. 2000). In a series of artificial stream studies, Schaefer (2001) found that shallower, faster riffles were greater barriers than deeper, slower riffles. Fish colonization rates in natural streams also were significantly reduced by the presence of shallow riffles (Lonzarich et al. 1998).

- Culverts. In the study of leopard darter movement discussed above, researchers also examined the effects of culverts (Schaefer et al. 2003). They found no movement upstream and little movement downstream through a culvert. In a series of experimental trials in an artificial stream, the same researchers found that culverts of various types greatly reduced movement of leopard darters, although in no case did they block movement entirely (Schaefer et al. 2003). A larger mark-recapture study in small Arkansas streams found that open box culverts and fords were not barriers to fish movement, but pipe culverts and a flat concrete slab road crossing significantly impeded movement (Warren and Pardew 1998). Researchers found that movement across a potential barrier was negatively correlated with water velocity across the barrier.

Culverts are ubiquitous in the landscape and increase in density with urbanization. Unlike riffles, many culverts are permanent barriers: they impede movement at both low and high flows. Most of the culverts that block movement are on small streams, so small stream fish species may be most severely affected. However, larger stream fish species generally have fewer distinct populations (i.e., because there are fewer large streams), so the effect of an individual barrier on a large tributary may be dramatic.

A study of 70 stream crossings in the Etowah River Basin found that 34% of surveyed crossings had characteristics likely to make them impassable to small-bodied fish (Millington 2004). Fifty-five percent of pipe culverts were considered impassable. In addition, most of the surveyed culverts appeared to be undersized, which produces high velocities and channel scouring at high flows. An unpublished fish movement study in the

Etowah basin found that fishes were much less likely to move through pipe or box culverts than stream crossings with bridges (Bill Ensign, Kennesaw State University, pers. com.). Taken together, research on stream crossings in the Etowah River basin illustrates that as many as one-third or more of the existing crossings on streams draining up to 50 km^2 are likely to impede passage by small fish, and that passage problems are likely to occur where pipe, and to a lesser extent, box culverts are used to cross streams.

- Reservoirs. The construction of Allatoona Reservoir isolated many populations in watersheds that previously were connected. This may be a factor in the extirpation of several fish species from small watersheds that are now tributaries to the reservoir rather than the Etowah mainstem. There are over 2000 smaller reservoirs in the Etowah that fragment streams (Figure 2.7). Most are on small (first or second order) streams, but a number are located on larger tributaries, effectively isolating large sections of headwaters.

Management Strategies

Movement barriers are a major threat to the species covered by the Etowah HCP. The main policy to manage the threat of movement barriers is the Stream Crossing Policy (referenced elsewhere as the Road Crossing Policy and the Road Crossings of Streams Policy). This requires that for new stream crossings, bridges must be used for streams draining areas of 20mi^2 or greater. Box and pipe culverts may be used on smaller streams, but these must be embedded or bottomless, and sized at 1.2 times the stream width, plus two feet. Multi-barrel pipe culverts are prohibited, although multi-barrel box culverts are allowed. These requirements apply to both privately constructed road crossings and those built by city and county governments and their contractors. Only new road crossings are affected, not replacement of existing crossings, except in the case where a bridge is to be replaced by a culvert.

In addition, the Etowah HCP includes a protocol to assist local governments in identifying reservoir locations with the least impact on protected fishes. The protocol is a procedure for evaluating the impacts of potential reservoir locations by examining:

- the number of habitat patches disturbed;
- the habitat quality in patches disturbed;
- the connectivity among patches disturbed; and
- the diversity of patch types disturbed.

These guidelines are intended to avoid conflicts between water resource development and stream conservation by removing from consideration those technically feasible options that would likely jeopardize the survival of the HCP species. The policy will also greatly streamline the reservoir review process by federal agencies, saving considerable time and expense for local governments and water utilities.

Channelization and piping of streams

Channelization includes the straightening, deepening, widening, embanking, stabilizing and/or clearing of streams and rivers for purposes of flood control, drainage improvement, navigation and relocation (Brookes 1988, Simpson et al. 1982, Swales 1982). *Piping* is the extensive culvertization of a length of stream designed to remove the waterway to allow other land uses, such as large buildings and parking lots. These two stressors are grouped together because both involve direct physical modification of the stream itself:

- Removal of habitat. Straightening, widening and deepening of channels usually includes the physical destruction of riffles and pools (Brookes 1988). Extreme channelization may replace the stream with a concrete-lined channel; similarly, piping replaces the natural stream channel with a metal or masonry pipe. In most cases, essential elements of habitat are entirely lost from the affected length of stream, and the remaining channel is very homogeneous. Channel straightening also reduces the total length of habitat available (Simpson et al. 1982). Loss of habitat affects all aspects of the lives of fish, leading to lack of spawning habitat, refugia, and/or food sources. Studies have shown that lack of habitat is a problem in channelized streams at both low flow (Brookes 1988, Simpson et al. 1982, Swales 1982) and high flow (Negishi et al. 2002). Piping a stream "eliminates aquatic habitat" outright (Meyer et al. 2005b).
- Reduction in food sources. Studies have shown that invertebrate biomass and diversity in channelized stream segments is much lower than in natural stream segments (Moyle 1976). Virtually no organisms can live within a piped stream, and insect diversity downstream from piped segments is greatly reduced (Meyer et al. 2005a).
- Hydrologic alteration. Channelization is often intended to increase the hydraulic efficiency of the channel and increase flow velocity, which results in large increases in peak discharge (Swales 1982).
- Sedimentation. There are often upstream and downstream geomorphic impacts of channelization and piping. Because the hydraulic efficiency is increased in the affected segment, erosion may occur downstream, resulting in sedimentation (Simpson et al. 1982).

- Downstream effects from loss of headwater streams. It is typically the small, headwater streams that are piped. Meyer and Wallace (2001) documented the important role of headwater streams in maintaining the overall ecological integrity of the aquatic system. Loss of headwater streams through piping may lead to decreased sediment retention, reduced processing of nutrients, contaminants and organic matter, and hydrologic changes, among other effects (Meyer et al. 2005a).

The effect of channelization on fish populations can be dramatic. Studies have shown that number, biomass and richness of fish in channelized stream reaches is typically far below that of comparable natural stream reaches (e.g., Huggins and Moss 1975, Moyle 1976). The reduction in biomass in channelized streams can be over 90% (Brookes 1988). The impact of piping appears to be less studied but possibly even more dramatic.

Sources

- Historic agricultural channelization. Most of the existing channelization in the Etowah Basin is probably associated with row crop agriculture. The extent of historic channelization is unknown and likely to be less extensive than in other parts of the country (e.g., the Midwest and lower Mississippi), but examples are evident from aerial photographs and from field observations.
- Urban channelization. Some streams are channelized in urban areas. Such projects are less common today than in the past; today, it appears more common for small treams to be piped and buried, while larger streams are better protected.
- Urban piping. Stream piping is common with large commercial and industrial construction projects. Current regulations in Georgia permit the piping of up to 200 ft of

small headwater streams without a permit, and larger streams and additional length with a permit.

Management Strategies

Piping of streams is common for large construction projects and constitutes a significant threat ot the species covered by the Etowah HCP. While there are no management actions under the Etowah HCP explicitly devoted to preventing channelization or piping of streams, riparian buffer regulations prohibit these activities for streams draining more than 20 acres. If buffer ordinances are properly enforced, streams over this threshold should be protected. Agriculture and forestry are exempt from these regulations, although they are expected to follow BMPs, which also mandate buffers. Other ordinances, such as conservation subdivision regulations, provide incentives for stream protection. Under the adaptive management provisions of the Etowah HCP, additional measures will be considered if monitoring and research show that channelization and piping remain significant threats in the Etowah Basin.

Invasive Species

The homogenization of fish communities due to the introduction of cosmopolitan species is occurring across the United States, but southeastern fish communities have suffered less than many other parts of the U.S. (Rahel 2000). Southeastern fish assemblages may be resistant to invasion due to their high diversity: the principle that more diverse communities are less invasible has a long history in the ecological literature (Elton 1958) and is supported by experimental evidence (Shurin 2000). Others (e.g., Moyle and Light 1996) disagree that aquatic community invasibility is related to diversity. Furthermore, there is ample evidence that southeastern fish communities are at risk of internal homogenization, in which habitat degradation eliminates specialists and local endemics in favor of habitat generalists (Scott and Helfman 2001, Walters et al. 2003a).

Thirteen non-native species are known from the Etowah (Table 2.4; Freeman et al. 2002). Of these, the red shiner (*Cyprinella lutrensis)* is considered the species of greatest concern because of its adaptability, tolerance, rapid reproduction and ability to hybridize with native minnows (Etnier and Starnes 1993, Marsh-Matthews and Matthews 2000). The redbreast sunfish, although widely distributed, has long been naturalized in the Etowah system and is not known to have led to declines in native fish species. Non-native trout species are confined to cool headwater streams and other temporary stocking locations, although they are sympatric with *Etheostoma brevirostrum*, one of the species covered by the Etowah HCP. The *Morone* species and threadfin shad are common in Lake Allatoona and the Etowah mainstem, but again are not thought to have had a noticeable impact on native species. Carp species are of some concern because of their ability to heavily graze macrophytes. The bluntnose minnow (*Pimephales notatus*) is an uncommon species in the Etowah mainstem.

Invasive species impact natives by both replacement and displacement (Helfman in press). Somewhat more specifically, mechanisms include:

- Competition. Some invasive species are highly aggressive competitors that may exclude native species from feeding, spawning or other essential activities. The red shiner may fall in this category.

- Predation. Introduced predators may eliminate native species by predation on adults, juveniles or eggs.
- Habitat Modification. It is possible that introduced herbivores, such as grass carp and common carp, could reduce native macrophytes, indirectly impacting other fish species. Thus far, there is no evidence of this in the Etowah.
- Hybridization. Invasive species can hybridize with native species, such as has been observed in western sucker species (Scoppettone et al. 1991)and with the red shiner and native *Cyprinella* species where the red shiner has been introduced (Hubbs and Strawn 1956, Taylor et al. 1994). This is threat is currently under study by David Walters, US EPA, Byron Freeman, Georgia Museum of Natural History, and Noel Burkhead, USGS.

Sources

Listed here are both the sources of non-native species and factors involved in their spread.

- Deliberate stocking. Worldwide, this may be the most common source of invasive species (Helfman in press). Trout are stocked in tributaries of the Etowah and have established permanent populations in higher-altitude streams with sufficiently cool water. Other species may be stocked in impoundments and subsequently escape upstream or downstream.
- Baitfish introductions. Various non-native species of minnows have been or are currently used for bait in the Etowah. The red shiner is thought to have been introduced as a baitfish.
- Aquarium introductions. Many species have spread as the results of the release of aquarium species (Helfman in press).

- Invasion from downstream. Some species may not have been introduced locally, but may have invaded the basin from downstream after they were introduced elsewhere in the Coosa system.
- Facilitation by degradation. Although the rate of introduction of nonnative fish species has not been found to be closely correlated with human population density (McKinney 2001), urbanization may indirectly facilitate species invasions by degrading aquatic habitat. Homogenization of fish communities has been observed in highland Southeastern stream systems that have been degraded by deforestation and sedimentation (Scott and Helfman 2001). Walters et al. (2003a) associated homogenization of fish communities with habitat sedimentation and alteration in the Etowah. In both of these cases, invading species were native downstream or elsewhere in the basin, although assumably certain non-native cosmopolitan species would also benefit from the same conditions. Hydrologic alteration (particularly reservoir construction) also has been cited as a factor facilitating the spread of invasive species (Bunn and Arthington 2002, Meffe 1991).

Management Strategies

At this time we do not have evidence that invasive species are a major threat to the species covered under the Etowah HCP, although trout may have impacts on species that inhabit the headwaters, such as *E. brevirostrum*. There are no management policies explicitly devoted to preventing species introduction or spread. Trout introduction is performed primarily by the state of Georgia, and outside the jurisdiction of the Etowah HCP. Several HCP provisions are intended to prevent degradation of aquatic habitat, which should reduce the threat of internal homogenization and perhaps reduce the invasibility of the system.

Temperature Alteration

Aquatic organisms are adapted to a limited temperature range. If stream water temperatures are raised or lowered beyond this range, potential effects include:

- Metabolic stress and mortality. Water temperatures outside the thermal tolerances of fish can lead to reduced metabolic activity and mortality. Although the thermal tolerances of many cold-water species have been thoroughly evaluated, those of most warm-water fish are little-studied (Eaton and Scheller 1996).
- Alteration of spawning times. Changes in water temperature may lead to earlier or later spawning. For example, spawning by river-dwelling basses (*Micropterus*) may vary depending on thermal regime (Graham and Orth 1986, Peterson and Kwak 1999), and the duration of spawning by many darter species is regulated by temperature (Hubbs 1985)
- Temperature shock. Sudden pulses of high or low temperature water may negatively impact fish species that would not be affected by the change if they had time to acclimate.
- Reduction in food sources/alteration in food webs. As with other stressors, temperature alteration may indirectly affect fish by impacting leaf decomposition, invertebrate life history, or otherwise disrupting natural food webs.

Sources

- Loss of riparian buffers. Riparian forests are critical in controlling stream temperature (Barton et al. 1985, Brazier and Brown 1973, Pusey and Arthington 2003). Recent studies in North Georgia showed that reduced forest cover in the riparian zone was correlated with increased stream temperatures (Jones et al. 2006, Meyer et al. 2005a).

- Stormwater runoff. Stormwater runoff from impervious surfaces tends to be warmer than runoff from natural vegetated soils, leading to elevated water temperatures in urban streams (Hatt et al. 2004, Walsh et al. 2001). Runoff from Atlanta during summer storm events has been associated with trout mortality in the Chattahoochee River, downstream from Buford Dam (John Biagi, pers. com.). Additionally, impervious cover prevents infiltration into shallow groundwater, which under natural conditions buffers stream temperature (Poole and Berman 2001).
- Reservoirs. Large hydropower dams are typically bottom-release and can maintain downstream water temperatures much lower than natural levels, resulting in such anomalies as the trout fishery of the Middle Chattahoochee in Atlanta, Georgia. In contrast to large dams, most small reservoirs are top-release, which can produce elevated downstream water temperatures.
- Water withdrawals. Reducing the flow in a stream reduces its ability to maintain a consistent temperature (Poole and Berman 2001).
- Thermal effluent discharges. Point source discharges, especially of power plant cooling water, may be warmer than receiving water bodies.

Management Strategies

At this time there is not evidence that temperature alteration is a major threat to the species covered by the Etowah HCP. The riparian buffer ordinance, stormwater management program and reservoir siting guidelines should help maintain natural stream temperature regimes essential to persistence of the HCP species.

Loss of Woody Debris

The presence of large woody debris is a critical element in structuring fish assemblages in streams and rivers in many locations, especially those with sandy subtrates. In these locations, removal of woody debris tends to reduce the abundance and diversity of fish (Angermeier and Karr 1984). Mechanisms include:

- Alteration of channel morphology and habitat. Removal of woody debris can lead to a loss of pool habitat and a homogenization of habitat characteristics, such as water velocity and benthic material (Wallace et al. 1995). Loss of woody debris can eliminate shelter from high-velocity flows (Crook and Robertson 1999).
- Decreased retention of organic and inorganic matter. Nutrient uptake lengths tend to be shorter in pools behind debris dams (Bilby and Likens 1980, Wallace et al. 1995), so loss of woody debris tends to decrease the "efficiency" of the stream in processing organic matter. This can decrease the overall productivity of the stream system.
- Loss of food sources/foraging sites. Woody debris provides substrate for invertebrates, which may be especially important in low gradient, sandy-bottom streams lacking other surfaces for attachment (Wallace and Benke 1984).

Sources

Although it is a problem elsewhere, researchers have not observed a lack of woody debris in Etowah streams, suggesting that this is not a major stressor. However, there are several potential causes of a lack of woody debris:

- Deliberate removal. Woody debris is regularly removed from bridge pilings to prevent excessive scour which could compromise the structures.

- Loss of riparian forests (Karr and Schlosser 1978). Without a source, woody debris in streams will eventually disappear.
- Hydrologic alteration. Increased magnitude and frequency of stormflows could increase export of woody debris from streams.
- Channelization. By increasing flow velocity and decreasing sinuosity, channelization can increase export of woody debris. However, a stream recovering from channelization may have unstable banks that generate large amounts of woody debris.

Management Strategies

Because lack of woody debris does not appear to be a major stressor at this time, there are no management strategies explicitly focused on this threat. However, the riparian buffer ordinance and stormwater management ordinance are expected to help ensure a supply of woody debris and minimize excessive washout.

Eutrophication

Eutrophication, or excessive nutrient input , is a widespread problem in surface waters of the U.S. (Carpenter et al. 1998). To date, concerns over nutrients in the Etowah basin have focused on the possible eutrophication of Lake Allatoona, the large multipurpose reservoir bisecting the system and providing drinking water to parts of the Atlanta metropolitan area. A comprehensive water quality assessment of Lake Allatoona (Rose 1999) characterized the impoundment as midway between mesotrophic and eutrophic, and predicted that the reservoir would be unfit for drinking water or recreation within 10 years unless phosphorus inputs were reduced. While nutrient pollution has long been implicated in the degradation of lentic water bodies, its effects on streams and rivers are less studied (Nijboer and Verdonschot 2004), and we have found few published cases that attribute fish kills or changes in fish assemblages to nutrients.

- Shifts in algal assemblages. It has been noted that there is a weaker causal relationship between nutrients and chlorophyll in streams than in lakes (Dodds et al. 2002). Nevertheless, nutrient enrichment can lead to shifts in the structure of benthic algal communities, as summarized by Carpenter et al. (1998). During low flow periods in recent years, algal blooms in the neighboring Conasauga River have covered shoals in a filamentous slime (Freeman and Wenger 2001) that may have degraded habitat for benthic fishes. Such blooms have not been described in the Etowah, but a combination of high nutrients and low flows, as occurred in the Conasauga, might permit a similar event.
- Death of *Podostemum*. We hypothesize that dense algal blooms could smother the benthic macrophyte *Podostemum*, which provides cover for benthic fishes as well as increases the productivity of invertebrate prey for stream fishes (Grubaugh and Wallace 1995, Hutchens et al. 2004).
- Declines in dissolved oxygen. In lentic water bodies, large algal blooms are followed by die-offs, which lead to oxygen sags as microorganisms degrade the dead algal material (Carpenter et al. 1998); this decline in dissolved oxygen can cause fish kills. Under low flow conditions, such events are possible in rivers as well.
- Rapid decomposition of leaves. Small, tree-shaded tributaries are light-limited and are not expected to suffer algal blooms and related problems. However, nutrient enrichment can accelerate decomposition of leaves and other heterotrophic food sources, causing unnatural seasonal shortages of primary food sources for the system (Greenwood 2004).

- Toxicity. At high concentrations both ammonium and nitrate can be toxic, although such cases are rare (Nijboer and Verdonschot 2004).

Sources

Although both nitrogen and phosphorus can be limiting in freshwater systems (Dodds et al. 2002), Lake Allatoona has been identified as phosphorus-limited (Rose 1999). Therefore, our focus is on phosphorus sources.

- Point sources. The wastewater treatment plants (WTPs) above Lake Allatoona are permitted for phosphorus loads totalling 67,026 lbs per year (Rose 1999), although several WTPs do not have phosphorus limits, so their contributions are unknown.
- Agriculture. In the Etowah, the main agricultural sources of phosphorus are likely to be poultry and cattle farming, both of which are still practiced extensively in portions of the basin (Boatright 2004). It is common practice to dispose of poultry litter by spreading it on pasture, sometimes in excess of the rate that can be used by vegetation or bound by soil. When it rains shortly after application, or when phosphorus accumulates to high levels in the soil, the likelihood that nutrients will be transmitted to surface water is increased (Chapman 1996).
- Septic systems. Under the right conditions, septic systems achieve very good performance. Studies have found 99% removal of phosphorus within 40 horizontal feet from a drainfield (McNeillie et al. 1994) and total nitrogen reduction of 99% two feet below a drainfield (Anderson et al. 1994). However, improperly located and poorly maintained septic systems can and do contribute to surface water pollution, and some consider septic systems the greatest threat to groundwater (Nizeyimana et al. 1996). Much of the population of the

Etowah basin is served by septic systems, although the exact proportion has not been determined and the proportion of failing systems has also not been estimated.

- Sewer systems. A sewer collection system conveys wastewater to a treatment plant, where the effluent becomes a point source (see above). Along the way, however, there are opportunities for leakage, especially at pump stations and other junctures. While septic system failures usually discharge partially treated wastewater, sewer line failures result in raw wastewater discharges, usually in close proximity to streams. The frequency of sewer line failures in the Etowah is unknown.
- Stormwater runoff. Urban runoff can be high in nutrients. The ultimate sources of nutrients in runoff are likely to include lawn fertilization, pet waste and atmospheric deposition, although partitioning contributions of these sources is difficult. Homeowners often apply lawn fertilizers at much higher rates than are required or specified, often exceeding agricultural rates (Barth 1995). In suburban areas, the great majority of nutrients in shallow groundwater may originate as lawn fertilizers (Flipse et al. 1984). Although pet waste in urban areas is thought to be a significant source of microbial pollution (Schueler 1998), its contribution to nutrient loading is unknown, though possibly significant. Atmospheric deposition on impervious surfaces is likely to result in nutrients reaching surface waters with little processing.
- Erosion of phosphorus-rich soils. Construction activities may mobilize soils saturated in phosphorus as a result of previous agricultural activities (Bennett et al. 1999).

Management Strategies

Because there is not strong evidence that nutrient pollution is an immediate threat to the imperiled species covered by the Etowah HCP, there are no management policies explicitly devoted to its control. The Steering Committee considered strategies focused on sewer and septic systems, but ultimately voted not to include them in the plan. Lawn fertilization and pet waste are difficult to regulate and are likely to be of secondary importance, so they are also not included in the management strategy. Point sources and agricultural activities are not covered under the Etowah HCP.

Nutrients in stormwater runoff may be trapped and removed by stormwater management practices. The emphasis of the Etowah HCP is on infiltration practices, which appear to have mixed success in terms of nutrient removal performance. Studies of bioretention areas found only moderate removal rates for ammonium and little to no removal of phosphorus (Dietz and Clausen 2005, Dietz and Clausen 2006), although a study of porous pavers showed significant removal of both phosphorus and nitrogen for stormwater passing through pavers (Dreelin 2006).

In short, nutrient pollution may not be well managed by the Etowah HCP. Because there is currently little evidence that eutrophication is a problem for the imperiled species covered by the plan, this omission may not be too damaging. If future research should prove otherwise, however, additional measures—outside of the Etowah HCP—may need to be taken.

Conclusions

Within this review we have identified sedimentation, hydrologic alteration, extensive riparian buffer loss, contaminants, movement barriers and channelization and piping as significant threats that require management by the Etowah HCP. Other stressors—invasive species, temperature alteration, loss of woody debris and eutrophication—appear to be less immediate or severe threats at this time, based on existing evidence. However, most of these other stressors are also reduced incidentally by the management policies of the Etowah HCP.

There are certain sources of stressors that also demand more attention than others. In particular, stormwater runoff is the most significant source of hydrologic alteration and contaminants, and may also be a major source of sedimentation, temperature alteration, loss of woody debris and eutrophication. This makes it the paramount source of stressors and the major focus of management efforts. This is consistent with findings from other researchers. In a recent paper evaluating the impacts of urbanization on streams—termed the "urban stream syndrome"—the authors concluded that stormwater runoff was the dominant source of impairment: "The mechanisms driving the [urban stream] syndrome are complex and interactive, but most impacts can be ascribed to a few major large-scale sources, primarily urban stormwater runoff delivered to streams by hydraulically efficient drainage systems" (Walsh et al. 2005b).

For this reason, the stormwater management policy of the Etowah HCP is absolutely critical. In particular, the runoff limits performance standard that requires the use of infiltration is essential for reducing hydrologic alteration and contaminants from runoff. There are five other major policies that are considered essential components of the Etowah HCP. These are erosion and sedimentation control, the stream buffer ordinance, road crossings of streams, utility crossings of streams, and the water supply planning protocol. Properly implemented, enforced and supported by adaptive management when necessary, we contend that these policies will be sufficient for maintaining healthy populations of the imperiled fish species covered by the Etowah HCP.

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Table 2.1. Fish species covered under the Etowah HCP. Status refers to federal (Fed.) or state (GA) listing as endangered (E) or threatened (T).

Scientific Name	Common Name	Family	Status
Macrhybopsis sp. cf. *aestivalis*[1]	Coosa chub	Cyprinidae	GA E
Noturus sp. cf. *munitus*[1]	Coosa madtom	Ictaluridae	GA E
Percina antesella (Williams and Etnier)	amber darter	Percidae	Fed. E / GA E
Percina lenticula (Richards and Knapp)	freckled darter	Percidae	GA E
Percina sp. cf. *macrocephala*[1]	bridled darter	Percidae	GA E
Etheostoma etowahae (Wood and Mayden)	Etowah darter	Percidae	Fed. E / GA E
Etheostoma scotti (Bauer, Etnier and Burkhead)	Cherokee darter	Percidae	Fed. T / GA E
Etheostoma sp. cf. *brevirostrum* A[1]	holiday darter	Percidae	GA E
Etheostoma sp. cf. *brevirostrum* B[1]	holiday darter	Percidae	GA E

[1] Undescribed species assumed most closely related to *Macrhybopsis aestivalis, Noturus munitus, Percina macrocephala, and Etheostoma brevirostrum*, respectively.

Table 2.2. Major Land Cover Categories in the Etowah, 1974 and 2001. Data source: National Land Cover Database.

Category	1974	2001
urban	5%	11%
forest	68%	59%
ag	19%	14%

Table 2.3. Stressors to sensitive aquatic species in the Etowah Basin.

Stressor	Sources	HCP Management Policy
Sedimentation	Construction sites Channel erosion Utility and road crossings Agriculture Forestry Historic land use	Erosion and sedimentation control Stormwater management policy Utility crossing policy
Hydrologic alteration	Stormwater runoff Reservoirs Water withdrawals	Stormwater management policy Water supply planning protocol
Extensive riparian buffer loss	Agriculture Golf courses Other construction	Riparian buffer ordinance
Contaminants (heavy metals, pesticides, etc.)	Point sources Stormwater runoff Agriculture Forestry	Stormwater management policy
Movement barriers	Natural barriers Road crossings Reservoirs and Ponds	Road crossing policy Water supply planning protocol
Channelization / piping	Agriculture Urban channelization Urban piping	Riparian buffer ordinance
Invasive species	Deliberate stocking Baitfish introductions Aquarium introductions Invasion from downstream Hybridization Facilitation by degradation	(none)
Temperature alteration	Loss of riparian buffers Stormwater runoff Reservoirs Water withdrawals Point sources	Stormwater management policy Water supply planning protocol Riparian buffer ordinance
Loss of woody debris	Deliberate removal Loss of riparian buffers Hydrologic alteration Channelization	See: extensive loss of riparian buffers and hydrologic alteration
Eutrophication	Point sources Agriculture Septic systems Sewer systems Stormwater runoff Erosion	Stormwater management policy Erosion and sedimentation control

Table 2.4. Nonindigenous fishes of the Etowah basin.

Common name	**Family**	**Scientific Name**
threadfin shad	Clupeidae	*Dorosoma petenense*
grass carp	Cyprinidae	*Ctenopharnygodon idella*
red shiner	Cyprinidae	*Cyprinella lutrensis*
common carp	Cyprinidae	*Cyprinus carpio*
bluntnose minnow	Cyprinidae	*Pimephales notatus*
rainbow trout	Salmonidae	*Oncorhynchus mykiss*
brown trout	Salmonidae	*Salmo trutta*
brook trout	Salmonidae	*Salvelinus fontinalis*
white bass	Moronidae	*Morone chrysops*
yellow bass	Moronidae	*M. mississippiensis*
striped bass	Moronidae	*M. saxatilis*
hybrid bass	Moronidae	*M. chrysops* x *M. saxatilis*
redbreast sunfish	Centrarchidae	*Lepomis auritus*

Figure 2.1. Land Cover in the Etowah in 1974 and 2001. Data source: National Land Cover Database.

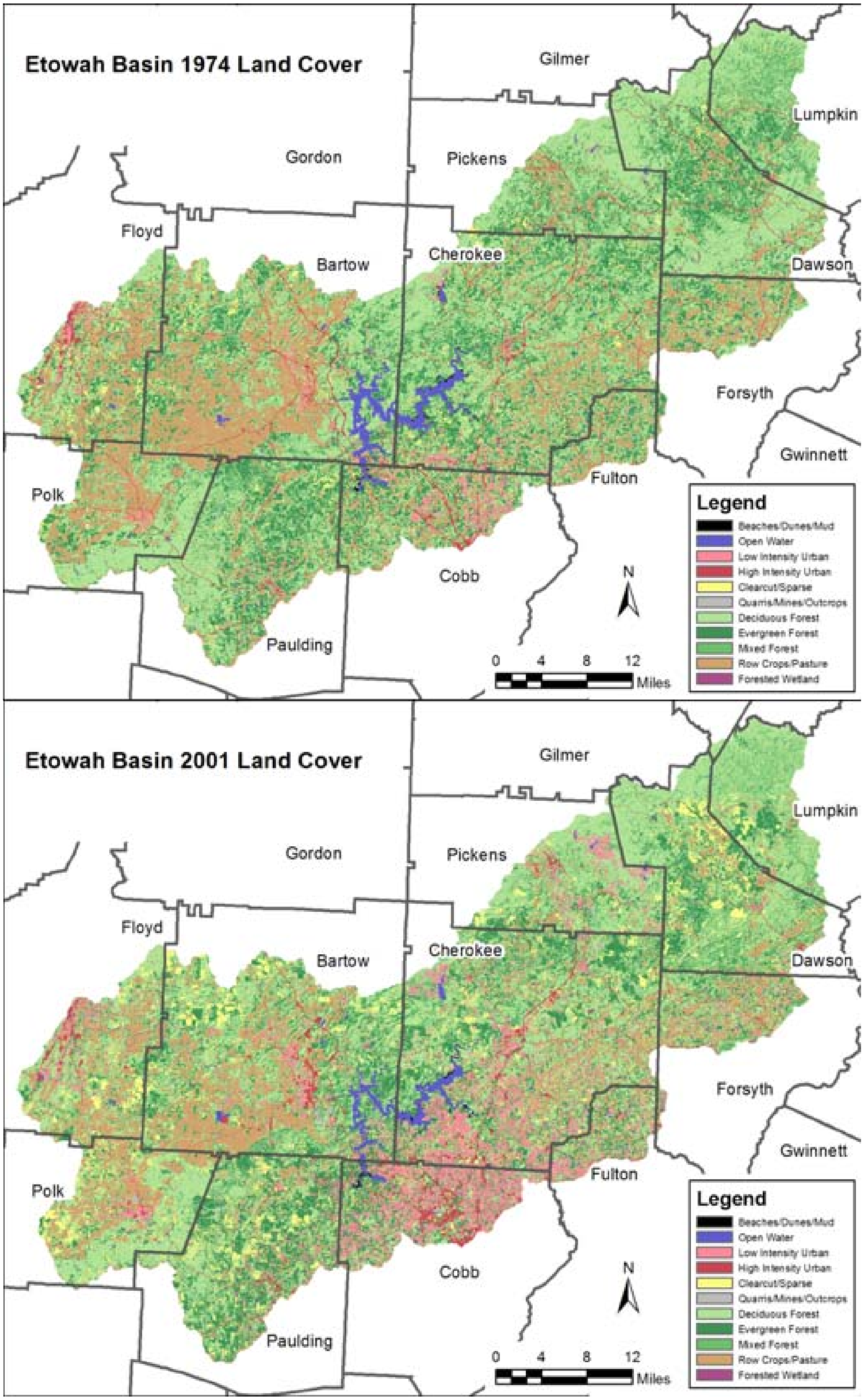


Figure 2.2. Influence diagram showing how increased sediment affects sensitive fish species in the Etowah Basin. Sources are shown in red, stressors in yellow, mechanisms in blue, and affected vital rates of fish in green.

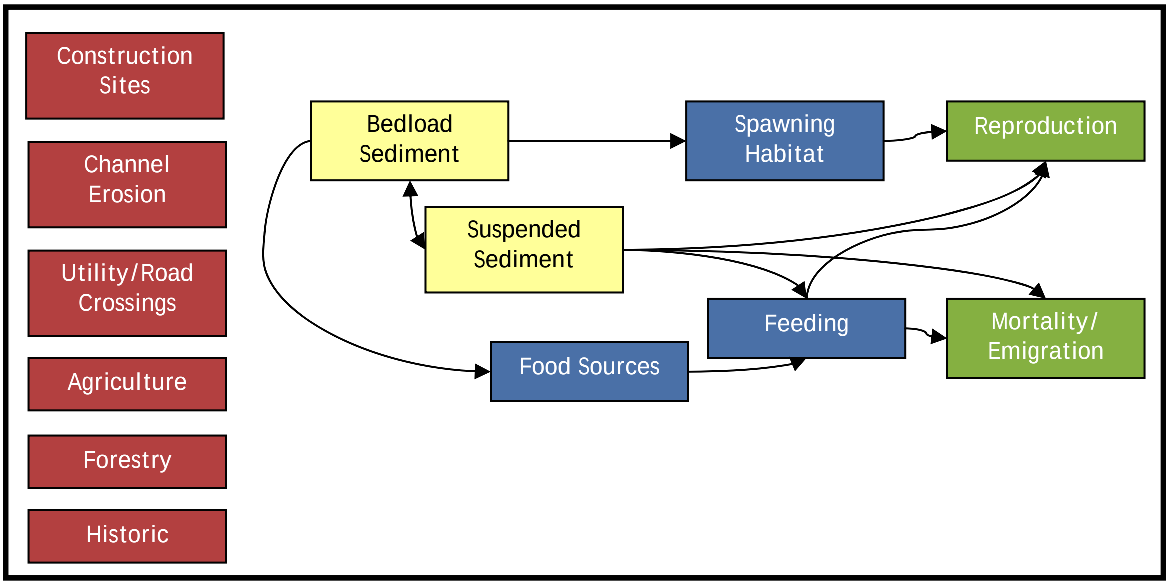


Figure 2.3. Influence diagram showing how hydrologic alteration affects sensitive fish species in the Etowah Basin. Sources are shown in red, stressors in yellow, mechanisms in blue, and affected vital rates of the fish in green.

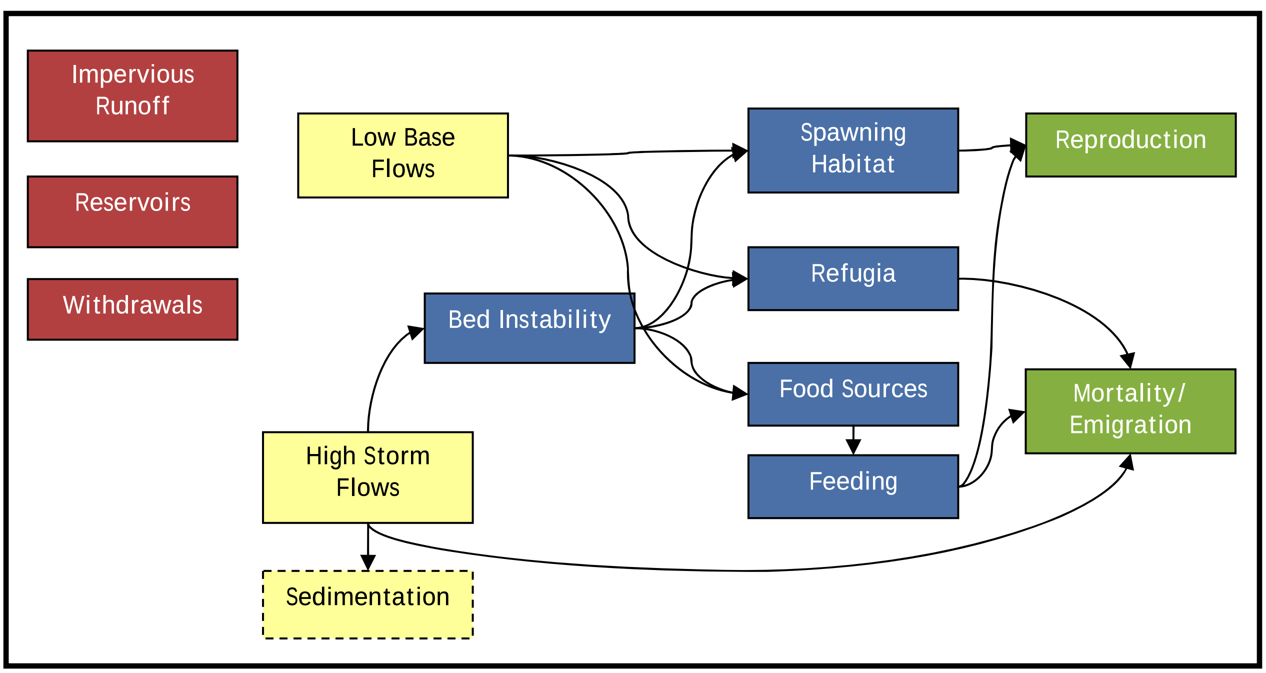


Figure 2.4. Diagram of flow response to rainfall (heavy bars) in a stream draining a forested watershed (solid line) versus a stream draining an urban watershed (dashed line). From Walsh et al. (2004a).

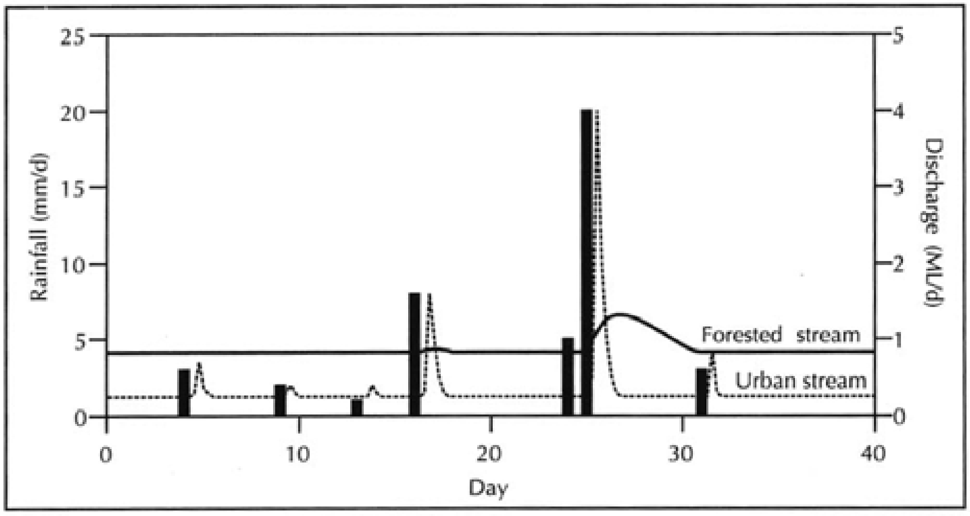


Figure 2.5. Probability of occurrence of the Etowah darter in response to increasing effective impervious area (EIA). Black line represents a large stream; gray line, a mid-sized stream.

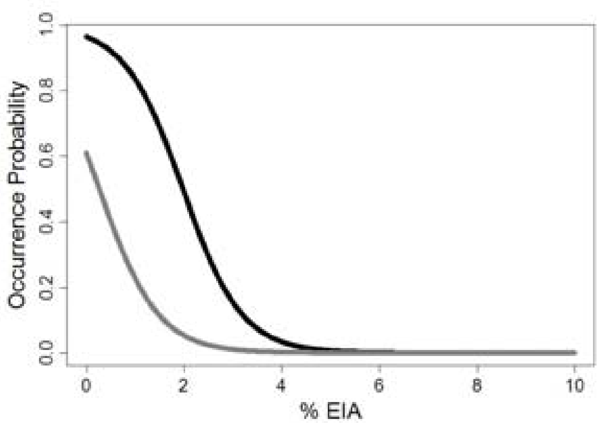


Figure 2.6. Influence diagram showing how extensive riparian buffer loss affects sensitive fish species in the Etowah Basin. Sources are shown in red, stressors in yellow, mechanisms in blue, and affected vital rates of the fish in green.

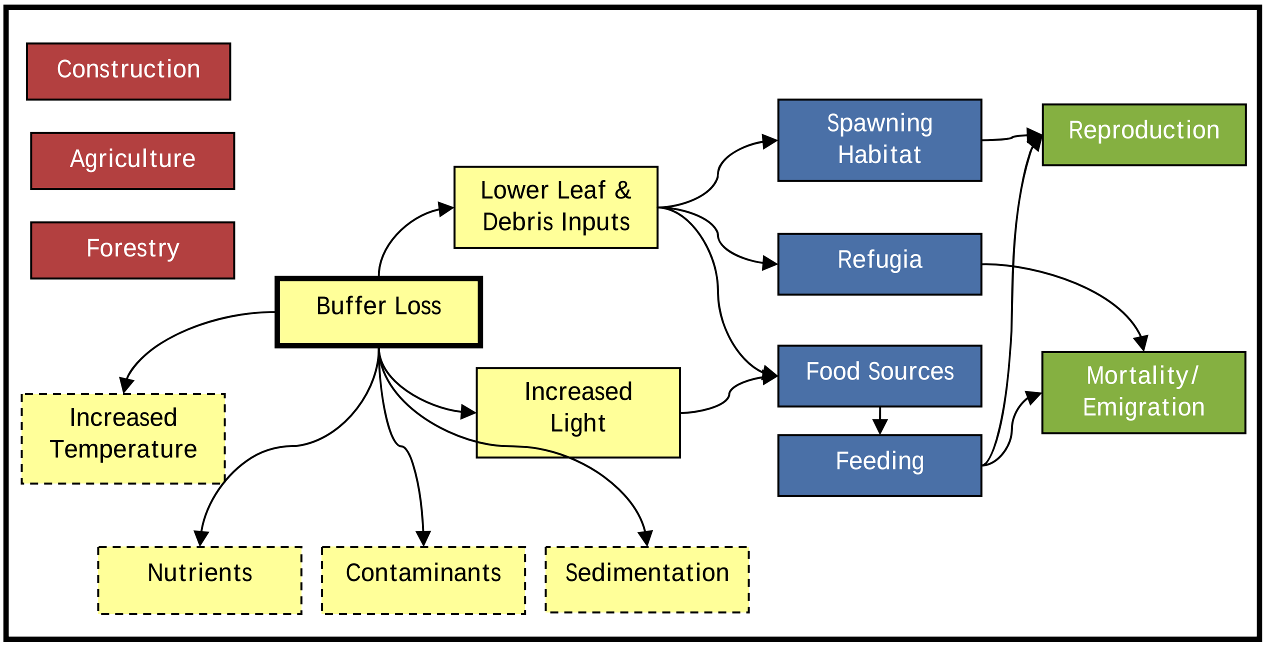
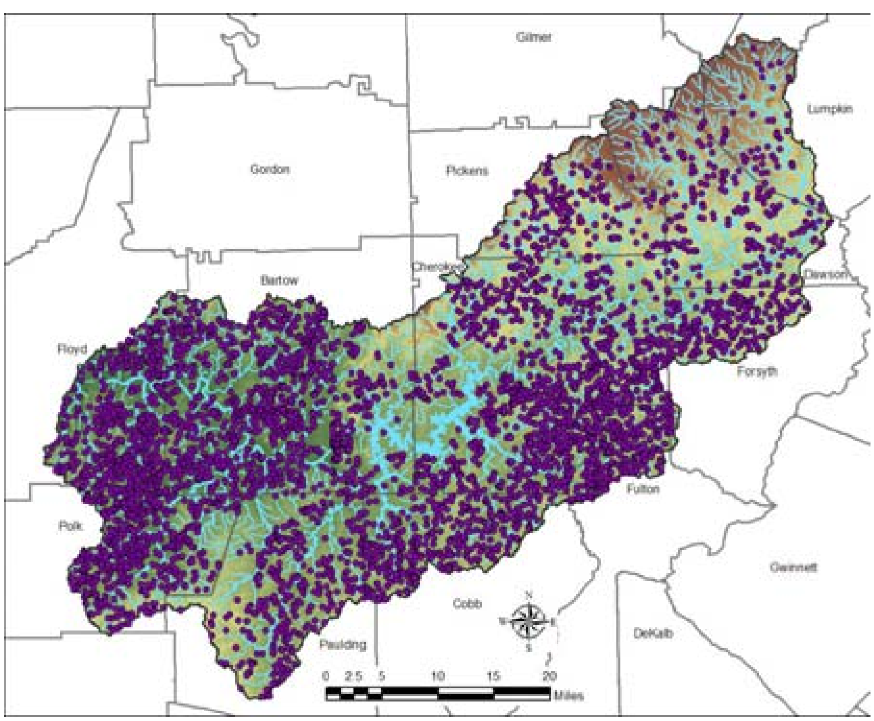


Figure 2.7. Reservoirs and ponds in the Etowah basin. Digitized from USGS topographic maps and aerial photos.



CHAPTER 3

EFFECT OF IMPERVIOUS COVER ON STREAM FISHES AFTER ACCOUNTING FOR HISTORIC LAND USE AND HYDROGEOMORPHIC FACTORS[3]

[3] Wenger, S.J., J.T. Peterson, M.C. Freeman, B.J. Freeman and D.D.Homans. To be submitted to *Canadian Journal of Fisheries and Aquatic Sciences*

ABSTRACT

Stormwater runoff from impervious cover threatens many stream fishes, including rare and endemic species. To date, few studies have examined the response of individual fish species to increasing impervious cover, although such models are essential for effective management and conservation efforts. Relationships between imperviousness and species occurrence are potentially confounded by other explanatory variables, including hydrogeomorphic characteristics (e.g., stream size, elevation, geology) and historic land use (e.g., agriculture, impoundments). We compared models predicting occurrence as a function of (1) hydrogeomorphic characteristics, (2) hydrogeomorphic characteristics plus historic land use, (3) hydrogeomorphic characteristics plus current land use (especially effective impervious area), and (4) hydrogeomorphic characteristics plus historic and current land use. We used data for five species from 357 fish collections made in the Etowah Basin, Georgia, USA, between 1998 and 2003. For four of five species, the best-supported models were those that included both current effective impervious area (EIA) and historic land use predictor variables. For the best supported models for three species, occurrence probability was predicted to approach zero when effective impervious area reached two to four percent, based on the USGS impervious map for the region. This may be approximately equivalent to the generally-cited threshold of 10% TIA. Our results illustrate the large potential effect of impervious cover on persistence of aquatic species, and an approach for quantifying species-specific responses to a landscape change while accounting for hydrogeomorphic and historic factors.

INTRODUCTION

Many studies have demonstrated that fish assemblage structure is related to a gradient of urbanization (Helms et al. 2005, Klein 1979, Meador et al. 2005, Morgan and Cushman 2005, Onorato et al. 2000, Roy et al. 2005, Walters et al. 2005, Walters et al. 2003a, Wang et al. 2001, Wang et al. 2000). Most of these studies have used characteristics of the fish assemblage—such as an index of biotic integrity (IBI), species diversity or a ratio of homogenization—as response variables (but see Olden 2003, Walsh et al. 2004b, the latter of which is for an amphipod). The limitation of assemblage-level analyses is that they do not provide information about the response of individual species, especially rare ones. This can be a matter of significant interest in the management of imperiled fish species. As urban land cover increases globally, a growing number of species will be impacted by urbanization, and knowledge of species-specific relationships between indicators of urban cover and fish occurrence or abundance will be essential to develop effective conservation strategies.

In the absence of sufficient long-term data sets relating changes in fish occurrence to land use, most studies employ a space-for-time substitution. That is, variation in species occurrence over space is related to variation in land use patterns at a fixed point in time. The space-for-time substitution requires the assumption that observed species distribution patterns are due to the effects of contemporary land use patterns. However, this assumption may not be supported. Contemporary fish distributions are likely the result of complex interactions between (1) hydrogeomorphic characteristics of streams, the landscape, and other biota (Allan 2004); and (2) past human land use activities (Harding et al. 1998), both of which are often strongly related. For example, cities tend to be built on low-slope, formerly agricultural land, rather than high-slope, formerly forested land. If we fail to account for the influence of hydrogeomorphic influences and historic land use, we run the risk of misinterpreting the role of current land use.

This study involves fish species of the Etowah River basin, Georgia, USA. Previous studies in the Etowah basin have identified a relationship between urban cover and fish assemblage attributes. Walters et al. (2003a) found evidence that urbanization leads to homogenization of fish communities in the Etowah. That study considered the relationship of fish occurrence with both hydrogeomorphic predictor variables and urban cover, and found that local stream gradient was a dominant factor controlling stream habitat and fish assemblages at the local scale, although there was a clear correlation between urban cover and assemblage homogenization (Walters et al. 2003a, Walters et al. 2003b). A subsequent study evaluated hydrologic alteration as a mechanism for changes in the fish assemblage, and found correlations between several measures of hydrologic change and fish assemblage characteristics (Roy et al. 2005). However, neither study evaluated whether the observed patterns could have arisen as a result of historic land use. Like other areas in the Georgia Piedmont, much of the Etowah experienced intense row-crop agriculture from the 1800s to the early 1900s. The agricultural practices caused massive erosion and the sedimentation of stream valleys (Trimble 1974), which could have led to extirpation of sensitive fish species from many tributary systems. Subsequently, many impoundments were constructed across the basin, which may have prevented recolonization and otherwise influenced (and may continue to influence) fish distributions.

We focus on impervious area as an indicator of urban cover. A recent review identified stormwater runoff from impervious surfaces as the primary source of stressors to urban streams

(Walsh et al. 2005b). Thus, there is a mechanistic connection between impervious area and fish occurrence. Previous researchers have suggested that the most problematic impervious surfaces are those that are directly connected to streams via drainage and conveyance systems (Alley and Veenhuis 1983, Booth and Jackson 1997, Walsh et al. 2005a, Walsh et al. 2004b). Studies have demonstrated that this effective impervious area (EIA) is a better predictor of stream biological and chemical response than total impervious area (TIA) (e.g., Hatt et al. 2004, Walsh et al. 2004b, Wang et al. 2001).

In this study, our first objective is to examine the distributions of five potentially sensitive fish species to determine whether a relationship with current indicators of urbanization (EIA, TIA or other) is detectable once we account for the influence of hydrogeomorphic stream characteristics and historic land use. We do this by testing four hypotheses:

1) Species occurrence is best explained by a model with only hydrogeomorphic predictor variables (watershed area, elevation, slope, etc.);
2) Species occurrence is best explained by a model with hydrogeomorphic variables and an historic land use predictor variable;
3) Species occurrence is best explained by a model with hydrogeomorphic variables and a current land use (EIA, TIA or other) predictor variable;
4) Species occurrence is best explained by a combination of hydrogeomorphic, historic, and current land use predictor variables.

If urbanization proves a useful predictor of current species occurrence (i.e., if hypothesis 3 or 4 is supported), our second objective is to develop predictive models for evaluating the likely effects of increased urbanization on species occurrence at local (stream reach) scales. These models can then be used as part of predictive tools for evaluating the influence of future growth scenarios and management strategies on the status of at-risk fishes.

METHODS

Study Area and Species

The Etowah River is a major tributary of the Coosa River system in the Mobile River Basin (Figure 3.1). It drains 4871 km^2 of land, with the upper half of the basin primarily in the Piedmont physiographic province (with a small area of Blue Ridge in the headwaters), and the lower half of the basin primarily in the Valley and Ridge physiographic province. A portion of the Etowah lies within the Southern Appalachian highlands, a global hotspot of fish endemism (Warren et al. 2000). The basin supports a diverse aquatic fauna, with 76 extant native species of fish (Burkhead et al. 1997), including three that are listed under the Endangered Species Act and six others that are considered imperiled but not currently listed. At least four of these species are endemic to the basin. A significant threat facing these organisms is rapid urbanization from the Metropolitan Atlanta region (Chapter 2). The region added more people in the 1990s than any other part of the U.S. except Los Angeles, and continues to add nearly 100,000 people per year (Frankston 2005, McCosh 2000). Urban and suburban land cover are increasing rapidly in the Etowah, while agricultural and forestry land cover are declining (Kramer 2004). In response to these concerns, the local governments of the Etowah watershed have begun a process to develop the Etowah Habitat Conservation Plan or Etowah HCP (Etowah HCP Advisory Committee 2006). The purpose of the plan is to implement a set of growth management policies and ordinances that minimize the impact of future development on the aquatic fauna, thus permitting additional growth without threatening the persistence of federally protected organisms.

We examined five species native to the Etowah system hypothesized to be sensitive to urbanization (Table 3.1). These species were selected on the basis of previously published (Roy et al. 2005, Walters et al. 2003b) and unpublished studies, some of which involved a subset (61 sites) of the data used for our study. Two of the species (*Etheostoma etowahae* and *Etheostoma scotti)* are federally listed and targets of the Etowah HCP.

Data Preparation

Fish Collections. We selected 357 records of fish collections from a database maintained by the Georgia Museum of Natural History. We used collections made in the Etowah basin between January 1, 1999 and December 31, 2003, which we considered approximately contemporaneous with the available "current" land cover data (see below). We selected only collections intended to characterize the full assemblage of sampled habitats using electroshocking, kick-seining, and seine hauling. Some of the data were used in previously published studies (Roy et al. 2005, Walters et al. 2003a). We excluded collections from streams draining less than 0.5 km^2 and those of uncertain reliability, which included collections targeting only certain species, collections that appeared to be missing information and collections where notes indicated that an incomplete or low-effort sample was taken. Sample reaches at sites were 50m to 200m in length. Collections from localities that were very close together (less than 0.5 km apart within the same stream, without large intervening tributaries) were assumed to be from the same site. However, collections from the same locality but more than two years apart were treated as if they were from independent sites with regard to estimating detection probability, under the assumption that populations could not be considered "closed" across this time (see below). With these adjustments, the final data set included 252 distinct sites, each sampled from one to five times.

We selected an additional set of 62 records for collections made at 31 sites to provide supplementary data for estimating species-specific probability of detection (following MacKenzie et al. 2002). We used sites where collections were made between Jan 1, 1990 and December 31, 1998 that were sampled twice within two years or three times in three years (temporal replicates); we also selected pairs of sites that were immediately adjacent and were sampled within a day of one another (spatial replicates) within this time period. It was assumed that a species was either present or absent for both samplings for each set of replicates; i.e., that the populations were closed.

Hydrogeomorphic Predictor Variables. For each collection site, we delineated the watershed that drained to the site and assigned it to one of 21 tributary systems (Figure 3.1). We derived seven hydrogeomorphic predictor variables: watershed area, downstream link magnitude (d-link), elevation, physiographic province, bedrock geology, surficial geology and stream slope (Table 3.2). All were calculated in Arcview 3.3 or ArcGIS 9.0 software. Watershed area was calculated as the total area draining to the collection site and served as an indicator of stream size at the fish sampling location. Downstream link magnitude was used as a way of describing a stream reach's position in a watershed—whether it was a headwater stream or directly connected to larger mainstem streams, for example. It was calculated as the number of unbranched streams draining to the next confluence downstream of the site, using 1:24,000 scale maps (Osborne and

Wiley 1992). Elevation was calculated at the collection site from 30 m resolution digital elevation models (DEMs) (US Geological Survey 1988). Physiographic province (Georgia Geologic Survey 1999), bedrock geology summarized by group (Georgia Geologic Survey 1999) and surficial (quaternary) geology (Richmond et al. 1987) were included as candidate measures of the influence of geology on physicochemical properties of streams. Local stream slope data were not available for most collection sites, so we estimated the mean slope of all streams draining at least 10 km^2 at the scale of each system, using 30m DEMs (US Geological Survey 1988), and assigned it to each site in that system.

Impervious Area and Other Measures of Current Land Cover. The 2001 National Land Cover Database Zone 54 Imperviousness layer (US Geological Survey 2003) was used as the source for total impervious area (TIA). This was a raster coverage with a resolution of 30m pixels derived from supervised classification of LandSat satellite imagery. To calculate EIA, we followed Alley and Veenhuis (1983) in developing our own empirical relationship between TIA and EIA, which we applied to the TIA coverage. We hand delineated both impervious and directly connected impervious surfaces (which we considered EIA) from high-resolution aerial photographs for 15 sites of 25 to 70 hectares in size. Impervious areas included roofs, roads, parking lots, sidewalks, and any other artificial, non-pervious surfaces distinguishable on the aerial photos. Directly connected impervious surfaces were a subset of impervious surfaces that were visually noted to drain to the stormwater conveyance network. We then determined the relationship between TIA and EIA by fitting the data to different candidate models. The best model, selected on the basis of the coefficient of determination, was linear with a threshold:

(1) EIA = (1.046*TIA) - 6.23%, where EIA= 0 for TIA values less than 6.23% ($R^2 = 0.98$).

We applied this formula to the TIA layer to create a raster EIA layer. For each fish collection site, we then calculated TIA and EIA at five scales: impervious area in the watershed upstream of the site, and impervious area within 0.5 km, 1 km, 1.5 km and 2 km radius of the site (Table 3.2). We also considered the possibility that urban land cover and forested land cover might be better predictor variables than impervious cover for some species. These variables were calculated for the upstream watershed for each site using 2001 land cover data (Kramer 2004) (Table 3.2).

Historic Land Cover. We investigated three candidate indicators of historic land cover, each measured at two scales. The first was historic modified land cover in the basin, which was quantified from 1938 aerial photographs. These were the oldest aerial photographs available for the entire region, and the best representation we could find of land use from the era of cotton production. We georectified scans of 1938 Agricultural Stabilization and Conservation Service (ASCS) 1:100,000 scale aerial photograph index sheets from the Georgia Aerial Photographs website (http://dbs.galib.uga.edu/gaph/html/). We classified the resulting images into forested areas and agricultural/developed land based on cell brightness.

The other candidate indicators of historic land cover were the number and area of reservoirs, which we expected to correlate with historic land use. Many reservoirs were built in the 1950s through the 1970s on agricultural lands, and these can be viewed both as indicators of agricultural influences and as potential stressors. All indicators of historic land cover were measured at two scales: (1) the watershed above each collection site, and (2) the tributary system within which the collection site was nested (Table 3.2).

Data Analysis

One of our goals in the modeling was to obtain covariate parameter estimates with minimal bias by accounting for spatial dependencies in the data and incomplete detectability of species. Failure to account for spatial correlations can lead to underestimates of the variance of parameter estimates (Snijders and Bosker 1999), while failure to correct for incomplete detection can lead to bias in the means of parameter estimates (Gu and Swihart 2004). We constructed logistic regression models using two-level hierarchical modeling to manage spatial correlations (Snijders and Bosker 1999), following the general approach for modeling species distributions outlined by Latimer et al. (2006). We adapted this to account for incomplete detectability using a species occupancy model (MacKenzie et al. 2002). While incorporating both hierarchy and incomplete detectability into a logistic regression model presents significant challenges for conventional maximum likelihood estimation, Monte Carlo Markov Chain (MCMC) model fitting techniques are able to accommodate such complexity (Conroy et al. 2005, Peterson et al. 2005). The disadvantage of MCMC techniques is that model-fitting is computer intensive and time-consuming. Therefore, we used a two-stage modeling approach, similar to the approach used by Howell et al. (*in review*):

(1) We screened potential predictor variables representing hydrogeomorphic characteristics, historic land use, and current land use with ordinary logistic regression that assumed complete detectability.

(2) We evaluated the relative fit of the best supported models from the initial screening using hierarchical species occupancy models.

Screening of Candidate Predictor Variables

For the candidate predictor variable selection, we ignored spatial dependencies and assumed complete detectability. For sites with multiple collections, we assumed a species was present if it was encountered in any of the collections. To increase linearity in the predictor variables, watershed area was square root-transformed and d-link was natural log-transformed. All continuous predictors were normalized with a mean 0 and standard deviation of 1 and we included quadratic terms for area, d-link and elevation as possible predictors.

We evaluated a series of logistic regression models for each of the five species, with species occurrence as the dependent variable. All were run with the statistical package R 2.0.1 (R Development Core Team 2004). There were four model categories: (a) models with only hydrogeomorphic predictors, or “hydrogeomorphic”, (b) models with hydrogeomorphic predictors plus an historic land use predictor, or “historic”, (c) models with hydrogeomorphic predictors plus a current land use predictor, or “current” and (d) models with hydrogeomorphic predictors plus an historic and a current land use predictor, or “global.” Our goal was to identify the best-fitting model in each category (a through d) for each species, using Akaike’s Information Criterion modified for small sample size (AICc) as the basis for selection (Burnham and Anderson 2002). To identify the most plausible model that included only hydrogeomorphic predictor variables, we fitted a model with all hydrogeomorphic predictors and then performed a series of stepwise removals, selecting the three best-supported models based on AICc. Next, we compared 18 models, each of which was based on one of the three best-supported hydrogeomorphic predictor models and included one of the six candidate historic land cover predictor variables. We again performed a stepwise removal of variables to determine if a reduced model was better supported. We then repeated the process for the 12 candidate predictor variables for current land use (comparing 36 models). Finally, we compared models with different combinations of both an historic predictor variable and a current predictor variable, to select the best global model. We compared the best supported model in each category to estimate which was best supported overall.

Hierarchical occupancy modeling

To account for incomplete detection, we jointly modeled species presence and detectability as:

(2) $P(d) = P(d|\psi)\ P(\psi)$

where P(d) is the probability that the species is present and detected at the site; $P(\psi)$ is the probability of species presence, and $P(d|\psi)$ is the probability that the species is detected given that it is present—i.e., detectability. Detection probability was calculated using a site occupancy model (MacKenzie et al. 2002) based on the encounter history of the species at sites sampled multiple times. In our data set, 63 sites were sampled a second time, 21 of those were sampled a third time, 13 of those were sampled a fourth time and 8 were sampled a fifth time (the remaining 189 sites were sampled once each). To improve our estimate of detectability, we extended the dataset to include an additional 31 sites collected at least twice prior to 1998. These sites were allowed to influence the estimate of $P(d|\psi)$ but not any of the other model parameters. Note that this approach assumed closure, which was likely violated in some cases, so the method was expected to slightly underestimate detectability (since any local colonization and extirpation events would be incorrectly attributed to non-detection). It is possible to include covariates on detectability to account for differences in sample effort and methods, but we did not do so because we lacked relevant collection-level data across all samples.

Prior to running the full models, we tested for spatial autocorrelation at the level of the tributary systems by performing an analysis of variance on the residuals of each of the best-supported screening models, using the tributary systems as treatments. Since we detected significant dependence among the tributary systems ($\underline{P} < 0.001$), we defined a two-level hierarchical structure with sites nested within the 21 tributary systems. We implemented this by adding a normally distributed random effect at level two (Snijders and Bosker 1999). Level one of the model can be represented as:

$$(3) \qquad Logit\left(P(\psi_{i,j})\right) = \beta_{0j} + \sum_{s=1}^{m} \beta_s x_{s,i,j}$$

where $x_{s,i,j}$ are s = 1, 2 …*m* predictors for site *i* within tributary system *j*. The intercept then is modeled as a function of tributary system characteristics (level two):

$$(4) \qquad \beta_{0j} = \gamma_0 + \sum_{r=1}^{n} \gamma_r w_{r,j} + \delta_j \,,$$

where $w_{r,j}$ are the r = 1,2,…*n* predictors corresponding to tributary system *j* and a random effect δ_j that varies normally among reaches with a mean zero and variance σ_j^2.

For each of the five species, we fit the best-supported screening model in each of the four categories (hydrogeomorphic, historic, current and global) to the hierarchical occupancy models. If the screening analyses showed that the second- or third-best models in a category also had considerable support, we also fit these to the hierarchical occupancy models. We used Monte

Carlo Markov Chain (MCMC) methods as implemented in WinBUGS 1.4 (Spiegelhalter et al. 2003) for all hierarchical occupancy modeling. We ran six parallel chains and tested each model for convergence using the Gelman- Rubin diagnostic (Gelman and Rubin 1992). Models converged within 8000 iterations, and the values from this "burn in" period were discarded. Models were then run for a further 60,000 iterations to estimate parameters and deviance. We used diffuse priors for all parameters. To reduce MCMC autocorrelation, models were thinned by a factor of 10, which means that only every 10th sample was used in calculating statistics. The use of this technique greatly reduced autocorrelation, but did not eliminate it in all cases. We tested increased iterations with even greater thinning, up to 600,000 iterations with 100x thinning, but parameter estimates, deviance and convergence diagnostics remained stable throughout the range of iterations evaluated. Therefore, we considered 10x thinning adequate.

We used three-fold cross validation to select models and estimate their out-of-sample predictive performance. For test sites we assumed that tributary system membership was unknown. We ranked models by their predictive performance using the area under the curve (AUC) of the receiver-operating characteristic (ROC) plot as a summary statistic. The ROC curve is the ratio of true positives to false positives when the species occurrence decision threshold is varied between zero and one; the AUC of the ROC curve is considered a robust measure that is invariant to species prevalence (Fielding and Bell 1997, Latimer et al. 2006, Manel et al. 2001, Olden et al. 2002).

We found that for models with large variances on the random effect, fixed effect parameter estimates were proportionately large. This behavior is a result of the fixed level one variance of logistic regression models, which leads to inflation of parameter estimates in order to maintain proportionality between levels one (e.g., stream site) and two (e.g., tributary) when random effects are added (Snijders and Bosker 1999). We corrected for this phenomenon by standardizing the parameter estimates of each model by the sum of level one and level two variances. These standardized values were used to calculate odds ratios for the mean and 90% credible intervals for the fixed effect parameter estimates for all variables of the best supported current, historic and global models.

RESULTS

Impervious cover, historic land use and reservoir density varied considerably across the basin and among tributary systems (Figures 3.1 - 3.4). This is reflected in the high variance of most variables across the collection sites/watersheds (Table 3.2).

In a few of the screening models, we observed complete or quasi-complete separation in one or more of the class variables. That is, a response (species presence or absence) and predictor were perfectly related (for complete separation) or nearly so (for quasi-complete separation). When this occurred, parameters could not be estimated. We refit well-supported models with the offending variables removed.

Based on the screening model analysis, we selected 7-14 models to run for each species in the hierarchical occupancy modeling (Table 3.3). Among hydrogeomorphic predictor variables evaluated, watershed area, downstream link magnitude, tributary system slope and elevation were the most commonly included in the best fitting models. Among historic land use predictors, area inundated by impoundments was the most commonly included, but historic land use was included for at least some models for two species. Effective impervious area (EIA) within 500 m to 1.5 km of the collection site was the most common measure of current land use selected for inclusion in models, based on the screening analysis. The only exception was *E. scotti*, for which forest cover in the watershed was the best predictor.

Gelman-Rubin convergence diagnostics showed that all of the hierarchical occupancy models converged. Based on the AUC values, the best model for each species was a global model, with the exception of *E. scotti*, for which an historic model (i.e., one without a predictor variable for current land use) was best supported (Table 3.3). For *N. leptacanthus*, *E. etowahae* and *P. palmaris* the best current land use model (i.e., one without a predictor variable for historic land use) ranked above the best historic land use model. For all species, the best hydrogeomorphic-only model was an inferior predictor to the best global, historic, and current models. According to a rule of thumb for AUC scores (Swets 1988), models with AUC values > 0.9 have high accuracy. The best models for three of the species met this threshold, although those for *N. leptacanthus* were slightly lower and those for *E. scotti* were substantially lower (Table 3.3).

The species with the strongest response to current land use was *C. trichroistia* (Table 3.4). Using the best predicting model, we estimate that the species was almost 20 times less likely to occur for each 1% increase in EIA within 1.5 km (note that this was not a relative increase in EIA, but an absolute increase; e.g., a change from 5% to 6% would be a 1% increase). Although the precision of this value was low, the 90% credible interval indicated that the species was at least 3.5 times less likely to occur for each 1% increase in EIA. Occurrence probability approached zero when EIA exceeded about 2%, when other predictor variables were held to their mean values (Figure 3.5). The presence of *P. palmaris* and *E. etowahae* also was strongly negatively related to EIA, although the 90% credible interval for *E. etowahae* almost included zero (Table 3.4). For both species, we predict that the occurrence probabilities approach zero at 4% EIA and above (Figure 3.5). *N. leptacanthus* showed a weaker relationship with EIA, and *E. scotti* showed essentially no relationship, with the mean credible interval for the odds ratio centered near one and broadly overlapping on either side (Table 3.4, Figure 3.6). We interpret the latter as a lack of a biologically important relationship between current land use and *E. scotti* occurrence.

Historic land use was strongly related to the current presence of four species (Table 3.4). Under the mean values of the best supported models for *C. trichroistia*, *E. etowahae* and *P. palmaris*, species were 1.7 to 2.5 times less likely to occur for each increase of 0.25% in the area of upstream watershed or tributary system that were impounded. *N. leptacanthus* was 2.3 times less likely to occur for each 10% increase in the area of the upstream watershed in historic modified land cover, but the credible interval of the odds ratio was very wide and included one, indicating large uncertainty around the species' response. The odds ratio credible interval for *E. scotti* was centered near 1, suggesting no biologically important relationship.

The current presence of four of the five species was positively related to watershed area and downstream link magnitude; the exception was *E. scotti*, which was negatively related to watershed area (Table 3.4). The best supported model for *P. palmaris* indicated that species occurrence was positively related to elevation, whereas *E. scotti* occurrence was negatively related to elevation. Slope was positively related with the occurrence of *C. trichroistia, E.*

etowahae and *P. palmaris*, although in all cases the 90% credible interval overlapped one. The occurrence of *N. leptacanthus* also was strongly negatively related to slope (Table 3.4).

There was unexplained variation at the tributary system level, as indicated by the level 2 random effect variance estimates (Table 3.4). Variation was greatest for *N. leptacanthus* and lowest for *C. trichroistia*. The variability among tributary systems is reflected in shifted intercepts, which affects the relationship of fixed effects (such as EIA) by shifting the curve to the left or right of the overall mean (Figure 3.6, using *E. etowahae* as an example).

DISCUSSION

We found that for four of the five species evaluated, current land use was a good predictor of species occurrence in the Etowah, even after including historic and hydrogeomorphic predictors in models. The hierarchical occupancy models showed that several species responded strongly to low levels of EIA. Many previous studies have reported declines in aquatic fauna in watersheds draining more than 10-12% impervious cover (Klein 1979, Schueler 1994, Wang et al. 2000). Our results indicated that some species become rare at levels as low as 2% EIA, which translates to a TIA of about 3.5-4.0% (note that the level of TIA relative to EIA is lower than expected from equation 1. The transformation of TIA to EIA is applied on a cell-by-cell basis, while the mean value of EIA/TIA is estimated within a radius of the collection point, which encompasses many cells. In areas of low development, most cells have both EIA and TIA values of zero, which reduces the mean difference between EIA and TIA below what is expected. For example, if 60% of cells around a collection point have a TIA of 0 and 40% of cells have a TIA of 0.3, TIA will be 0.4*0.3=0.12, while EIA will be 0.4*0.25=0.10). However, we noted that the impervious coverage map used in this study (US Geological Survey 2003) appeared to underestimate imperviousness relative to values quoted in the literature, despite the fact that the map is the standard source for landscape-scale impervious data for the Southeast. Examination of aerial photos confirmed that many rooftops, parking lots and roads did not appear on the impervious surface coverage, especially in forested portions of the basin. As an example, we estimated the mean impervious coverage for residential development on two-acre lots at 2.0% EIA and 3.7% TIA, based on the impervious cover map. However, published estimates of TIA for a two-acre lot-size residential development average 10.6% (Capiella and Brown 2001). This suggests to us that our finding that sensitive fish species become rare at levels of 2% EIA, measured with the regional map that appears to be biased low, may actually be consistent with the oft-cited threshold of 10% TIA, measured accurately. The apparent bias in the standard impervious cover map for the region is somewhat troubling. We recommend that studies always report the source of impervious cover data, and that caution be exercised in comparing TIA and EIA values across studies.

In previous studies, *C. trichroistia, P. palmaris, E. etowahae* and *E. scotti* were included in metrics of endemic species richness, which were found to respond negatively to increasing urbanization (Walters et al. 2003a) and hydrologic alteration associated with imperviousness (Roy et al. 2005). Our results provide strong evidence that the first three species are indeed sensitive to these stressors, but that occurrence of *E. scotti* is not strongly related to current land use. We also found that *N. leptacanthus* appeared to be influenced by imperviousness, despite the fact that it has previously been included among metrics of cosmopolitan species (i.e., species which as a group responded neutrally or positively to urban impacts; Walters et al. 2003a). This is a reminder that species groupings based on traits and classifications—such as endemics and cosmopolitans, or any of various IBI metrics—may contain considerable noise in the form of species that respond in the direction opposite to what is expected. Membership in such a group should not be assumed to indicate sensitivity or tolerance.

It was interesting to note that, based on the screening models, impervious area within a radius of the collection site was in nearly all cases a better predictor than impervious area in the watershed above the collection site. This implies that at least some impacts of urbanization are local in effect and dissipate with distance downstream. Since the radii included both downstream aand upstream reaches, as well as tributaries to downstream reaches, this also suggests that the presence of a species at a locality may be influenced by the quality of habitat in neighboring reaches. This finding is especially interesting considering that most previous studies have used an indicator of urbanization measured in the watershed upstream of the collection point (Helms et al. 2005, Klein 1979, Meador et al. 2005, Morgan and Cushman 2005, Walters et al. 2003a, Wang et al. 2001, Wang et al. 2000). If our finding is general, it is possible that those studies slightly underestimated the effect of urbanization on fish by measuring it in a suboptimal manner.

Models with historic land use (but not current land use) were good predictors of fish occurrence for many species, but in nearly all cases inferior to models that included current land use predictors. This is in contrast to the study of Harding et al. (1998), which found that 1950s land use was a superior predictor of fish and invertebrate diversity compared to current land use. However, Harding et al. examined only forested and agricultural watersheds, whereas we considered urban watersheds as well. We found that the best models for most species included *both* current and historic land use predictors, providing evidence that current fish distributions are the product of past land use legacies and recent activities, especially urban development. Of the historic variables, the density and area inundated by impoundments were generally better predictors than historic modified land cover as mapped from aerial photos. This may be because the mapped historic land cover was only a snapshot at one point in time, and perhaps not an accurate predictor of the locations that suffered the greatest impacts. Cotton agriculture crashed in the early 20th century with the arrival of the boll weevil (Haney et al. 1996), and by 1938 some agricultural areas may already have been abandoned and reforested. Alternatively, fish may have recolonized some watersheds that were still heavily impacted in 1938, diluting the strength of the relationship we observe today.

It may be somewhat misleading to consider impoundments only as historic influences, because they continue to exert an effect on current fish populations. Reservoirs eliminate fluvial habitat, alter downstream water quality and quantity, block fish movements, and have been shown to alter downstream fish assemblages (Collier et al. 1996, Freeman and Marcinek 2006). It is impossible to tell whether the observed negative relationships between fish species presence and impoundments are due to historic or current factors. We suspect both are important. In our data set, the number of dams in the tributary system is highly correlated with the historic modified land cover in the tributary system (r=0.88). However, the area inundated by impoundments in the collection watershed, which was a good predictor for many species, is relatively uncorrelated with historic modified land cover (r=0.34). Models of occurrence that include both historic land use and area inundated by impoundments might prove useful for many species.

The appearance of slope in the best-supported models for most species is consistent with the findings of previous studies in the Etowah, which identified slope (i.e., stream channel gradient) as a critical variable controlling the distribution of many species (Walters et al. 2003a, Walters et al. 2003b). The strengths of the relationships are somewhat remarkable considering that we only were able to use map slope, rather than field-measured reach slope, which was unavailable for many sites. We measured slope at the tributary system scale, under the assumption that tributary systems with lower average slope had less riffle habitat, which made these systems less suited to riffle-dwelling species such as the five fishes modeled here. Thus, at the tributary system scale mean slope is a potential filtering mechanism, determining whether fish species are likely to be present or absent from the system as a whole (we also hypothesized that low slope could indicate high suitability for agriculture, but we found no relationship between slope and historic land cover at the tributary system scale; r=0.2). Reach-scale slope may serve as a second filter, determining whether a species is locally present, given its presence in the tributary system. Thus, measuring slope at the site level would likely yield a stronger relationship between slope and occurrence because it would account for filtering at both scales.

This study is one of the first to quantify the relationship between EIA and the occurrence of individual fish species, and the only one we know of which addresses historic land use and other confounding factors in testing these relationships. Nevertheless, we note a few limitations of our approach. First, because model selection depended heavily on the simplified screening models, in some cases we may have omitted some useful parameters from the selected models, or used a sub-optimal measure of current or historic land use. Second, we considered only relatively simple models, without interaction terms. This decision was motivated by a desire to minimize the number of models in the candidate set and to yield parameter estimates for current land use effects that were readily interpretable. Third, by using species presence rather than abundance as the response variable, we potentially lost useful information; this was a tradeoff of constructing a large dataset from diverse sources, with the result that that abundance data were not comparable across collections.

There are two major ways whereby stormwater runoff from impervious surfaces can impact fishes. Runoff can contribute to hydrologic alteration, which in turn can lead to numerous other stressors, and it can serve as a vector for contaminants (Walsh et al. 2005b, Walsh et al. 2001; see also review in Chapter 2). Our study does not reveal mechanistic relationships. A previous study in the Etowah investigated the role of hydrologic alteration in fish assemblage changes (Roy et al. 2005), and found that hydrologic variables explained 22-66% of variation in fish assemblage richness and abundance. Additional studies are necessary to further elucidate the relative contributions of hydrologic alteration and contaminants in the decline of sensitive fish species. Fortunately, however, management action need not await the results of these studies, because both hydrologic alteration and contaminants can be managed by adopting progressive stormwater management techniques that maximize stormwater infiltration (Walsh et al. 2005a, Walsh et al. 2004a). This is the approach being pursued as part of the Etowah HCP. Properly implemented in an appropriate adaptive management framework, it has the potential both to allow future development and to ensure the continued survival of sensitive fish species in the Etowah.

CONCLUSIONS

This study demonstrates that a combination of historic land use and current effective impervious cover are important in explaining the observed distributions of at least some species of fish in an urbanizing watershed. Several of the fishes evaluated show a strong response to low levels of impervious cover, even after accounting for hydrogeomorphic variables and historic land use. Species respond more strongly to local impervious cover than to upstream watershed impervious cover. This is the first study we know of to quantify the response of individual fish species to increasing imperviousness while accounting for historic land use. The results lend themselves to immediate policy applications and represent a critical step toward effective management of fishes threatened by urbanization.

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Table 3.1. Species analyzed for occurrence in relation to hydrogeomorphic variables, historic land use and current land use. Status follows Warren et al. 2000: currently stable (CS), threatened (T), and endangered (E).

Species	Family	Distribution	Status
Cyprinella trichroistia (Jordan and Gilbert), Tricolor Shiner	Cyprinidae	Mobile River Basin, AL, GA, TN	CS; sensitive to increasing turbidity[1]; representative of other sensitive minnows
Noturus leptacanthus (Jordan), Speckled Madtom	Ictaluridae	Atlantic and Gulf Slope drainages, SC to LA	CS; potential surrogate for *N. sp.cf N.munitus*, Coosa madtom (T)
Etheostoma etowahae Wood and Mayden, Etowah Darter	Percidae	Etowah River system, GA	E; Federally listed
Etheostoma scotti Bauer, Etnier and Burkhead, Cherokee Darter	Percidae	Etowah River system, GA	T; Federally listed
Percina palmaris (Bailey), Bronze Darter	Percidae	Coosa and Tallapoosa river systems, AL,GA	CS; potential surrogate for *P. antesella* Williams and Etnier, Amber darter (E)

[1] Burkhead and Jelks 2001

Table 3.2. Summary statistics on continuous predictor variables measured for 252 collection sites used in models of species occurrence.

Parameter	Abbrev.	Mean	SD	Min	Max
Watershed area (km^2)	area	9.3	28.3	0.5	1591
Elevation (m a.s.l.)	elev	305.01	53.06	207.43	536.52
Downstream link magnitude	dlink	56.5	118.4	0.00	415
Density of dams in watershed (#/km2)	damw	0.98	0.87	0.00	5.95
% of watershed in impoundments	waterw	0.70	0.64	0.00	3.73
% watershed in intense historic land use	histw	33.39	22.48	0.00	90.69
% watershed urban cover	urbanw	16.11	17.60	0.00	73.12
% watershed forest cover	forestw	65.12	17.74	0.00	99.15
% watershed TIA	tiaw	3.83	5.55	0.01	28.37
% TIA in 500m radius	tia500	3.32	5.39	0.00	29.78
% TIA in 1 km radius	tia1	3.47	5.41	0.00	28.89
% TIA in 1.5 km radius	tia15	3.57	5.23	0.00	26.37
% TIA in 2 km radius	tia2	3.71	5.26	0.02	24.71
% watershed EIA	eiaw	2.49	4.44	0.00	23.67
% EIA in 500m radius	eia500	2.07	4.33	0.00	26.23
% EIA in 1 km radius	eia1	2.23	4.34	0.00	24.56
% EIA in 1.5 km radius	eia1.5	2.32	4.14	0.00	22.07
% EIA in 2 km radius	eia2	2.44	4.19	0.00	20.41
Mean slope of large streams in tributary system (delta elev./watershed area)	slope	83.75	73.03	14.62	289.64
Density of dams in tributary system (#/km2)	damtr	0.83	0.41	0.29	1.90
% of tributary system in impoundments	watertr	0.58	0.37	0.12	1.35
% tributary system in intense historic land use	histtr	32.37	18.60	4.84	61.47

Table 3.3. Hierarchical occupancy models for each species with model selection statistics. The best model in each category for each species is shown in bold. Models are shown sorted from best to worst fitting based on AUC of ROC. Variable abbreviations are defined in Table 3.1.

Model No.	Category	Predictor variables	AUC
Cyprinella trichroistia			
2	**Global**	**area, area^2, dlink, dlink^2 , slope, waterw, eia1.5**	**0.933**
1	Global	area, area^2, dlink, dlink^2, waterw, eia1.5	0.931
4	Global	area, area^2, dlink, dlink^2, histtr, eia1.5	0.929
3	Global	area, area^2, dlink, dlink^2 , slope, histtr, eia1.5	0.926
5	**Current**	**area, area^2, dlink, dlink^2, eia1.5**	**0.922**
6	Current	area, area^2, dlink, dlink^2, slope, eia1.5	0.922
8	**Historic**	**area, area^2, dlink, dlink^2, histtr**	**0.915**
7	Historic	area, area^2, dlink, dlink^2, slope, histtr	0.904
9	Historic	area, area^2, dlink, dlink^2, waterw	0.875
10	**Hydrogeomorphic**	**area, area^2, dlink, dlink^2, slope**	**0.788**
Noturus leptacanthus			
5	**Global**	**area, slope, histw, eia1**	**0.892**
8	**Current**	**area, slope, eia1**	**0.892**
7	Current	area, elev, slope, eia1	0.887
4	Global	area, dlink^2, elev^2, slope, histw, eia1	0.885
6	Current	area, dlink^2, elev^2, slope, eia1	0.878
3	Global	area, elev, slope, histtr, eia1	0.875
2	Global	area, slope, histtr, eia1	0.874
1	Global	area, dlink^2, elev^2, slope, histtr, eia1	0.873
12	**Historic**	**area, slope, histtr**	**0.858**
14	**Hydrogeomorphic**	**area, slope**	**0.855**
11	Historic	area, elev, slope, histtr	0.851
9	Historic	area, dlink^2, elev^2, slope, histw	0.843
10	Historic	area, dlink^2, elev^2, slope, histtr	0.834
13	Hydrogeomorphic	area, dlink^2, elev^2, slope	0.814
Etheostoma etowahae			
3	**Global**	**area, dlink, dlink^2 , slope, watertr, eia1.5**	**0.946**
1	**Current**	**area, dlink, dlink^2, slope, eia1.5**	**0.945**
6	Global	area, dlink, dlink^2, slope, damswden, eia1.5	0.945
4	Global	area, area^2, dlink, dlink^2 , slope, watertr, eia1.5	0.943
7	**Historic**	**area, area^2, dlink, dlink^2, slope, damswden**	**0.936**
5	Global	area, dlink, dlink^2, watertr, eia1.5	0.932
9	Historic	area, area^2, dlink, dlink^2, watertr, slope	0.931
2	Current	area, dlink, dlink^2, eia1.5	0.927
11	**Hydrogeomorphic**	**area, area^2, dlink, dlink^2, slope**	**0.912**
8	Historic	area, dlink, dlink^2, damswden	0.903
10	Hydrogeomorphic	area, area^2, dlink, dlink^2, elev, elev^2, slope	0.896
Etheostoma scotti			
4	**Historic**	**area, elev^2, Meta. Mafic, rcc/rck, ssb, watertr**	**0.737**
5	**Global**	**area, elev^2, watertr, Meta. Mafic, rcc/rck, ssb, forestw**	**0.727**
2	**Current**	**area +elev^2, Meta. Mafic, rcc/rck, ssb, forestw**	**0.724**
7	**Hydrogeomorphic**	**area, elev^2, Meta. Mafic, rcc/rck, ssb**	**0.711**
3	Historic	area, elev, Meta_Mafic, rcc/rck, ssb, watertr	0.700
6	Hydrogeomorphic	area, dlink^2, elev^2, geology	0.675
1	Current	area, dlink^2, elev^2 +forestw	0.624
Percina palmaris			
1	**Global**	**area, area^2, dlink, dlink^2, elev, slope, watertr, eia500**	**0.921**
2	Global	area, dlink, dlink, elev, watertr, eia500	0.918
6	**Current**	**area, dlink, dlink^2, elev, eia500**	**0.908**
5	Current	area, area^2, dlink, dlink^2, elev, slope, eia500	0.905
3	**Historic**	**area, area^2, dlink, dlink^2, elev, slope, watertr**	**0.895**
4	Historic	area, dlink, dlink^2, elev, watertr	0.891
7	**Hydrogeomorphic**	**area, area^2, dlink, dlink^2, elev , slope**	**0.870**
8	Hydrogeomorphic	area, dlink, dlink^2, elev	0.853

Table 3.4. Parameter estimates for detection probability, intercepts and fixed effects of best-supported hierarchical occupancy models for each species. Detection probability estimates are given as percentages. For the intercept term, estimates correspond to site occupancy (occurrence probability) when other parameters are zero. For fixed effects, values are given as odds ratios per specified unit of increase. For example, *Cyprinella trichroistia* is 95% less likely to occur for each 1% increase in EIA within 1.5 km. For the level two random effect, values are variance estimates.

Parameter	Estimate	5% CI	95% CI	Unit of increase
Cyprinella trichroistia				
Detection probability	82%	75%	88%	
Intercept	1%	0%	15%	
area	37.83	5.46	340.22	standard deviation
$area^2$	0.48	0.23	0.86	standard deviation
dlink	1.79	0.59	5.64	standard deviation
$dlink^2$	0.22	0.04	0.69	standard deviation
slope	1.97	0.89	4.31	standard deviation
waterw	0.57	0.37	0.80	0.25%
eia1.5	0.05	0.01	0.29	1%
Level 2 random effect variance	3.28	0.43	8.08	
Noturus leptacanthus				
Detection probability	55%	44%	67%	
Intercept	24%	10%	46%	
area	9.14	2.85	44.41	standard deviation
slope	0.11	0.02	0.43	standard deviation
histw	0.44	0.08	1.80	10%
eia1	0.70	0.50	0.88	
Level 2 random effect variance	27.94	6.46	60.82	
Etheostoma etowahae				
Detection probability	55%	44%	65%	
Intercept	1%	0%	22%	
area	17.13	2.95	372.81	standard deviation
dlink	80.02	6.39	6664.24	standard deviation
$dlink^2$	0.01	0.00	0.16	standard deviation
slope	4.36	0.93	32.95	standard deviation
watertr	0.41	0.14	0.95	0.25%
eia1.5	0.19	0.01	0.85	1%
Level 2 random effect variance	15.62	4.64	34.64	
Etheostoma scotti				
Detection probability	81%	75%	86%	
Intercept	73%	55%	86%	
area	0.74	0.50	0.93	standard deviation
$elev^2$	0.84	0.66	0.98	standard deviation
watertr	0.95	0.53	1.67	0.25%
metamafic	0.48	0.19	0.88	present (binary)
rccrck	121.58	0.12	4.40×10^7	present (binary)
ssb	0.47	0.19	0.85	present (binary)
Level 2 random effect variance	7.78	3.02	16.86	
Percina palmaris				
Detection probability	86%	79%	92%	
Intercept	6%	0%	41%	
area	47.09	5.67	782.68	standard deviation
$area^2$	0.88	0.40	2.20	standard deviation
dlink	3.23	1.27	9.79	standard deviation

$dlink^2$	0.23	0.05	0.77	standard deviation
elev	1.47	0.79	3.28	standard deviation
slope	1.55	0.60	3.80	standard deviation
watertr	0.44	0.23	0.85	0.25%
eia500	0.19	0.04	0.57	1%
Level 2 random effect variance	5.26	1.16	17.39	

Figure 3.1. The Etowah Basin, showing collection sites and tributary system boundaries.

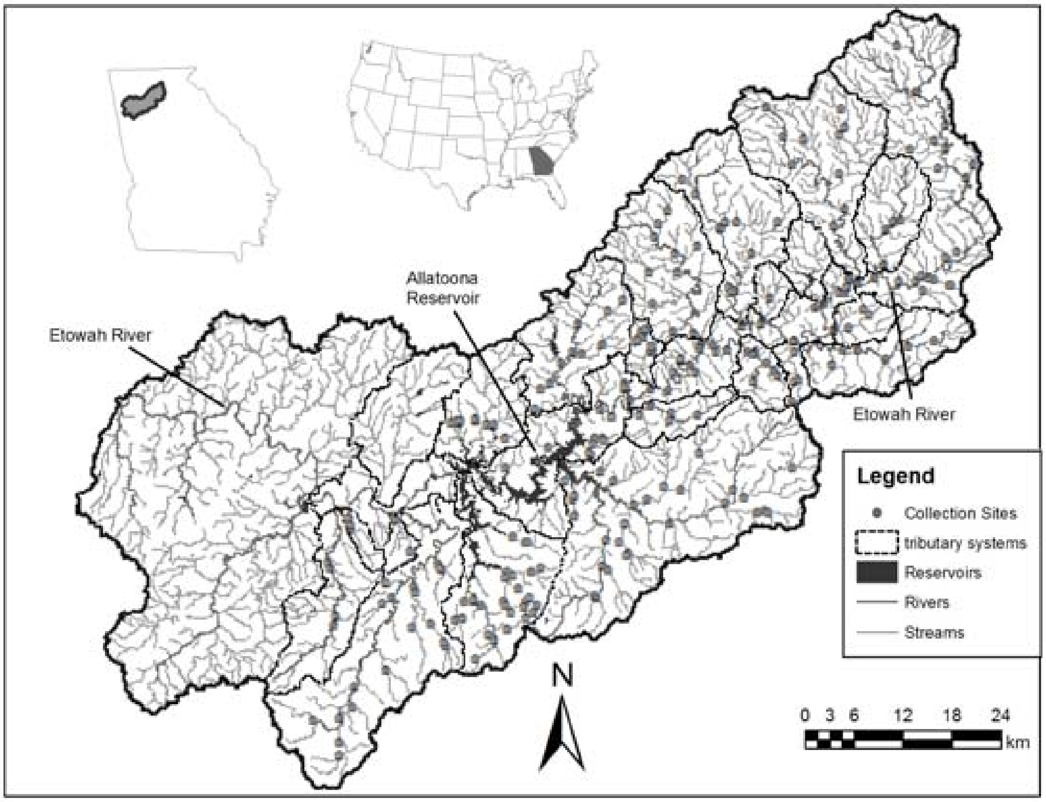


Figure 3.2. Total impervious area in the Etowah basin.

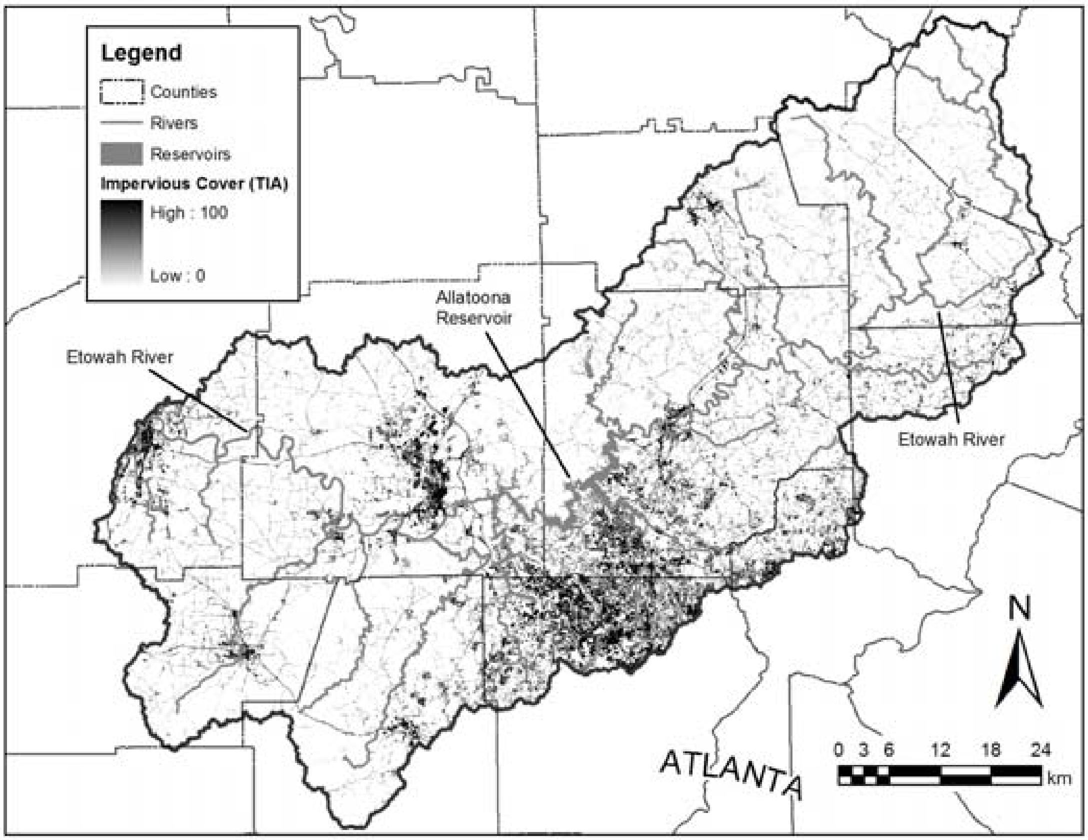


Figure 3.3. Historic modified land cover in the Etowah basin. Classification was limited to areas east of the dashed line.

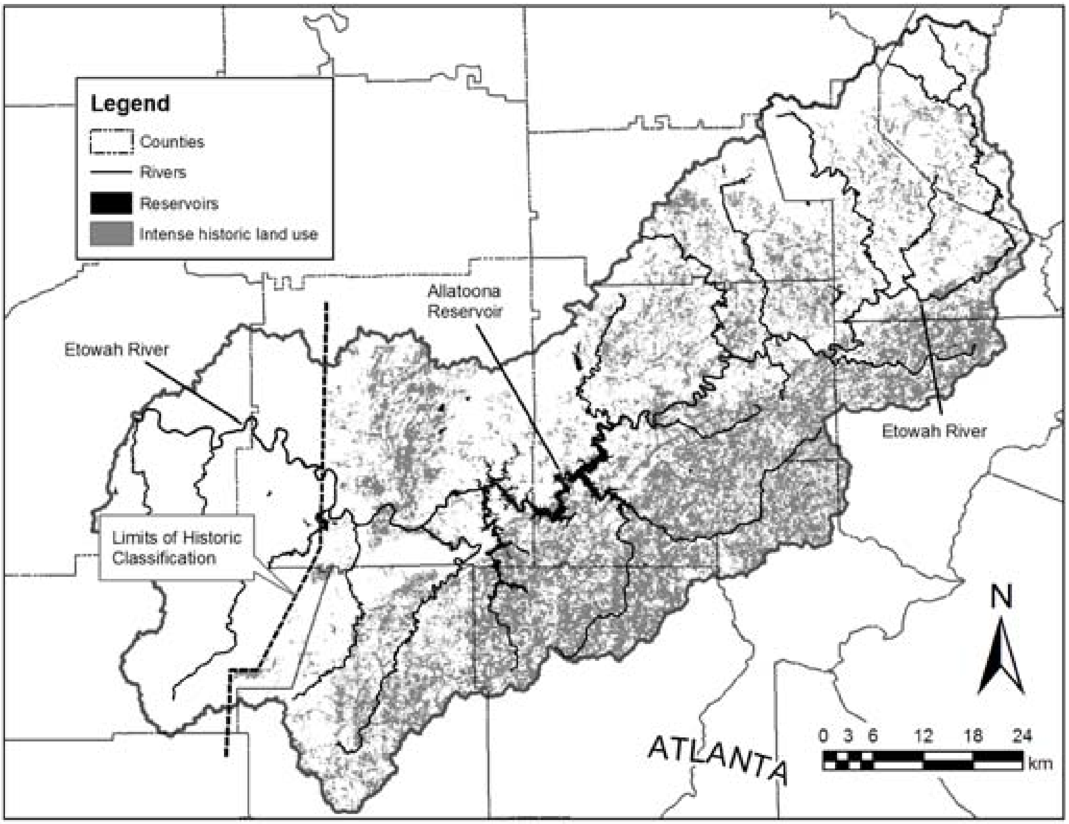


Figure 3.4. Dams in the Etowah basin.

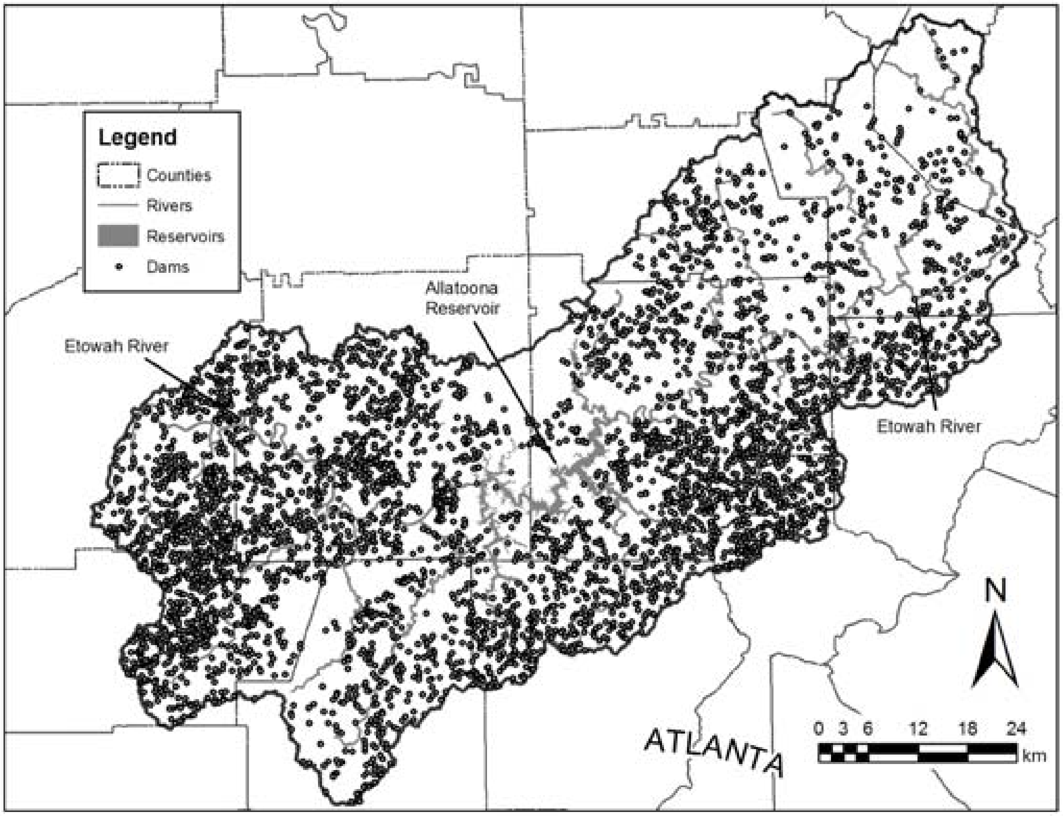


Figure 3.5. Occurrence probability of each species under the best supported model in response to increasing impervious cover or forest cover, assuming mean values for all coefficients. Black line indicates response when watershed area and dlink are one standard deviation larger than the mean; gray line indicates response with mean values for watershed area and dlink; other variables are set at mean values.

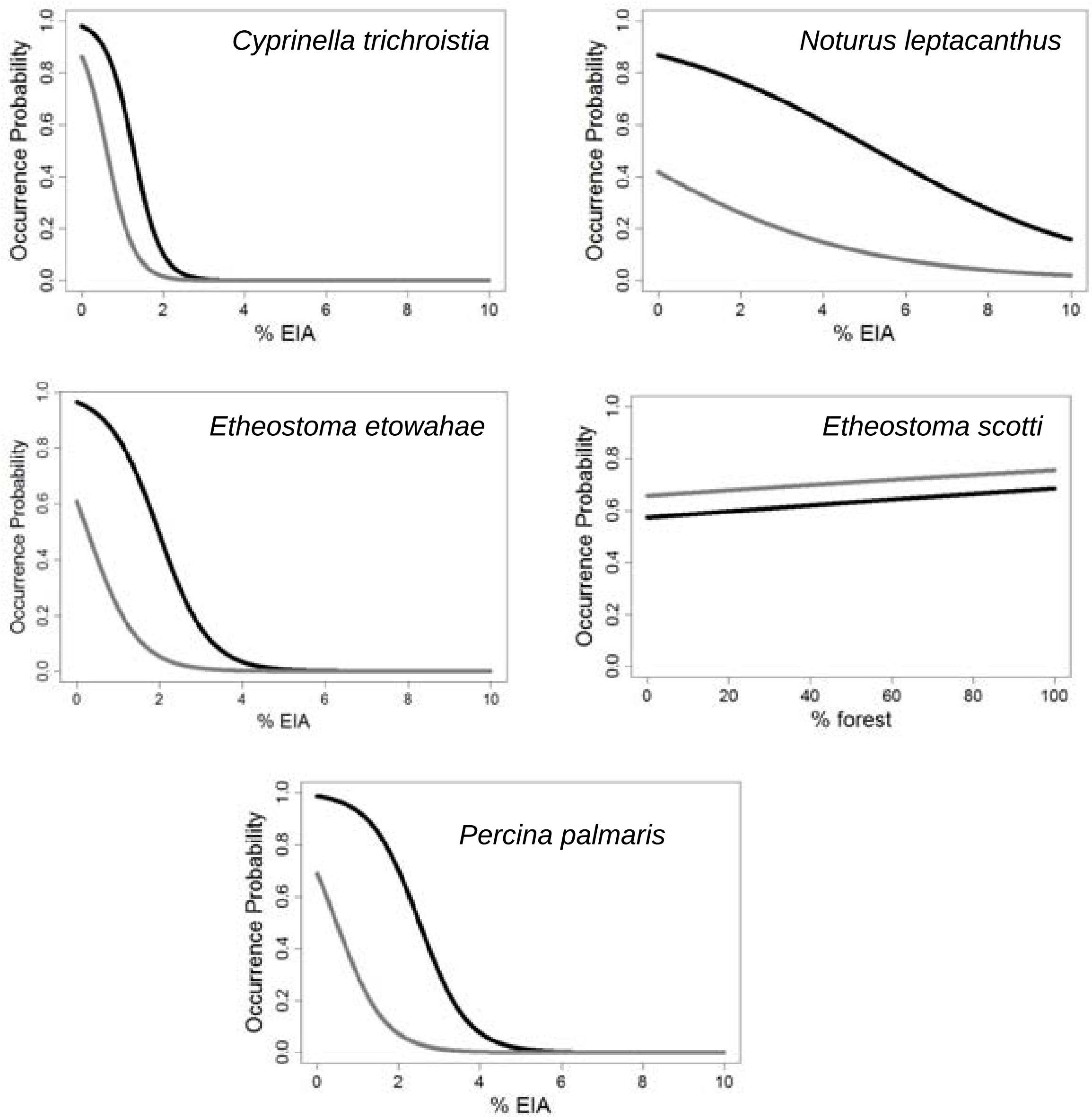
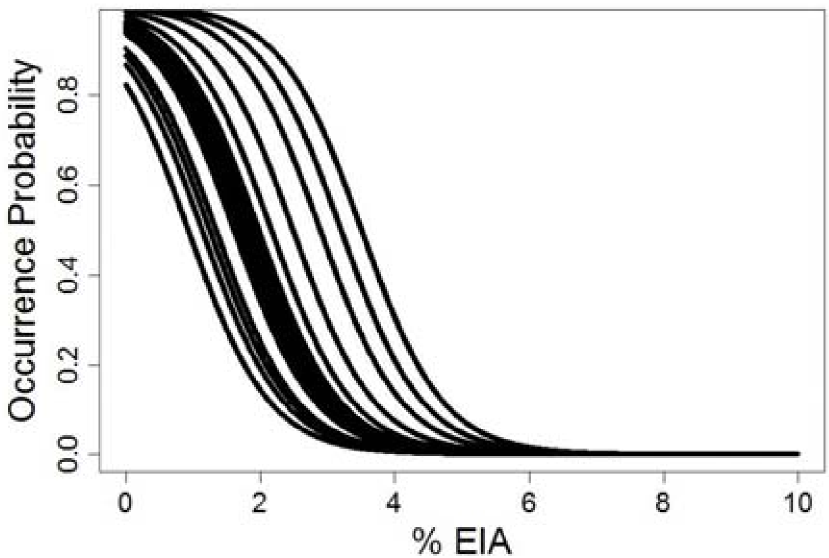


Figure 3.6. Occurrence probability for *Etheostoma etowahae* in each of 21 tributary systems as a function of increasing EIA under the best-supported model. Coefficients for fixed and random effects are held to their mean estimates. Predictor values for watershed area and dlink are set to one standard deviation larger than the mean, while other predictors are set to mean values.



CHAPTER 4

ESTIMATING SPECIES OCCURRENCE, ABUNDANCE AND DETECTION PROBABILITY FROM COUNT DATA AND PRESENCE-ABSENCE DATA[4]

[4] Wenger, S.J. and M.C. Freeman. To be submitted to *Ecology*.

Abstract

Researchers have developed methods to account for imperfect detection of species with either occupancy (presence-absence) or count data using replicated sampling. We show how these approaches can be combined to simultaneously estimate occurrence, abundance and detection probability by assuming a zero-inflated distribution. We also extend the approach to latent-abundance models of species occupancy for cases when count data are unavailable. We compare performance of these models with non-zero inflated versions using simulated data and count data for a stream fish species, *Etheostoma scotti*. We show that in many cases, a zero-inflated modeling approach yields a superior fit to the data and produces lower error rates than other models. However, there are occasions in which a simpler model is favored. We propose that zero-inflated models accounting for incomplete detectability be considered as candidates alongside other modeling approaches when replicate sample data are available.

Introduction

Species abundance and site occupancy (occurrence) of species are both useful measures of population status, and therefore of considerable interest to ecologists. Both measures may be confounded when species detection is less than perfect (Bayley and Peterson 2001, Gu and Swihart 2004, MacKenzie et al. 2003, MacKenzie et al. 2002, Royle and Nichols 2003, Royle et al. 2005). Models have been developed to address this problem in occupancy estimation, based on observed presences and absences of species from replicated samples (MacKenzie et al. 2003, MacKenzie et al. 2002, Royle and Nichols 2003), and in abundance estimation, using replicated count samples (Royle 2004, Royle et al. 2005). In the latter model, occupancy is a derived parameter based on locations where abundance is greater than zero.

However, there are cases in which the occupancy pattern and the local abundance distribution of a species arise from two distinct processes, which may be operating at different temporal or spatial scales. Consider a species that has been extirpated from a portion of its range by historic land use activities, with re-colonization limited by movement barriers. The relationship between historic land use and species occurrence may be well represented by an occupancy model. Where the species does occur, however, its abundance may be predictable by other covariates, which of course are useless for prediction in the region of extirpation. In such cases it is reasonable to consider models in which species abundance is modeled as the product of two processes: (1) species presence and (2) species abundance when present. Such an approach may also be useful when abundances exhibit a threshold effect, such that a species is either not present, or present at moderate to high abundances. In both cases, the abundance may be best represented by a bimodal, zero-inflated distribution (Welsh et al. 1996).

Zero-inflated distributions have been proposed as appropriate models for describing the spatial distribution of rare species, because of their ability to account for extra absences in the data (Cunningham and Lindenmayer 2005, Martin et al. 2005, Ridout et al. 1998, Welsh et al. 1996). A zero-inflated distribution can be viewed as a two-part model, in which (a) the probability of species presence and (b) the mean abundance, given presence, are modeled from the same data. So far, published uses of zero-inflated models has been restricted to cases where detection probability is assumed to be 1. However, their application to models accounting for incomplete detection is a natural one.

In this paper we propose an extension of the class of N-mixture models introduced by Royle (2004) and Royle et al. (2005). In those models, abundance was considered to be Poisson-distributed or negative binomially-distributed. Here, we model abundance using zero-inflated Poisson and zero-inflated negative binomial distributions. We illustrate the approach using several sets of simulated data, and compare performance of these new models to N-mixture models with simple Poisson and negative binomial distributions. We then apply the approach to a set of count data for the Cherokee darter (*Etheostoma scotti*), a federally threatened fish species. Finally, we extend the approach to cases where only presence/absence data are available, such that abundance is a latent variable to be inferred from species occurrence data. We show that the occupancy models of Royle and Nichols (2003) and MacKenzie (2002, 2003) are both special cases of a more general latent zero-inflated distribution model, and compare the performance of the models with simulated data sets.

N-Mixture Models for Abundance Estimation

Royle (2004) and Royle et al. (2005) introduced a class of models in which it is assumed that the study organism is counted at R sites, i= 1, 2, …, R, with multiple counts of organisms made at time t=1,2,…,T at each site, with such counts denoted y_{it}. These counts are viewed as realizations of a binomial process with index N_i (abundance) and detection probability p_{it}, which we denote as:

$$y_{it} \sim \text{binomial}(N_i, p_{it}).$$

Note that p is the per-individual detection probability, which is also referred to as capture efficiency. This model assumes that the population at any given site is "closed" across sampling counts, i.e., there is no change in abundance from count to count, which allows estimation of both abundance and detection probability. Detection probability can be assumed to be constant or it can be modeled as a function of covariates. For the latter, generalized linear modeling can be employed, for example with the logit link:

$$\text{logit}(p_{it}) = a_0 + a_1 x_{it}$$

where x_{it} represents a covariate on detectability as measured at site *i* on visit *t*.

Estimation of abundance is greatly facilitated if a prior distribution is assumed. An obvious choice for such a distribution is the Poisson, which arises naturally under the assumption that individuals are distributed at random:

$$N_i \sim \text{Poisson}(\lambda)$$

Of course, most species are not distributed at random; their distributions follow patterns that may be predicted based on relevant environmental conditions. This can be accommodated by adding covariates to the mean of the Poisson distribution (denoted λ), using a log link:

$$\log(\lambda_i) = b_0 + b_1 x_i$$

where x_i is the value of a covariate at site i. An alternative distribution is the negative binomial, which can be conceived as a Poisson distribution with extra dispersion. In this formulation, covariates may be placed on the mean of the negative binomial distribution, and the variance or "size" parameter provides an estimate of unexplained deviation from the mean.

Using Zero-Inflated Distributions to Model Occupancy and Abundance

Zero-inflated mixture distributions (Martin et al. 2005) or zero-modified distributions (Ridout et al. 1998), such as the zero-inflated Poisson (ZIP), are mixtures of two probability distributions, one with a point mass only at zero. Such a zero-inflated distribution provides a simple method for simultaneously modeling both occupancy and abundance. Imagine a species whose abundance at a site is described by a Poisson distribution with a λ=5, when the species is present (Figure 4.1). Now, further imagine that the species is only present at 50% of such sites (ψ=0.5). The probability density would appear as in Figure 4.2. This ZIP distribution has two parameters, a probability that the species is present (ψ) and the mean abundance of the species, if present (λ). More properly, we should say that ψ is the probability that a species is *potentially* present, because even if ψ=1, lambda may be zero; for convenience, however, we refer to ψ as the probability of presence or site occupancy.

For a ZIP distribution, the probability distribution function (Pr(Y=y)) is given by:

$$(1-\psi) + (\psi)*\exp(-\lambda) \qquad y=0$$

$$(\psi)*\exp(-\lambda)*\lambda^y / y! \qquad y>0$$

(Ridout et al. 1998). Alternatively, we can represent the mixture distribution as follows, using notation from the previous section:

$N_i = pres_i * K_i$

$pres_i \sim \text{Bernoulli}(\psi_i)$

$K_i \sim \text{Poisson}(\lambda_i)$

where *pres* is a binary value indicating whether or not the species is present, and K is the realized abundance, given presence.

We will usually wish to model ψ as a function of covariates, which we can do with a logit link:

$\text{logit}(\psi) = c_0 + c_1 x_i$

where x_i is the value of a covariate at site i. We can write the entire model as follows:

$y_{it} \sim \text{Binomial}(N_i, p_{it})$

$N_i = pres_i * K_i$

$pres_i \sim \text{Bernoulli}(\psi_i)$

$K_i \sim \text{Poisson}(\lambda_i)$

$\text{logit}(p_{it}) = a_0 + a_1 x_{it}$

$\log(\lambda_i) = b_0 + b_1 x_i$

$\text{logit}(\psi) = c_0 + c_1 x_i$

The parameters of this model can be estimated using various approaches, including maximum likelihood and Bayesian Monte Carlo Markov Chain methods. In some cases, there may be independent information on detection probability available to the researcher. In these cases, detection may be held fixed at the known value, or, in a Baysesian framework, the additional information may be used as a prior.

Naturally, it is possible to adapt this model to use zero-inflated distributions other than the Poisson, such as the zero-inflated negative binomial (ZINB). We consider both ZIP and ZINB distributions for abundance in this paper. Hereafter, we will refer to the model as a "zero inflated abundance model" to distinguish it from the models of Royle (2004) and Royle et al. (2005), which we will refer to simply as "abundance models."

Extension to Presence-Absence (Occupancy) Models

Zero-inflated abundance models can also be extended to cases where only presence-absence data are available. Instead of modeling counts as draws from a binomial distribution, we model the recorded detection or non-detection of a species as a Bernoulli probability that at least one individual is detected, given an assumed *latent* zero-inflated abundance distribution. The probability of detecting at least one individual is:

$$P(y \geq 1) = 1-((1-p)^N)$$

where p is detection probability and N is the estimated latent abundance. With this modification, we can write the entire model as follows (ignoring covariates):

$$y_{it} \sim \text{Bernoulli } (1-((1- p_{it})^{\wedge} N_i))$$

$$N_i = pres_i * K_i$$

$$pres_i \sim \text{Bernoulli}(\psi_i)$$

$$K_i \sim \text{Poisson } (\lambda_i)$$

Here, y_{it} is not a count but a recorded detection or non-detection.

We will refer to this as a zero-inflated latent abundance occupancy model. Although it appears a bit complex, in reality it is simply an extension of previously developed occupancy models, which can be derived by making simplifying assumptions. By removing the zero-inflation term, the model simplifies to the latent-abundance occupancy model of Royle and Nichols (2003). In that model, occupancy is a derived parameter of the latent abundance distribution; the species is absent when abundance is zero.

Alternatively, we can assume that abundance is constant across all sites where the species is present. Ignoring covariates, this allows the entire model to be simplified to:

$$y_{it} \sim \text{Bernoulli}(P_{it}) * pres_i$$

$$pres_i \sim \text{Bernoulli}(\psi_i)$$

Here we use P to indicate the probability of encountering at least one individual of a species. Although it may not be immediately apparent, this is a way of writing the commonly used constant-abundance occupancy model of MacKenzie et al. (2002, 3003). Therefore, both the latent abundance occupancy model and the constant-abundance occupancy model can be considered simplifications of the full zero-inflated latent abundance occupancy model. It is worth exploring when the assumptions underlying the two simplifications are supportable, and when (if ever) the more complex version may prove useful. We consider these questions using the simulation studies described in the next section.

Model Evaluation and Comparison: Methods

We conducted three studies to test the performance of the zero-inflated abundance models and zero-inflated latent abundance models: (1) a comparison of zero-inflated abundance models with abundance models using simulated sets of count data; (2) a comparison of zero-inflated abundance models and abundance models using a data set of count data for the Cherokee darter, an imperiled fish species, and (3) a comparison of zero-inflated latent abundance occupancy models with latent abundance occupancy models and constant-abundance occupancy models using simulated sets of presence-absence data.

Abundance Model Simulation Study

We created simulated sets of count data to compare the performance of three models: a Poisson abundance model ("Poisson"), a negative binomial abundance model ("Neg binomial"), and a zero-inflated Poisson model ("ZIP"). Each set was composed of 200 sites with three collections at each site. An abundance value was randomly generated for each site. In Data Set 1, abundances were drawn from Poisson distributions with an overall mean of 3, and no extra zeros. We hypothesized that there would be no benefit to using a zero-inflated abundance model or a negative binomial distribution to model this dataset, so that the Poisson model would be the most parsimonious. In Data Set 2, abundances were drawn from Poisson distributions (with overall mean of 3, when present) with extra zeros, with both the abundances and extra zeros generated by the same underlying random gradient. In Data Set 3, abundances were drawn from Poisson distributions (again with overall mean of 3, when present) with extra zeros, but the abundances and extra zeros were generated by different underlying random gradients. We hypothesized that a zero-inflated abundance model would prove superior to a Poisson abundance model with Data Sets 2 and 3, but that a negative binomial abundance model might have an intermediate fit. Once the abundance values were generated for each site, random binomial draws were made to represent the count of individuals observed during each "collection," using a detection probability of 30%. In addition, for Data Sets 1 and 2 we generated one covariate representing the underlying gradient of abundance and extra zeros, while for Data Set 3 we generated two covariates representing the two underlying gradients of abundance and extra zeros.

We generated 100 replicates of each data set, where each replicate was a realization of a set of random draws as described above. We fit each of the three models to each replicate of each data set. For each model, we included relevant covariates placed on the appropriate terms to maximize predictive ability (Table 4.1). No covariates were placed on detection (P). Models were run by maximizing the likelihood using routine "nlm" in the statistical software R (R Development Core Team 2005). To compare models, we calculated Akaike's information criterion adjusted for small sample size (AICc) (Burnham and Anderson 2002) and converted this to a delta AICc by subtracting the lowest value for each replicate from each model. We then averaged the delta AICc across all replicates for each data set. We also calculated the mean per-site error in estimating abundance, the average error in predicting occupancy and the mean detection probability for each model, averaged across all replicates for each data set. Finally, we explored model performance when independent estimates of detection probability were available by running versions of the Poisson and ZIP models with detection fixed at the true value of 30% for a further 100 replicates of Data Set 2. We calculated the same summary statistics for these models as for the others.

Cherokee Darter Survey Data Study

A real-world illustration of a zero-inflated abundance model is provided using data for the Cherokee darter (*Etheostoma scotti),* a federally protected stream fish endemic to the Etowah River Basin, Georgia, USA. Data were collected at 215 sites in small and medium-sized streams between 1999 and 2003 using backpack electrofishing and kick seining. During each collection, captured Cherokee darters were counted and returned to the stream. A subset of sites was sampled more than once; those collections made at the same site within two consecutive years were included in the analysis under the assumption that the population was closed during that period (consequences of violation of this assumption are discussed below). This subset comprised 54 sites sampled a second time and seven sites that were sampled a third time. Additional non-consecutive collections at those sites were not used in this analysis. A total of 276 collection records were included. These data were originally collected for various purposes, and potentially incomplete or unreliable records were discarded in paring the data set down to the 276 collection records used here. Of these, the methods and effort at each site appeared to be comparable, although we hypothesized that there was some variation among collectors associated with the three major institutions performing the collections. Collector institution was therefore considered a covariate on detection. Cherokee darter counts ranged from 0 to 145, with a mean of 10 and standard deviation of 19.

Six potential site-level covariates of abundance and occurrence, including a variable of management concern (effective impervious area, EIA) were recorded from mapped data using a Geographic Information System (GIS). Reach-scale and date-specific data (such as habitat type and water temperature) were not recorded with sufficient consistency across the data set to permit inclusion in models. The site-level variables were identified through exploratory analyses of 24 potential variables in relation to darter abundance and occurrence, using linear and logistic regression, respectively. The five variables exhibiting strongest association with fish abundance or occurrence, in addition to EIA, were: elevation, bedrock (BR) geology, surficial (Quaternary) geology, sub-basin reservoir area (area inundated by reservoirs in the tributary system to which the site belongs), and occurrence in the Little River tributary system (where the Cherokee darter appears to be widely extirpated; Burkhead et al.1997). EIA was measured as the proportion of the area within 1 km of the collection site that was impervious and drained by storm sewers. These six variables were not strongly intercorrelated (i.e., $r < 0.4$), although impoundment area tended to be high within the Little River system.

Five sets of model covariates were formulated to represent alternative hypothesized effects of site variables on fish abundance, occurrence, or both (Table 4.2). Based on the results of the exploratory analysis, our strategy was to include two variables (Quaternary geology and EIA) in all models as covariates on abundance and two variables (elevation and BR geology) in all models as covariates on occurrence, and to test alternative combinations of the remaining variables as covariates on occurrence and/or abundance. Each of five sets of covariates (Table 4.2, Covariate Sets A-E) was modeled using the zero-inflated Poisson and zero-inflated negative binomial abundance models. Variations of these covariate sets were also fit using Poisson and negative binomial models without zero-inflation terms, in which cases all covariates were placed on the abundance term (Table 4.2, Covariate Sets F-H). A total of 16 models were evaluated.

From the fitted models we used the mean of the abundance distribution and the detection probability for each site to calculate an expected count. We compared this to the actual observed count to calculate a within-sample abundance estimation error. We calculated a predicted probability of a species not being observed at each site based on the probability of occupancy being zero, abundance being zero, and a species being present but unobserved. We compared this to actual detection-nondetection to calculate a within-sample occupancy estimation error. Note that these error rates are fundamentally different from the error rates of the same names used in the simulation studies, where truth is known.

Occupancy Model Simulation Study

We created three simulated datasets using the same methods as for the Abundance Model Simulation Study, except that we generated a simulated history of observed detections and non-detections rather than observed counts at each of 200 sites. We created 100 replicates of each of the data sets and fit a latent-abundance Poisson model ("Poisson"), a zero-inflated latent-abundance Poisson model ("ZIP"), and two versions of a constant-abundance occupancy model ("Constant") to each replicate. In the first Constant model ("Constant A") covariates were placed only on the occupancy parameter, Psi (Table 4.3).

In the second Constant model (“Constant B”) covariates were placed on detection probability, P. The second approach simulates the decline in detection due to change in abundance, even though abundance itself is not modeled. We hypothesized that the Poisson model would prove superior to the other models for Data Set 1, but that the ZIP and Constant B models would be best supported with Data Sets 2 and 3.

Models were run by maximizing the likelihood using routine “nlm” in the statistical software R (R Development Core Team 2005). To compare models, we calculated Akaike’s information criterion adjusted for small sample size (AICc) (Burnham and Anderson 2002), and converted this to a delta AICc by subtracting the lowest value for each replicate from each model. We then averaged the delta AICc across all replicates for each data set. We also calculated the mean per-site error in estimating abundance, the mean error in predicting occupancy and the mean detection probability estimated by each model.

Model Evaluation and Comparison: Results

Abundance Model Simulation Study

The ZIP model was well-supported relative to alternative models for each of the simulated data sets (Table 4.4). For Data Set 1, in which abundances followed a Poisson distribution without extra zeros, all three models had similar support based on AICc. In half the replicates the ZIP model had the lowest AICc score and in slightly less than half of the replicates the Poisson model had the lowest AICc score, while the negative binomial model had the lowest AICc score in only 3 replicates. In all cases the differences in AICc scores were small. The ZIP model generally had the lowest occurrence errors for most replicates, but the Poisson model had the lowest abundance errors. All models estimated detection as close to the true value of 30%.

For Data Set 2, the ZIP model had the lowest (best) AICc score in all cases, an average of 19 points lower than the negative binomial model and an average of 38 points less than the Poisson model. The ZIP model had the lowest occurrence error but considerably higher abundance error than the other two models. Data set 3 yielded similar results, with the lowest occurrence error associated with the ZIP model but the lowest abundance error associated with the Poisson model. The ZIP model had the most accurate estimates of detection, while the Poisson model tended to overestimate detection considerably.

Examination of the results revealed that in 18 of the replicates in Set 2 and six of the replicates in Set 3, the ZIP model significantly underestimated detection, leading to inflated estimates of abundance. These models were nevertheless the best supported of the alternatives given the data, probably because they had exceptionally low occupancy error rates. In the other 88% of replicates, however, the ZIP models had a slightly lower average abundance error than the Poisson and negative binomial models.

When detection was held fixed at 30%, the ZIP model had a lower AICc score than the Poisson model (mean difference of 46 points). The ZIP model also had lower occurrence error (31% compared to 38%) and slightly lower abundance error (1.35 rather than 1.39) than the Poisson.

Cherokee Darter Data Study

Based on AICc, the best-supported model was the zero-inflated negative binomial abundance model with covariate set B (Table 4.5). There was a clear separation of models based on the assumed abundance distributions. The negative binomial models performed better than the Poisson models, while the zero-inflated models performed better than models with the same distribution but no zero-inflation term. The negative binomial and Poisson models also differed in which covariate sets were best-supported, with the most complex covariate sets (E and H) best-supported in Poisson models (Table 4.5).

The abundance error rates are similar across all models and covariate sets (Table 4.5) and do not capture the large differences between the negative binomial and the Poisson models, as indicated by the AICc scores. We believe this is because the error rates are based on a point estimate from the mean of the abundance distribution. However, the Poisson distribution has a fixed variance, while the negative binomial distribution has an independent variance term (the size term), allowing the distribution to widen to encompass more of the data points. All of the negative binomial models fitted to the Cherokee darter data have a wide variance, indicating that there is a considerable amount of unexplained variation in the data. Trying to fit the Poisson models' relatively narrow fixed variance omits many points, producing a lower likelihood.

Occupancy Simulation Study

Latent-abundance and occupancy models varied in performance when applied to simulated detection data (Table 4.6). For Data Set 1, the Poisson, ZIP and Constant B models (constant abundance model with covariates on both Psi and P) had similar AICc scores, but the Constant A model (with covariates only on Psi) was less well supported. Error in predicting occurrence was similar across models. Error in predicting abundance was significantly lower for the Poisson model than for the ZIP model (abundances were not predicted by the constant abundance models). Estimates of detection were most accurate for the Poisson model. Note that the Constant models do not provide per-individual detection estimates, but rather estimates of detection probability for the species at the site (i.e., the chance of encountering at least one individual), and are not directly comparable to the per-individual detection estimates of the Poisson and ZIP models. However, values were very close to the "true" value of 0.66, calculated based on the mean N of 3.

For Data Set 2, each of the four models was selected as best in at least 20 of the 100 replicates, indicating similar levels of support, although the Constant A model tended to have higher AICc scores on the average. Occurrence error was similar among all models, while abundance error was lower in the Poisson model than the ZIP model. The Poisson model tended to overestimate detection, while the ZIP model was closer to the true value of 30%. For Data Set 3, the ZIP model and the Constant B model were considerably better supported than the Poisson and Constant A models. Again, however, all models had similar levels of error in estimating occurrence, and the Poisson model was lower than the ZIP model in abundance error. The Poisson model again overestimated detection.

Detailed examination of the results showed that the high average abundance errors of the ZIP models were driven by a minority of replicates in which detection was considerably underestimated, leading to large overestimates of abundance (just as was the case in the abundance model simulation study described previously). In the case of the occupancy models, however, when the ZIP model performed poorly it always had a worse AICc score than at least one of the competing models. Thus, in real-world applications where a ZIP model is evaluated as one of several competing models, it would not be selected in cases where it performs poorly. For the replicates where the ZIP model was the best supported, its abundance errors were close to the values for the Poisson model, and often considerably lower.

Discussion

Our results for simulated data and an application to actual abundance data for a fish species show that zero-inflated models can be useful in estimating species occurrence probabilities and abundances from replicate count data, while accounting for incomplete detection. In particular, models with a zero-inflation term are better supported when species absences occur independently of, or in addition to, the processes driving abundance, and can provide more accurate estimates of species occupancy rates than models lacking the zero-inflation term. As predicted, however, with a Poisson-distributed set of counts (Abundance Data Set 1) there is little advantage to adding the extra complexity of a zero-inflation term, except for a small improvement in predicting presences and absences. In addition, the ZIP models occasionally (12% of replicates) underestimate detection and overestimate abundances while providing apparent good fit to the data, a problem that can be avoided if independent estimates of detection probability are available. Latent-abundance models with a zero-inflation term also provide good fits to detection/non-detection ('presence/absence') data, and allow one to separate effects of changing abundance and detectability underlying apparent changes in species occupancy rates.

Zero-inflated models have been recommended as appropriate for rare species (Cunningham and Lindenmayer 2005, Welsh et al. 1996), although it has been demonstrated that a negative binomial model can provide a superior fit to a ZIP model in some apparently zero-inflated data sets (Warton 2005). We believe that rarity alone is not sufficient grounds for selecting a zero-inflated model, and we agree with Warton (2005) that negative binomial and other distributions without zero-inflation terms should also be considered as alternatives. Even for rare species, it is often possible to add appropriate class covariates to the mean abundance term of a model to account for absences, without the need for a zero-inflation term. For example, if a rare plant is only found in association with certain soil types, adding covariates for those soil types to an abundance model (without zero inflation) may adequately explain species presence (although additional covariates will be necessary to explain its abundance). In such a case a zero-inflated structure may add nothing but unnecessary complexity. In thinking about this problem, we find it helpful to bear in mind that the choice of a modeling distribution is a specification of the *error* distribution, not the abundance distribution of the data themselves. Examination of a histogram of abundance data will not necessarily give insight into the appropriate distribution to use, although examination of residuals after fitting covariates *can* be informative.

When is a zero-inflated distribution likely to be better supported than a non-inflated distribution? There are at least two circumstances. First, if a continuous covariate is a useful predictor of occurrence but not of abundance, then applying that covariate to the occurrence term of a zero-inflated abundance model will likely yield a better fit than applying that covariate to abundance in a model without a zero-inflation term. However, if the covariate can be converted to a class variable, it may be a useful covariate for the latter model type. For example, if the occurrence (but not abundance) of a plant is related to elevation, then adding elevation as a predictor of abundance will not be very fruitful, although adding a dummy variable for sites where elevation is less than a threshold value *may* be useful. Second, if abundance shows a threshold relationship to a continuous covariate, a better fit should be obtained by adding that covariate to both the occurrence and abundance terms, rather than just abundance. In both of these cases a zero-inflated model should prove better supported than a negative binomial, Poisson, or other simple abundance model. In addition, the separate occupancy and abundance terms of the zero-inflated model can have heuristic value in representing the different mechanisms that gave rise to the present observed pattern of species abundance.

Our formulation of the zero-inflated model was as a simple mixture using a Poisson or negative binomial distribution. Other authors (Cunningham and Lindenmayer 2005, Warton 2005, Welsh et al. 1996) have suggested using a *truncated* Poisson or negative binomial distribution rather than the full distribution. This produces a two-part, conditional model in which occurrence and abundance are truly separate and orthogonal, and eliminates the possibility that a species may be predicted as present (occupancy=1) but with an abundance of zero. Cunningham and Lindenmayer (2005) note that this approach also has computational advantages. On the other hand, if mean abundance is moderate or high, the two approaches are functionally equivalent (Welsh et al. 1996). We found the simple mixture easy to construct in the software programs we used (WinBUGS and R), and note that conceptually it is not illogical to envision a species as potentially present based on the covariates that govern its distribution (occurrence), but absent at a patch because the covariates that govern its abundance are unfavorable. Therefore, we see both formulations as valid alternatives.

The Cherokee darter example illustrates the real-world value of considering models with a zero-inflation term. Our interest here was in developing the best predictive model for populations of the Cherokee darter in response to future changes in EIA. The best zero-inflated model was much better supported than the abundance models without a zero-inflation term. Both models produced similar estimates of abundance, and in both cases the size term (variance parameter) for the negative binomial distribution was large, indicating considerable unexplained variation in abundance. However, the zero-inflated negative binomial model had lower rates of within-sample error in predicting presence and absence of Cherokee darters: 27% rather than 38%. For our purposes, it was useful to identify a model with superior ability to predict occupancy without a sacrifice in the ability to predict abundance.

The cases in which zero-inflated abundance models underestimate detection and overestimate abundance are a potential cause for concern in the application of these models. If independent detection data are available, they can provide a useful check as to whether the zero-inflated abundance model's predictions are reasonable. In our analysis of the Cherokee darter data, we were reassured to find that the estimated detection was similar to an independent estimate derived from a mark-recapture study (unpublished data). Such independent data can be incorporated as a prior if the model is formulated under a Bayesian approach and implemented in software such as WinBUGS (Spiegelhalter et al. 2003). Alternatively, if the independent estimates of detection are considered highly reliable, then detection can be fixed at this value in the zero-inflated abundance model. Our simulations showed that under these circumstances the zero-inflated model provides consistently more accurate estimates of occupancy and similar or better estimates of abundance than a model without a zero-inflation term, for the data set evaluated.

Occupancy Models

Constant abundance models of occupancy and detection are now widely employed (MacKenzie et al. 2006). Our results indicate that these simple models can provide a good fit to the data, even when detection varies according to abundance, by placing covariates on both the occupancy and detectability terms. Royle and Nichols (2003) found that under some circumstances, latent abundance models can provide a superior fit to the data, and we also found that latent abundance and zero-inflated latent abundance models can provide equally good or better fits to some data sets. Additionally, latent abundance and zero-inflated latent abundance models have the advantage of separating covariates for abundance from those on detectability. That is, one can separately account for a decline in detection due to low abundance from a decline in detection due to gear efficiency, collector skill or environmental conditions. This can be advantageous for increasing mechanistic understanding of the system under study. It can also reduce misinterpretations of the cause of an observed decline in occurrences.

Abundance vs. occupancy models

When working with existing data sets, there is often no choice but to use an occupancy model. In many cases only presence/absence data are available. In other cases abundance data exist, but there may be uncertainty as to whether methods were comparable across all sites (especially when multiple data sets are combined in a meta-analysis) or whether records are accurate. Although it is possible to apply covariates to detection to reflect differences in collectors and conditions, if the variability is too high or adequate covariates are not available, it may make sense to reduce the data to ones and zeros. However, in the design of new studies, there is considerable advantage to recording counts rather than presence/absence. Even when costs must be controlled and surveys are necessarily perfunctory, there will be some sites where several individuals are observed rather than one, and it is usually only a minor issue to record the number rather than a one (in some sites a great many individuals may be observed, making counts difficult; but we argue that it is still far more informative to estimate the number than to record it as a one). Consequently, with count data we can produce more precise parameter estimates and considerably reduce our uncertainty.

As an example, we compared the parameter estimate error for the ZIP abundance models to the parameter estimate error for the ZIP occupancy models (Figure 4.3) fitted to a randomly generated replicate of Data Set 3. The same data were used for both models, although the response variables were expressed as counts for the ZIP abundance model and presence/absence for the ZIP occupancy model. In all cases the error for the parameters of the occupancy model was higher, although the difference was most notable for the intercept terms.

Practical considerations

The zero-inflated abundance and latent abundance models introduced here retain the same assumptions as their respective non-zero-inflated cousins (MacKenzie et al. 2002, Royle 2004). Important among these is the assumption of closure: that is, the population of a site is closed to immigration and emigration across repeat collections made at the site. For mobile species this assumption will frequently be violated to some degree, which will lead to underestimates of detection probability, as variability in occupancy or abundance estimates will be attributed to non-detection rather than changes in population (MacKenzie et al. 2006). Count data can also be misleading when only a portion of a population is available to the collector. For example, some fish populations may occur in both easily-sampled shallow-water habitat and hard-to-sample deepwater habitat. Clearly, population estimates based on counts can only be applied to the habitat patches that were actually sampled, which means that the true population cannot be known unless other habitats are sampled as well.

We have used both Bayesian methods implemented in WinBUGS (Spiegelhalter et al. 2003) and maximum likelihood methods using package nlm in R (R Development Core Team 2005) to fit these models. The code in WinBUGS is fairly straightforward. Coding for maximum likelihood estimation is considerably more complex, although we are exploring development of a package to allow more streamlined model formulation. Our code for nlm is based on Royle et al. (2005). In exploratory analyses we encountered occasional convergence problems with both approaches, and found that these problems tended to be worse for small data sets (less than 100 total samples).

Conclusions

Zero-inflated abundance and zero-inflated latent abundance models accounting for incomplete detectability provided better fits to some data sets than alternative models. These models are worthy of consideration under many circumstances where repeat count data or repeat presence-absence data are available. For such data sets, we recommend formulating a candidate set of models that includes both zero-inflated and non-zero-inflated model formulations with covariates reflecting a priori hypotheses to the extent possible, following a general information theoretic framework (Burnham and Anderson 2002). The best-supported models can be selected based on AICc or cross validation methods. As with any modeling approach, care must be taken to verify that the output is reasonable and consistent with independent data. Whether or not this approach ultimately provides the best fit to a given data set, there is value in considering the different mechanisms that give rise to observed species abundance distributions. Attempting to fit the separate occupancy and abundance terms of a zero-inflated model can provide a useful conceptual framework for species distribution analysis, potentially leading to better understanding of species and their relationships with their environment.

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Table 4.1. Covariates applied to abundance (lambda or mu) and occupancy (psi) for each model and data set in the abundance simulation study**.**

Data Set	Poisson Model	Neg. binomial Model	ZIP Model
Set 1: Poisson distributed abundances	Lambda: C1	Mu: C1	Lambda: C1 Psi: C1
Set 2: Poisson distributed abundances with extra zeros	Lambda: C1	Mu: C1	Lambda: C1 Psi: C1
Set 3: Poisson distributed abundances with extra zeros generated by a different random gradient	Lambda: C1, C2	Mu: C1, C2	Lambda: C1 Psi: C2

Table 4.2. Covariates on abundance and occurrence for Cherokee darter models.

Covariate Set	Covariates: Abundance	Covariates: Occurrence
A	Quat. geology, EIA	Elevation, BR geology, impoundments
B	Quat. geology, EIA	Elevation, BR geology, LR system
C	Quat. geology, EIA	Elevation, BR geology, impoundments, LR system
D	Quat. geology, EIA, elevation, BR geology	Elevation, BR geology, impoundments
E	Quat. geology, EIA, elevation, BR geology	Elevation, BR geology, impoundments, LR system
F	Quat. geology, EIA, elevation, BR geology, impoundments	-
G	Quat. geology, EIA, elevation, BR geology, LR system	-
H	Quat. geology, EIA, elevation, BR geology, impoundments, LR system	-

Table 4.3. Covariates on abundance (lambda), occupancy (Psi) and detection (P) for each model and data set in the occupancy simulation study. Data sets are as defined in Table 4.1.

	Model			
Data Set	**Poisson**	**ZIP**	**Constant A**	**Constant B**
Data Set 1	Lambda: C1	Lambda: C1 Psi: C1	Psi: C1	Psi: C1 P: C1
Data Set 2	Lambda: C1	Lambda: C1 Psi: C1	Psi: C1	Psi: C1 P: C1
Data Set 3	Lambda: C1, C2	Lambda: C1 Psi: C2	Psi: C1, C2	Psi: C2 P: C1

Table 4.4. Results of the abundance model simulation study. Values for delta AIC_c, occupancy error, abundance error and mean detection probability are averages across 100 replicate simulations.

Data Set	Model	delta AICc (std err)	Occupancy Error (std err)	Abundance Error (std err)	Mean Detection
Data Set 1	Poisson	2.1 (2.9)	10.5% (3.8%)	1.36 (0.15)	30%
	Neg binomial	3.6 (2.9)	11.3% (8.5%)	1.46 (0.74)	29%
	ZIP	1.1 (1.5)	9.5% (3.6%)	1.4 (0.20)	29%
Data Set 2	Poisson	37.8 (17.0)	33% (3.7%)	1.4 (0.19)	43%
	Neg binomial	19.4 (10.8)	31.9% (3.3%)	1.43 (0.19)	35%
	ZIP	0 (0)	30.7% (2.9%)	1.7 (1.26)	28%
Data Set 3	Poisson	44.2 (15.5)	36.1% (4%)	1.34 (0.19)	44%
	Neg binomial	22.4 (8.6)	35% (3.6%)	1.37 (0.19)	35%
	ZIP	0 (0)	32.5% (3.3%)	1.36 (0.28)	30%

Table 4.5. Results of the Cherokee darter study. Values for delta AIC_c are shown for alternative covariate combinations applied to each model. Occupancy and abundance error rates are within-sample estimates, based on model predictions and observed values (see text).

Model	Covariate Set	Number of Terms	Delta AICc	Occupancy Error	Abundance Error
ZINB	B	12	0	27.4%	8.57
ZINB	D	13	2	27.9%	8.65
ZINB	C	13	2	27.0%	8.52
ZINB	E	15	5	27.0%	8.40
ZINB	A	12	15	31.2%	8.96
NB	G	11	59	38.6%	8.55
NB	H	12	61	38.6%	8.50
NB	F	11	75	46.5%	8.97
ZIP	E	14	1766	27.0%	8.78
ZIP	D	12	1812	27.9%	9.29
ZIP	B	11	1824	27.9%	9.13
ZIP	C	12	1825	27.0%	8.90
ZIP	A	11	1841	28.4%	8.60
Poisson	H	10	3750	45.6%	8.36
Poisson	G	11	3766	45.6%	8.27
Poisson	F	10	4115	46.5%	8.68

Table 4.6. Results of the occupancy simulation study. Values are the means and standard errors for 100 replicates of each data set fitted to each model.

Data Set	Model	delta AICc (std err)	Occurrence Error (std err)	Abundance Error (std err)	Mean Detection
Data Set 1	Poisson	2.6 (3.5)	11% (3.8%)	1.39 (0.22)	31.2%
	ZIP	1.3 (1.3)	10.2% (3.6%)	6.12 (10.08)	21.6%
	Constant A	12.7 (6.9)	10.9% (4%)	--	65.3%*
	Constant B	3.4 (3.5)	10.5% (3.7%)	--	63.8%*
Data Set 2	Poisson	2.5 (2.9)	30.9% (3.2%)	1.41 (0.22)	54.4%
	ZIP	1.6 (1.4)	30.3% (3.4%)	2.7 (4.15)	35.7%
	Constant A	5.0 (4.9)	30.7% (3.2%)	--	67.7%*
	Constant B	2.1 (2.5)	30.7% (3.2%)	--	63.5%*
Data Set 3	Poisson	6.4 (5.4)	33.4% (3.6%)	1.27 (0.17)	53.7%
	ZIP	1.0 (1.5)	32.2% (4.0%)	2.03 (2.57)	32.2%
	Constant A	8.0 (5.9)	32.8% (3.6%)	--	64.8%*
	Constant B	3.2 (3.8)	33.5% (3.6%)	--	61.3%*

** Estimate is for detection of at least one individual per site, rather than per-individual detection.*

Fig. 1. Poisson Abundance Distribution, lambda = 5

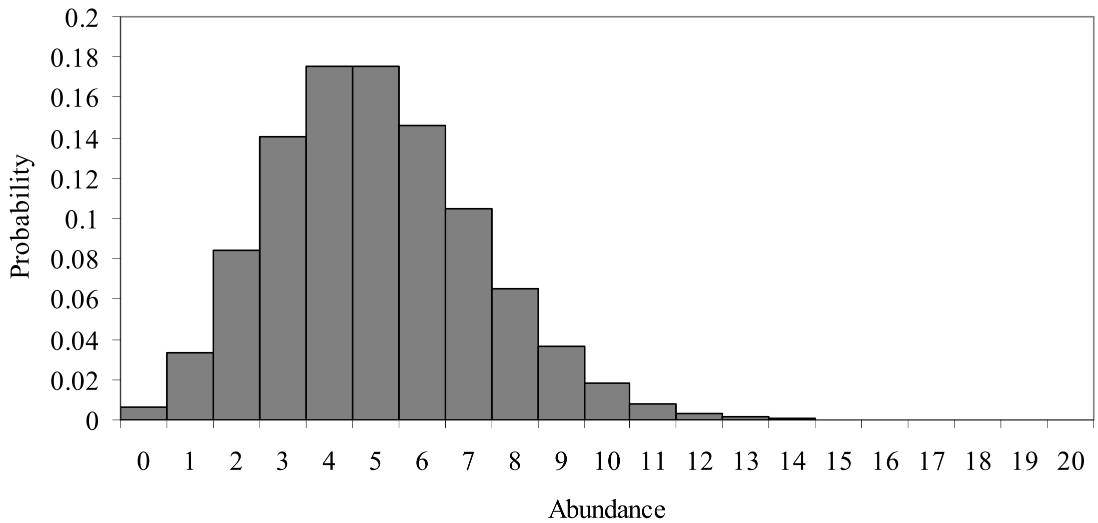


Fig.2. Zero Inflated Poisson Abundance Distribution, psi=0.5, lambda = 5

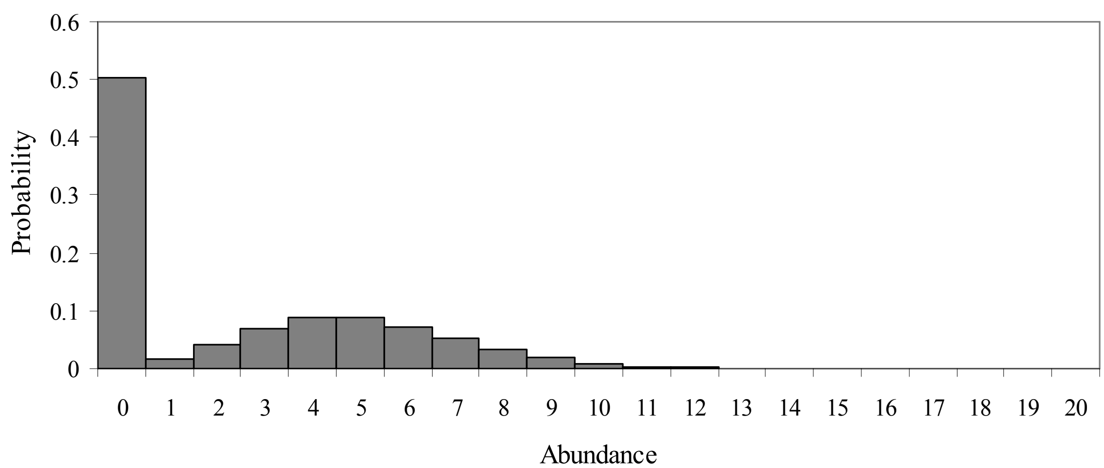
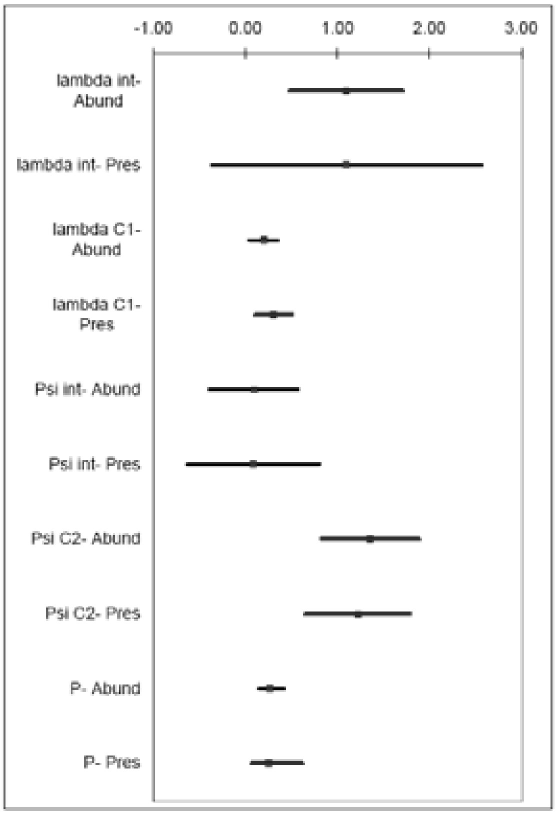


Figure 4.3. Means and 95% confidence interval parameter estimates for a ZIP abundance model and a ZIP occupancy model fitted to a replicate of Data Set 3. In the figure, "int" means intercept, "C1" and "C2" are covariates, "Abund" means ZIP abundance model, and "Pres" means ZIP occupancy (presence-absence) model.



CHAPTER 5

A STORMWATER MANAGEMENT PERFORMANCE STANDARD TO LIMIT RUNOFF VOLUMES[5]

[5] Wenger, S.J., T.L. Carter and L.A. Fowler. To be submitted to the *Journal of the American Water Resources Association.*

Abstract

Stormwater runoff from impervious surfaces is a major threat to stream fish species. Conventional policies for managing stormwater runoff or minimizing impervious cover are likely to be insufficient to protect very sensitive species of fish: impervious cover limits adequate to protect sensitive species are impractical in most cases, while commonly implemented stormwater regulations have limited ability to reduce impacts because they do not address underlying causes. We propose a stormwater management performance standard that limits runoff *volumes*, rather than discharge flow rate or contaminant concentrations. Such a standard requires the use and maintenance of infiltration management practices, which can maintain natural hydrologic flow patterns and have high rates of contaminant removal. It also provides an incentive for maintaining natural land cover and minimizing impervious cover. We describe a version of this standard that is being implemented in the Etowah River basin, Georgia, USA, to protect imperiled fish species threatened by increasing urbanization. We discuss how the standard can be adapted to more general use, and identify barriers to implementation and how they may be overcome.

Introduction

Stormwater runoff from impervious surfaces constitutes one of the greatest threats to stream ecosystem health in urban and suburban watersheds (Walsh et al. 2005b). Conversion of pervious land cover to impervious surfaces produces profound changes to stream hydrology (Poff et al. 2006, Roy et al. 2005, Walsh et al. 2004a), altering base flows and causing ongoing channel erosion that may constitute the major source of sediment to some streams (Trimble 1997). Runoff is also a major vector for stream contaminants (Walsh 2000), including metals

(Bannerman et al. 1993), nutrients (Barth 1995), pesticides (Robinson 2003), and other toxicants. Recognition of these problems has driven the development of stormwater management and land use regulations in many communities. However, while these rules are undoubtedly contributing to more effective stream protection, existing efforts may be insufficient to protect sensitive aquatic organisms. Many studies and reviews have indicated that aquatic communities are altered when the upstream watershed reaches 10% total impervious cover (Booth and Jackson 1997, Klein 1979, Schueler 1994, Wang et al. 2000). Other studies suggest a lower threshold (e.g., Ourso and Frenzel 2003). Considering that urban land cover and associated impervious surfaces are steadily increasing, society is faced with an important question: is it possible to manage stormwater and/or land use in a way that prevents the loss of aquatic biodiversity?

Fundamentally, there are two ways to reduce the impact of stormwater runoff from impervious surfaces. The first is to prevent the creation of increased runoff volumes and rates by limiting the amount of impervious surface itself. From a policy perspective, this entails a limitation on land use—perhaps constraining development density through zoning or enacting a regulatory impervious surface limit. The second method is to manage the runoff, after it has been generated, in a way that reduces its impact. Effective management entails (a) removing contaminants from the runoff and (b) reducing the hydrologic alteration the runoff causes to stream systems. This is implemented in the form of a stormwater management ordinance with one or more performance standards. To meet these performance standards, builders must install structural and nonstructural stormwater best management practices (BMPs). Typical stormwater systems collect runoff in pipes, drain it to a central location where it may be treated and detained in a single, large pond, then release it via another pipe to a stream. This direct connection between the impervious surface and the stream bypasses the soil, which under natural conditions provides the ecosystem services of filtration and detention. The amount of directly connected impervious area (DCIA, sometimes also called effective impervious area or EIA) is a better predictor of water quality and the occurrence of aquatic organisms than the total amount of impervious area (TIA) (e.g., Hatt et al. 2004, Walsh et al. 2004b, Wang et al. 2001), which suggests that if impervious area can be disconnected from the stream, then impacts can be greatly reduced (Walsh et al. 2005a). Disconnection is accomplished by substituting infiltration BMPs for conventional BMPs and minimizing or eliminating the conveyance network. When runoff is infiltrated close to where it is generated, not only is hydrologic alteration minimized, but contaminants are greatly reduced, since natural soil has high pollutant removal rates (Barraud et al. 2005, Barraud et al. 1999, Davis et al. 2003, Mikkelsen et al. 1997).

Infiltration has been used extensively both in the U.S. and abroad for many years (Ferguson 1998). Along with the minimization of impervious cover, it is a critical component of "Low Impact Development" (Williams and Wise 2006) approaches. However, despite numerous case studies (e.g., US Environmental Protection Agency 2000) and heavy use in localized areas (e.g. Prince George's County, MD, US Environmental Protection Agency 2000), infiltration of stormwater remains more the exception than the rule. We believe that one factor contributing to its limited use is the absence of a simple, generally accepted performance standard that effectively mandates infiltration by limiting the volume of stormwater that can leave a site as surface runoff.

In this paper we propose such a standard. We first review alternative approaches, including impervious surface limits and conventional stormwater management performance standards, and discuss why these are insufficient or impractical for protecting sensitive aquatic organisms. We then introduce the new volume control standard. We describe how this standard has been applied in a regional case study in Georgia, USA, and discuss versions of the standard that might be appropriate for widespread adoption. We then discuss barriers to implementation and how they may be overcome.

Options for Effective Stormwater Management

Impervious Surface Limits

Numerous jurisdictions have responded to the problem of stormwater runoff by limiting impervious surfaces, especially for the protection of designated environmentally sensitive areas, shorelines, and drinking water supply watersheds. Most of these are local ordinances, incorporated into county or municipal zoning codes, although some originate at the state level. Some examples are described below:

- Maryland's Chesapeake Bay Critical Area Act of 1984 limits imperviousness in designated Resource Conservation Areas (RCAs) and Limited Development Areas (LDAs) to a 15% impervious limit, although some grandfathered parcels have 25% limits (Annotated Code of Maryland § 8-1808).
- Montgomery County, VA illustrates how impervious limits are typically incorporated into a zoning code. Zoning regulations specify an impervious limit for each type of land

use class, with impervious limits ranging from 20% in agricultural districts to 85% in manufacturing districts (Montgomery County Code § 10.21-10.31).

- Impervious limits are frequently incorporated into overlay zones. The city of Durham, NC created watershed protection overlay districts to preserve the water quality of the city's drinking water. This ordinance limits impervious surfaces to 6-24% of lot area within designated watershed protection districts depending on the zoned land use (Durham Zoning Ordinance § 5.5.6).
- In Georgia, the Department of Natural Resources (DNR) promulgated rules under the Georgia Planning Act of 1989 limiting impervious area in small water supply watersheds to 25%, or existing use, whichever is greater (Georgia Rules and Regulations 391-3-16).

Impervious limits have some practical limitations, however. First, most impervious limits are set higher than the threshold (10% or less) necessary to protect sensitive aquatic species, primarily because setting it sufficiently low would be unacceptably burdensome to the regulated community. Second, if not carefully designed, impervious limits can encourage sprawl by prohibiting high-density clustering. Third, on its own this approach does not provide an incentive for better stormwater management: the developer who manages the stormwater runoff from his site so that it has virtually no impact is constrained by the same impervious area limits as the developer who provides no stormwater management. Finally, impervious limits implemented on the watershed or regional level (for example, Georgia's water supply watershed regulations) frequently lack a mechanism for ensuring that the limits are applied fairly to all parcels; they are operated under a first-come, first-served approach.

Water Quality Standards and Extended Detention

Performance-based stormwater regulations typically include standards for the peak rate of discharge and the safe conveyance of flood flows from large storm events. The goal of such standards is flood prevention, not aquatic resource protection. Many recent stormwater ordinances set additional performance standards for both water quality protection and reduction of hydrologic alteration for small storm events. A good example is the model stormwater ordinance of the Metropolitan North Georgia Water Planning District (the "Metro District") (Metropolitan North Georgia Water Planning District 2004), which has been adopted by local governments across metropolitan Atlanta. Its performance standards include:

- Water quality protection: capture and treat runoff from all storm events 1.2" or less, as well as the first 1.2" of runoff for all larger storm events (together, this comprises 85% of runoff). The treatment standard is 80% removal of total suspended solids.
- Channel protection volume: provide 24 hours of extended detention for runoff generated by the one-year, 24-hour storm event (this is the provision limiting hydrologic alteration).
- Overbank flood protection: reduce the post-development 25-year, 24-hour storm event peak discharge rate to no more than the pre-development discharge rate.
- Extreme flood protection: design all stormwater management facilities to safely convey the runoff from the 100-year, 24-hour storm event.

Despite the benefits of such an ordinance, there are no guarantees that it will reduce the impacts of stormwater runoff to levels that do not lead to detrimental effects on sensitive fish species. Most importantly—and somewhat surprisingly—to our knowledge the effectiveness of 24-hour extended detention in minimizing hydrologic alteration has never been quantified. If it were quantified, we might be able to assume that use of extended detention is equivalent to a certain reduction in hydrologic alteration. Second, the water quality protection requirement mandates a reduction in contaminant loads, but not an absolute limit. If impervious cover is very high, the concentrations of contaminants—although reduced—may still be excessive for sensitive fish species. Furthermore, depending on the design of the system, runoff from small events can flow through with only minimal treatment. Researchers have identified control of those small runoff-generating events as critical to effective stormwater management (Walsh et al. 2005a).

A volume control performance standard: the Runoff Limits Program

We propose an alternative stormwater management performance standard that limits the total volume of stormwater that leaves a site as surface runoff. Limiting volumes requires infiltration, and thus management of both contaminants and hydrologic alteration through a natural process. This eliminates the need for individual performance standards for contaminant removal and extended detention, and provides greater flexibility than impervious surface limits. In a sense, however, a runoff volume limit is like an *effective* impervious area limit, because it can limit the volume of runoff to the amount associated with a certain level of impervious cover—without restricting actual TIA.

There are several ways that a runoff volume standard can be formulated. We first describe the version introduced as part of the Etowah Habitat Conservation Plan (HCP), the “Runoff Limits”. The Etowah HCP is a plan to minimize the impact of urban development on nine imperiled fish species, including three federally-protected species, in the Etowah River basin, Georgia. The Etowah region is experiencing rapid growth from the greater Atlanta metropolitan area.

Stormwater runoff from impervious surfaces is considered the paramount threat to imperiled Etowah fishes (Chapter 2), and studies have demonstrated that several of the species are sensitive to low levels of EIA (Chapter 3). One of the challenges of the Etowah HCP, however, is to manage this threat with minimal restrictions on land use activities. For the reasons described above, limitations on total impervious area were rejected as unworkable and conventional stormwater performance standards were considered insufficient. The Runoff Limits volume standard was developed as an alternative. After describing this standard we discuss alternative, simpler formulations that may be more appropriate for widespread adoption.

Under the Runoff Limits program, watersheds (designated "Priority 1") that support two fish species federally listed as endangered (*Etheostoma etowahae* and *Percina palmaris*) have the following performance standard: for storms up to the two-year, 24-hour recurrence interval, the volume of runoff that leaves a site must not exceed the volume that would occur from the site under fully forested condition, given the soils present. This is essentially equivalent to a 0% EIA limit. For watersheds (designated "Priority 2") inhabited by the third federally-listed fish species (*Etheostoma scotti)*, which appears less sensitive to EIA (Chapters 3-4), the standard provides an allowance for runoff equivalent to what would be generated by 5% impervious cover (i.e., a 5% EIA limit). Because of the strictness of these standards, local governments are also allowed under the HCP to designate a limited number of "development nodes" for commercial, industrial and other high-intensity uses. Development nodes have a less strict performance standard: runoff volumes are limited to the volume that would occur if impervious cover were half what is actually present. For example, a site with 60% total imperviousness must reduce its volume of runoff to the amount that would come from the site if it had only 30% imperviousness, and the remainder forested. The size and locations of development nodes are limited so that they do not cause excessive impacts to sensitive fish species, as determined by predictive modeling (Chapter 6).

Volume calculations are made using the curve number method (Soil Conservation Service 1975). Applying the performance standard to a site is a straightforward 3-step process:

(1) Calculate the volume of runoff from the site using the curve number of a forest in good condition for the regional two-year, 24-hour design storm.

(2) Calculate the volume of runoff from the site using the curve numbers of the post-development conditions for the regional two-year, 24-hour design storm.

(3) Subtract the forested volume from the post-development volume. This is the volume of runoff that must be managed through BMPs for infiltration and evapotranspiration.

The above procedure applies to Priority 1 areas. The same procedure is used for Priority 2 areas and development nodes, except that the calculation in step 1 is modified accordingly.

In addition, the Runoff Limits program includes two auxiliary requirements. The first is that all post-development runoff generated by impervious surfaces must be directed through stormwater BMPs. Without this requirement, the primary performance standard could be met by over-infiltrating stormwater from a portion of the site, and allowing stormwater from another portion of the site to flow off untreated. With this requirement, all runoff must be directed to BMPs and given the opportunity to infiltrate, eliminating bypass of small storms (there may still be overflow from large and back-to-back storms). The second requirement is that BMPs should be distributed throughout the site to the extent practicable, to avoid concentrating excessive volumes in one or a few locations.

To meet the standard, developers can choose from a large menu of BMPs for infiltration and (to a lesser extent) evapotranspiration described in the Etowah HCP Runoff Limits Manual (Carter et al. 2006). These include bioretention areas, infiltration trenches, subsurface infiltration beds, green roofs, dry wells, infiltration basins, porous pavements, and variations of these. The cost of implementing these structural stormwater BMPs creates a significant incentive for developers to design or redesign sites to minimize impervious surfaces and to maximize forest cover. For this reason the Runoff Limits program can be viewed as a performance-based approach to low impact development. A developer can either choose to create a low-impact site design that requires fewer structural BMPs, or a conventional site design that requires more structural BMPs.

Many of the local governments participating in the Etowah HCP are obligated by state law to adopt ordinances consistent with the Metro District model ordinance described above. For this reason, the Runoff Limits performance standard is added to this ordinance as a fifth standard that must be met for development within Priority 1 and Priority 2 areas. This generates some redundancy, since the Runoff Limit standard plays the same role as the contaminant removal and channel protection volume standards. Fortunately, under most conditions a stormwater system designed to meet Runoff Limits standards will also meet the other performance standards, without the need for additional structures or measures (except in development nodes, where a hybrid system is required). The Etowah HCP model stormwater ordinance incorporating the Runoff Limits program is available at www.etowahhcp.org.

General Application of a Volume Control Performance Standard

The Runoff Limits Program under the Etowah HCP has a certain degree of complexity necessary to meet specific goals regarding protection of federally listed fish species. Jurisdictions that are not constrained in such a way but wish to implement protective stormwater management measures could adopt a somewhat simplified version. We recommend a standard similar to the one for Priority 1 areas described above: for storms up to the two-year, 24-hour recurrence interval, the volume of runoff that leaves a site must not exceed pre-development volumes. This differs from the Runoff Limits requirement in that the pre-development condition, rather than forested condition, is used as the baseline volume of stormwater runoff (see "Barriers to Implementation," below). This standard is appropriate for low to moderate density residential zones, where it can serve as the sole stormwater management performance standard (except for the safe conveyance of very large storms, which should apply universally). However, we recommend exempting high density zones such as central business districts from this standard, provided that the boundaries of such zones are carefully delineated with respect to aquatic ecosystems targeted for protection. In place of the volume control standard, a suite of performance standards for contaminant removal, channel protection and flood protection (such as those the Metro District standards described above) should be employed.

Properly implemented and enforced, this two-tiered system should maintain EIA levels within the tolerances of sensitive aquatic species across all areas designated as low-moderate density. Of course, those tolerances will be exceeded in designated high density areas where the standard does not apply, just as they are in development nodes under the Runoff Limits program. We regard some degradation of streams as inevitable in highly urban areas, and suggest that it is preferable to encourage density in designated areas and plan for the consequences, rather than risk the possibility of degradation across entire watersheds.

Similar Programs

This is not the first volume control performance standard to be developed. One of the earliest was a provision within Maryland's Chesapeake Bay Critical Area Act, adopted in 1984. Among other requirements, the Act mandates that, within designated limited development areas and resource conservation areas, "development may not cause downstream property, watercourses, channels or conduits to receive stormwater runoff at a higher volume or rate than would have resulted from a two- or ten-year storm, whichever is more restrictive, were the land in its predevelopment state" (Code of Maryland Regulations § 27.01.02). To properly meet this standard requires matching pre-development runoff volumes. Another example is from Huntersville, North Carolina. Huntersville requires that LID or a combination of LID and conventional stormwater BMPs be used to control and treat the increase in runoff volume from pre-development conditions for the 2-year, 24-hour storm (City of Huntersville, NC Zoning Ordinance § 8.17). This appears similar to the volume control performance standard recommended here, except that it permits at least a portion of the volume to be discharged as surface runoff.

Other examples exist. However, for various reasons, volume control performance standards have not yet moved into the mainstream of stormwater management. The next section explores some of the barriers to widespread implementation of these standards, and how such issues are being addressed under the Runoff Limits program.

Issues and Barriers to Implementation

The Runoff Limits program was developed with extensive participation of consulting engineers, local government officials, developers, and other interested stakeholders. In meetings held over the course of a year, participants raised concerns and identified real or perceived barriers to implementation of the program, which were subsequently addressed. Many of these are general issues likely to surface in other localities where a volume control performance standard is under consideration.

Soils

The first response of many engineers to a volume-based standard is that "infiltration is impossible with our soils." Certainly, soils vary greatly in their properties and some are relatively poor for infiltration. However, the proposed performance standard takes into consideration existing soil type in the calculation of pre-development runoff volumes. The less pervious the soil type, the higher the pre-development runoff volume, and the lower the volume that must be infiltrated. Of course, there are some conditions that preclude the use of infiltration: bedrock very close to the surface, water table very close to the surface, or soils with infiltration rates less than 0.1 inches per hour. Infiltration BMPs should not be sited in such locations, and we recommend providing a variance procedure for sites where 80% or more of the area is characterized by such conditions. We recommend that this be the *only* grounds for a variance, however, if the objective of the ordinance is protection of sensitive aquatic species.

Cost

In some cases, the cost of a system of infiltration BMPs can be substantially more than that of a conventional stormwater detention pond (Nelson 2006). Conversely, the cost advantage of an infiltration system is that it can eliminate the need for much of the conveyance network. Piping can be a major cost of a stormwater management system, and once this is taken into account, a system of infiltration BMPs can be less costly than a conventional system with a detention pond and extensive piping (Conservation Research Institute 2005). Under a worst-case scenario where both infiltration BMPs and conventional BMPs are constructed side-by-side, adding infiltration may add about $1100 per home for residential development (Nelson 2006).

The Question of Baseline for Calculating Runoff Volumes

A significant topic of debate among the consulting engineers and developers who helped guide the creation of the Runoff Limits Program was whether to use existing condition or forested condition as the baseline for calculating runoff volumes. Forest was the dominant land cover prior to intensive cultural modification, although substantial portions of the basin have been in agricultural use for two centuries. Many stakeholders claimed it was unfair to use forested condition as the baseline, because this would require a developer of pasture or crop land to *increase* the amount of infiltration over what was occurring under existing conditions. Ultimately the decision was made to use a forested baseline because it was more protective of the species and eliminated many potential avenues for abuse, such as speculative clearing. Due to the intense opposition this generated, however, other jurisdictions may wish to use existing condition as a baseline, except in cases where imperiled species protection is the primary management objective.

Engineering Specifications

Until recently, a barrier to implementation of a volume control performance standard was the limited availablity of engineering specifications for infiltration BMPs. This is no longer a major issue. We have developed a manual for the Runoff Limits program (Carter et al. 2006) that is readily adaptable to other jurisdictions. This manual is designed to complement the Georgia Stormwater Management Manual, which includes specifications for conventional stormwater BMPs. Many other states and jurisdictions have added infiltration practices to stormwater manuals as well, so there should be no lack of engineering specifications available for supporting the design of BMPs to meet this performance standard.

Training

A significant barrier to implementation of the Runoff Limits program has been the lack of training and experience for both engineers and installers. Under the Etowah HCP this problem is being addressed through workshops, with training by both local and outside experts. These workshops may continue on a long-term basis and may eventually develop into a certification program. Engineers involved in the development of the Runoff Limits program also recommended mandatory oversight of critical phases of installations, preferably by local government engineers or contracted engineers.

Conclusions

A runoff volume performance standard is a simple, elegant solution to the problem of stormwater runoff. Properly implemented, it should encourage infiltration of stormwater close to where it is generated, minimizing hydrologic alteration and maximizing contaminant removal. It provides inherent incentives for minimizing imperviousness, retaining forest cover and employing good site design techniques, while simultaneously providing developers with flexibility. Combined with other land use regulations, such as erosion and sedimentation controls and riparian buffer ordinances, a runoff volume performance standard has the potential to protect sensitive aquatic organisms from the effects of low to moderate density urban development.

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CHAPTER 6

MANAGING FOR IMPERILED SPECIES BY INTEGRATING PREDICTIVE MODELING AND LAND USE POLICY[6]

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ABSTRACT

Predictive models are essential for forecasting the outcomes of imperiled species management policies, especially for species on private lands where regulatory policies must be well justified. The favored approach for modeling imperiled species in the US has been population viability analysis, but for many species there are insufficient data to parameterize such models. We propose an alternative approach that uses simple, statistical models to predict and guide the outcome of management policies. The approach involves (1) identifying key stressors to the species; (2) developing models relating stressors to species occupancy/abundance; (3) developing a management policy to limit stressors; (4) setting minimum levels of species occupancy or abundance to be maintained; (5) using models to forecast occupancy/abundance under future management scenarios, and adjusting policy as necessary; and (6) implementing an adaptive management program to improve models as new data are collected and to adjust policies as necessary. We illustrate the approach with a case study of three imperiled fish species threatened by runoff from impervious surfaces, which is being managed as part of a Habitat Conservation Plan. We show that the approach can be applied even in cases of limited data and provides an effective way of supporting difficult conservation decisions.

INTRODUCTION

Effective conservation of imperiled species demands the ability to predict future population trends and to forecast the outcome of management policies. This is especially critical when managing for species on private lands, where policies to limit land use activities are likely to engender significant landowner opposition (Beatley 1994, Langpap and Wu 2004, Peterson et al. 2004). In such cases, predictive modeling can provide invaluable assistance in identifying policies that are effective in their species management goals and no more burdensome to landowners than necessary. This is challenging, but the difficulty is compounded when data limitations do not permit precise parameterization of models: how do managers establish a rational framework for decision making when the nature of species response is poorly known?

Few studies have tackled this thorny problem, although in recent years there has been considerable research on species modeling to support conservation decisions (Beissinger et al. 2006). A preferred modeling approach for guiding management of imperiled species in the U.S. is population viability analysis (PVA) (Morris and Doak 2002). Indeed, a U.S. Fish and Wildlife Service (FWS) database of modeling approaches for threatened and endangered species reveals that a PVA was performed in nearly all cases where any form of modeling was conducted (Jean Cochrane, FWS, pers. com.,10/25/2006). These models require population time series data in order to estimate growth rates and stochasticity, which are used to compute the probability of extinction or quasi-extinction over a specified time period (Morris and Doak 2002). However, Moilanen et al. (2005) note that “for most species in most landscapes, insufficient ecological data, population parameters or habitat distribution information are available to allow the application of simulation modeling” such as PVA, and several researchers have cautioned against over-reliance on PVA when data are lacking (Beissinger and Westphal 1998, Ellner et al. 2002). Population viability analysis *has* been applied with reasonable success to many species, such as key deer (Monroe County et al. 2003), red-cockaded woodpeckers (Schiegg et al. 2005), and even Pacific salmon with only limited available data (Ellner and Fieberg 2003). Nevertheless, there is great need for alternative modeling approaches to support decisions when management action is required despite a lack of extensive ecological and population parameter data.

Our purpose is to introduce such an alternative. The first step of the proposed approach is to use statistical species-habitat models, which only require species distribution data, to relate species patch occupancy or abundance to one or more key landscape-scale stressors, such as land use activities on private lands. Then, a policy tool is developed to limit the stressor(s), and predictive modeling is used to evaluate the outcome of the policy on the species in a spatially explicit manner. Evaluation may proceed iteratively with alternative scenarios to develop management options that limit species loss to acceptable levels (which must be defined). The policy tool should be designed to allow for adjustment, so that as additional data become available and predictive models are updated, the policy can be corrected via adaptive management. This allows management actions to begin immediately, despite considerable uncertainty in species responses and future conditions, guided by a model that reflects best available knowledge at the time.

We apply this approach to the conservation of three federally protected species of fish in a southeastern US river system, as part of the development of a multispecies, multi-jurisdiction

Habitat Conservation Plan (HCP). The species, whose ranges are mostly on private lands, are threatened by rapid outward growth from a major metropolitan region. Our available data are insufficient for a PVA, but we use an extensive set of occurrence records for two species to develop models relating occurrence or abundance to a key stressor, effective impervious area (EIA). For a third, data-limited species, we use a Bayesian modeling framework whereby a parameter estimate from a surrogate species serves as a “prior” on the coefficient for EIA. We next propose a policy to restrict the stressor through a program of locally-implemented stormwater runoff limits. To accommodate high-density development, the program includes a provision for development nodes in which less strict standards apply. We then use the models to construct spatially explicit predictions of occupancy or abundance basin-wide, to test the policy and to identify appropriate development node locations. Finally, we consider the role of adaptive management and monitoring in improving predictive performance over time.

STUDY AREA, SPECIES AND THE ETOWAH HABITAT CONSERVATION PLAN

The Etowah River is a major tributary of the Coosa River system in the Mobile River Basin (Figure 6.1). It drains 4871 km^2 of land, with the upper half of the basin primarily in the Piedmont physiographic province (with a small area of Blue Ridge in the headwaters), and the lower half of the basin primarily in the Valley and Ridge physiographic province. A portion of the Etowah lies within the Southern Appalachian highlands, a global hotspot of fish endemism (Warren et al. 2000). In addition to four locally endemic species, the Etowah supports another eight species endemic to the larger Mobile River Basin (Burkhead et al. 1997).

Three of these fish species, all members of the family Percidae, are federally listed under the US Endangered Species Act and are the subjects of this study. All are small (total length $\leq$ 80 mm), benthic fishes commonly found in shallow riffles: *Etheostoma etowahae*, the Etowah darter; *E. scotti*, the Cherokee darter; and *Percina antesella*, the amber darter. The Etowah darter and amber darter are federally listed as endangered and occur in the Etowah River mainstem and mid-sized tributaries (Etowah darter) or the lower reaches of large tributaries (amber darter). The Cherokee darter is endemic to small streams of the Etowah River system and is federally listed as threatened. Genetic and morphological variation within the Cherokee darter supports the recognition of three distinct evolutionarily significant units (ESUs), roughly corresponding to populations in the upper, middle and lower parts of the Etowah basin (Storey 2003).

This study is motivated by the development of a multispecies, multi-jurisdiction Habitat Conservation Plan (HCP) for the imperiled fish species of the Etowah basin. Covered species include the three federally listed species and six other fishes (Table 6.1). The focus of the HCP is on managing urban growth through a set of policies implemented by the seven participating county and numerous participating municipal governments. Developers and landowners who adhere to the policies are covered by the HCP by extension. The stormwater runoff management regulations are the most critical of the policies, but others include controls on erosion and sedimentation from construction, road crossing design requirements to maximize fish passage, and riparian buffer regulations. Details are available at www.etowahhcp.org.

PROPOSED APPROACH TO INTEGRATIVE PREDICTIVE MODELING AND MANAGEMENT POLICY

Our approach involved six steps:

1) Identify key stressors to the species.
2) Develop models relating stressors to species occupancy/abundance.
3) Develop a management policy to limit stressors.
4) Set minimum levels of species occupancy or abundance to be maintained.
5) Use models to forecast occupancy/abundance under future management scenarios, and adjust policy as necessary to meet minima established in step (4).
6) Establish an adaptive management program to improve models as new data are collected and to adjust policy as necessary.

As most of the details of steps 1-3 are published elsewhere (Chapters 2-5), both the methods and the most relevant results of those steps are summarized only briefly here. Steps 4, 5 and 6 are described in greater detail.

1. Stressors

An analysis of stressors identified stormwater runoff from impervious surfaces as the premier threat to the target species because of increasing urban development in the Etowah basin (Chapter 2). Accordingly, a central focus of the Etowah HCP is in managing stormwater runoff, of which EIA is considered a useful indicator (Walsh et al. 2005, Walsh et al. 2004). A study of five Etowah fish species found that for four of the species, EIA was a strong predictor of occupancy even after accounting for historic land use, incomplete detectability and spatial autocorrelation (Chapter 3).

2. Models Relating Stressors to Species

Three slightly different methods were used in developing models for each of the three federally-listed species. For the Etowah darter, we fit occupancy models accounting for incomplete detectability (MacKenzie et al. 2002) and spatial autocorrelation (Latimer et al. 2006, Snijders and Bosker 1999), selecting the best-supported model (Table 6.2) from a set of candidate models based on cross validation predictive success (Chapter 3). We attempted a similar approach with the Cherokee darter, but the best-supported model did not show a relationship between EIA and occurrence, although examination of the data showed a relationship between EIA and *abundance*. Consequently, we fit models that simultaneously estimated occurrence, abundance and detection probability, with EIA as a covariate on abundance (Chapter 4). The best of several competing models was selected using Akaike's Information Criterion (Table 6.2).

Our ability to develop a model for the amber darter was confounded by the species' limited distribution. It occurs in significant numbers in only two localities: the mainstem of the Etowah and the mainstem of the neighboring Conasauga River (small populations also inhabit the lower reaches of three large tributaries of the Etowah). Because of this restricted range, it is impossible to infer the sensitivity of the species to impervious cover based on its existing distribution. The species is absent from both rivers near urban areas (Canton in the Etowah and Dalton in the Conasauga), but this could be due to an effect of increasing stream size or decreasing elevation rather than imperviousness. Therefore, alternative models to explain the species' occurrence could not be distinguished using existing data. However, species related to the amber darter and with similar life histories display sensitivity to urban impacts (Chapter 3), and we reasoned there was a strong likelihood that the amber darter was sensitive as well. To develop a useful predictive model, we assumed that the amber darter had sensitivity to EIA similar to that of *Percina palmaris,* a congeneric, syntopic species for which we had a predictive occupancy model (Chapter 3). Using Bayesian modeling in WinBUGS 1.4 (Spiegelhalter et al. 2003), we used the posterior distribution of the EIA covariate parameter for *P. palmaris* as a prior for the EIA covariate parameter for *P. antesella.* The effect was to model the occurrence based on data from both species. The model contained only one other variable, drainage area (Table 6.2). Modeling methods were otherwise the same as in Chapter 3.

3. Management Policy: The Runoff Limits Program

As the management policy for limiting EIA, we defined a new stormwater performance standard called "runoff limits" (Chapter 5). The policy limits the volumes of precipitation from small storms permitted to leave a site as surface runoff. To meet the policy, developers must use stormwater infiltration management practices, which tend to have very high contaminant removal levels (Barraud et al. 1999, Davis et al. 2003, Walsh et al. 2004). By managing both hydrologic alteration and stormwater contaminants, the Runoff Limits program provides a means of limiting *effective* impervious area, without placing any limits on actual impervious area (Chapter 5). This performance-based approach provides developers with substantial flexibility and was far more acceptable to the regulated community than potential alternatives, such as strict limits on development density or impervious cover itself.

Three levels of control were established under the Runoff Limits Program, each of which applies in designated locations corresponding to the distributions of the federally-protected fishes within the Etowah basin. “Priority 1” areas (Figure 6.2) encompass the entire combined ranges of the Etowah and amber darters, and nearly the entire range of all other species covered by the Etowah HCP, except the Cherokee darter. The runoff limit for a site in a Priority 1 area is set equal to that of an undeveloped, forested site. That is, the volume of runoff from the site cannot exceed the volume of runoff that would occur under a forested condition, for small storms (up to the two year recurrence interval), given the soils present. “Priority 2 areas” (Figure 6.2) cover most of the remaining range of Cherokee darters (except for a few small, isolated populations in the Little River system) as well as all other tributary watersheds of the Upper Etowah mainstem. Because the modeling revealed that the Cherokee darter is not as sensitive to EIA as other species (Chapters 3-4), the runoff limit is set to the equivalent of 5% impervious cover in Priority 2 areas. Therefore, new development is required to employ stormwater management practices to make a site act as if it has no more than 5% impervious cover (and the remainder forested). In areas not designated Priority 1 or Priority 2, the Runoff Limits do not apply, although conventional stormwater regulations are still in place.

Local governments are also permitted to establish development nodes within Priority 1 and Priority 2 areas to make it less costly to construct commercial, industrial and high-density residential development. The runoff limit for a development node is set at 50% of the actual impervious cover for the site. For example, a site with 60% impervious cover has to reduce the volume of runoff to the amount expected from the site if it had only 30% impervious cover (and the remainder forested). This is the component of the Runoff Limits that contains the greatest amount of flexibility. Local governments are permitted to establish development nodes wherever they wish, provided that predictive modeling demonstrates that they will not cause a decline in species occupancy or abundance that exceeds the "Take Limits," described below. Thus, node designation is the element of the management policy that *requires* predictive modeling to implement.

4. Minimum Levels of Occupancy/Abundance: The "Take Limits"

The models provide a mechanism for predicting patch occupancy (for Etowah darters and amber darters) and patch occupancy and abundance (for Cherokee darters) under changing levels of EIA. However, prior to forecasting patch occupancy/abundance under future conditions, it was necessary to consider minimum levels of occupancy/abundance required for ensuring the long-term survival of the species. Under an HCP, the Endangered Species Act (ESA) allows "incidental take" of listed species, but it must be demonstrated that this take "will not appreciably reduce the likelihood of the survival and recovery of the species in the wild" (ESA §10(a)(2)). Thus, approval of the Etowah HCP is unlikely unless it can be well established that the predicted take will not jeopardize the survival of the three federally listed species.

We solicited expert opinion from a Scientific Advisory Committee to set limits on the decline in occupancy and abundance. Given the limited data available on the target species, the experts (listed in Acknowledgements) needed to make certain assumptions about species population and metapopulation dynamics in order to determine appropriate "take limits." These assumptions were:

- Patches that currently have high occupancy or abundance are most valuable to the population. This was equivalent to assuming that source/sink dynamics (Pulliam 2000), if present, do not result in higher occupancy or abundances in population sinks. Based on this assumption, the committee recommended that at least a subset of high occupancy/abundance patches should be maintained at high occupancy/abundance levels, and at least half such patches should be maintained at moderate occupancy/abundance levels.

- High occupancy/abundance habitat should be maintained throughout the species' ranges to minimize the risk of synchronous decline across populations. Toward this end, the Etowah darter range was assumed to comprise five "population areas," representing distinct tributary watersheds (Figure 6.3). The amber darter range was assumed to comprise four population areas, representing three sections of the mainstem and a disconnected tributary; and the Cherokee darter range was divided to correspond to its three ESUs (see Appendix A).
- Connectivity among patches should be maximized to minimize interruption of dispersal among presently connected habitat patches. Although the level of movement among patches was unknown, the committee assumed that movement was greater than zero and that movement barriers should be minimized. They assumed that patches of very low occupancy/abundance might constitute barriers, and determined that no patches should be allowed to decline to this level (note: other forms of barriers, including road crossings and dams, are addressed by other HCP policies and are not discussed here).

Based on these principles and the results of preliminary modeling, the Scientific Advisory Committee established numeric limits for each species. We provide the Take Limits for the Etowah darter here as an example. The Take Limits for the other species are given in Appendix A.

The Etowah Darter Take Limit requires that the projected occupancy of the Etowah darter meets the following criteria, as indicated by modeling using the most current version of the Etowah HCP Species Occurrence and Abundance Model:

1. *At least 30% of stream miles in which probability of occurrence is greater than or equal to 80% under 2006 conditions must maintain a predicted probability of occurrence greater than or equal to 80% under the buildout scenario.*
2. *At least 50% of stream miles in which the 2006 probability of occurrence is greater than or equal to 80% must maintain a predicted probability of occurrence greater than or equal to 50% under the buildout scenario.*

Conditions 1 and 2 apply to five designated population areas that have high probability of occurrence and known occupation under 2006 conditions (Figure 6.3):

   a. *Headwaters of the Etowah River mainstem*
   b. *Upper Etowah River mainstem and lower reach of Shoal Creek (Dawson Co.)*
   c. *Amicalola Creek system*
   d. *Long Swamp Creek system*
   e. *Raccoon Creek*

3. *100% of streams with a probability of occurrence 25% or greater under 2006 conditions must maintain a probability of occurrence above 5% under the buildout scenario.*

5. Forecasting

Methods

For each species, we used the selected model to predict occupancy (for Etowah darters and amber darters) or both patch occupancy and abundance (for Cherokee darters) for stream reaches across the Etowah basin, based on the values of the model covariates (Table 6.2). Predictions were made for three scenarios: (1) current conditions, (2) 50-year buildout with runoff limits management policy ("HCP scenario") and (3) 50-year buildout without management policy ("no-action alternative"). For buildout, we assumed complete conversion of forest and agricultural land uses to urban and suburban uses, except for lands currently under permanent conservation protection. The scenarios differed only in their values for EIA.

Data preparation was performed using ESRI ArcMap 9.0 Geographic Information System (GIS). Stream reaches were clipped according to a custom watershed coverage, created by subdividing the USGS hydrologic unit code level-12 watersheds at stream confluences (Figure 6.1). Watershed-scale covariates were measured as the dominant class (geologic variables), the maximum value (watershed area and dlink) or the mean value (elevation). Some covariates were measured at the scale of tributary systems (Table 6.1), which are higher-level watersheds shown with heavy outline on Figure 6.1. Mean EIA was measured within a 1 km or 1.5 km radius of stream reaches and averaged by watershed. In calculating EIA, stream reaches that were known to be too small to support the fish species were excluded. For *E. scotti* these were streams less than 0.5 km^2 drainage area, while for the other two species these were streams less than 10 km^2 drainage area.

Estimating EIA under 2006 conditions. The 2001 National Land Cover Database Imperviousness layer (US Geological Survey 2003)—the most recent available imperviousness data set—was used as the baseline for total impervious area (TIA). This was a 30m resolution (i.e., cell size = 30m) raster coverage derived from a supervised classification of LandSat satellite imagery. It was converted to EIA based on an empirical relationship between TIA and EIA, which we derived from analysis of high-resolution aerial photographs for 15 sites in the Etowah Basin of 25 to 70 hectares each in size (Chapter 3). We updated the 2001 EIA coverage to 2006 conditions by analyzing county existing land use maps, parcel maps and recent aerial photography to identify parcels that had developed since 2001. We applied appropriate EIA values to each identified location of recent development based on lot density, aerial photography and literature values (Capiella and Brown 2001).

Estimating EIA under HCP and no-action scenarios. We assumed that currently developed cells (EIA>0 under 2006 conditions) would not increase or decrease in EIA value at buildout under the HCP and no-action scenarios, but that all undeveloped cells would be developed unless they lay within designated conservation areas. This was equivalent to assuming complete infill with no redevelopment, a necessary simplifying assumption. To calculate EIA values for currently undeveloped cells, we first estimated buildout TIA parcel-by-parcel based on zoning class, future land use category, and whether the parcel lay within a node, conservation area, Priority 1 area or Priority 2 area (Table 6.3). Zoning and future land use maps were provided by participating counties and cities. We then converted TIA values to EIA based on assumed benefits of the runoff limits policy (for the HCP scenario) or conventional stormwater policy (for the no action scenario) in reducing hydrologic alteration and contaminant concentrations (Table 6.3; details of assumptions and methods are provided in Appendix B). These values were then applied to cells to create a raster map of predicted EIA at watershed buildout for each scenario.

Conservation areas were mapped from a statewide database of protected areas (Natural Resources Spatial Analysis Laboratory 2003), augmented by additional permanently protected lands identified by local governments and stakeholders. Development nodes were identified by local governments based on zoning maps and future land use maps. Generally, parcels identified for commercial, industrial and high-density residential uses were considered development nodes. In some cases, preliminary runs of the predictive model indicated that nodes needed to be reduced in area or relocated so as not to exceed the Take Limits. We made the necessary adjustments in close consultation with the participating jurisdictions, and were ultimately able to identify a set of development node locations that was acceptable to the local governments and which met the Take Limits.

Calculating Take. To estimate the take of each species under the HCP and no-action scenarios, we estimated the decline in modeled occupancy/abundance from 2006 conditions. We only modeled changes within the known range of each species as of 2006. We multiplied occupancy values for each reach by the reach length, which produced an estimate of occupied stream habitat for each reach. For Cherokee darters, raw abundance estimates reflected the number of individuals in a standard sampling length of 150 m. Therefore, for each reach, we adjusted the abundance by the length of the reach and the occupancy of the reach to produce an estimate of total individuals (example: for a 500 m reach with an estimated abundance of 80 and an occupancy of 70%, the estimated total abundance would be 500 m*(80 fish/150 m)*.7= 187 fish).

Modeling Uncertainty and Sensitivity. One of the advantages of Bayesian methods is that parameter uncertainty can be propagated forward through the model, so that the final predictions reflect all of the uncertainty of the inputs. In the case of our models, the predictions of occupancy and abundance were made directly from the full distributions of the covariate parameter estimates, rather than point estimates of their means. The exception was the Cherokee darter model, which was fitted using conventional maximum likelihood methods rather than Bayesian methods due to software limitations, and for which point estimates were used in predictions. We tested the contribution of uncertainty in each parameter of the Etowah darter and amber darter models to the overall uncertainty of the prediction by systematically holding all parameters but one to their mean values and observing the change in mean standard error of predicted occupancy. We also examined the variance parameter of the negative binomial abundance distribution of the Cherokee darter model as an indicator of the uncertainty in its predictions.

A second major source of uncertainty is in the assumptions used to calculate EIA under the HCP and no action scenarios (Appendix B). It is possible to represent these EIA estimates as probability distributions, as we did with the covariate parameters; however, while this increases the uncertainty in the final model outputs, it does not tell us of the consequences of a systematic bias in the EIA inputs. To explore the latter issue we analyzed the sensitivity of the Etowah darter modeling results to changes of +10%, +25%, -10% and -25% in estimated EIA values. We analyzed only the Etowah darter model because it showed considerably more sensitivity to EIA than the other two species models, and was the determining factor affecting development node size and location.

Forecasting Results

We provide results including maps for the Etowah darter here as an example, and summary results for the amber darter and Cherokee darter; maps showing results of predictive modeling for the amber darter and Cherokee darter are provided in Appendix C.

Etowah Darter. Our modeling predicted that the amount of occupied stream length would decline from 2006 levels by about 23% under the HCP scenario and 84% under the no-action alternative (Table 6.4 and Figure 6.4). The Take Limits were met for each of the five population areas under the HCP scenario, but were violated in all population areas under the no-action alternative. Model results are shown spatially for 2006 conditions (Figure 6.5) and under the HCP scenario (Figure 6.6); results under the no-action alternative are not mapped. Much of the decline under the HCP scenario was projected to occur in Pickens County and Dawson County, two jurisdictions that have large areas of Etowah darter habitat and substantial pressure for high-intensity development along road corridors adjacent to that habitat. The areas that were predicted to experience the greatest percentage reduction in habitat—the Etowah Middle Mainstem (Forsyth and Cherokee Counties), Smithwick Creek and Stamp Creek—are not considered major population areas essential to the survival of the Etowah darter (see step 4, above).

Amber Darter and Cherokee Darter. Amber darters were predicted to decline by an estimated 11% of occupied habitat under the HCP policy. Under the no-action alternative, 61% of the habitat was expected to be lost (Figure 6.7). Cherokee darter abundances were estimated to decline by an overall 21% under the HCP scenario, with most of the losses accruing to the middle and lower ESUs. Under the no-action alternative, take was estimated at 43% (Figure 6.8). Model results indicated that the Take Limits would be met for both species under the HCP scenario, but for neither species under the no-action alternative.

Model Prediction Uncertainty and Sensitivity. Predictions of occupancy and abundance were characterized by a high level of uncertainty. For the Etowah darter, the distributions of occupancy for many stream reaches were broad and flat (Figure 6.6, inset) or even bimodal. The contributions of covariates to the variance of predictions were directly proportional to the variances of the parameter estimates for the coefficients. The coefficient parameter estimate with the greatest variance was EIA, followed by dlink, watershed area, slope and percent of tributary system in reservoirs (Table 6.5). The Cherokee darter modeling also revealed a great deal of uncertainty, but in the form of unexplained variance in the abundance estimates. Abundances were modeled with a negative binomial distribution, which has a free parameter for variance. This variance parameter was estimated at 0.78, which means that for a mean abundance of 100, the 90% confidence interval ranges from 2 to 328. This suggests that much of the variance in abundances is unexplained by the model.

The sensitivity analysis examined changes in Etowah darter occupancy if buildout EIA were underestimated or overestimated by the assumptions we employed. We found that a 10% increase in EIA values would result in an additional 2% decline in occupancy (take), although the scenario still would meet the take limits for each of the five population areas (Table 6.6). If EIA values were 25% higher, the decline in occupancy would be 5% greater than estimated in the HCP scenario, and two of the five population areas would not meet the take limits (Table

6.6). If EIA values were 10% or 25% less than expected, estimates of occupancy would be proportionately higher, and take limits would be met (Table 6.6).

6. Adaptive Management.

Adaptive management is a required component of habitat conservation plans (Wilhere 2002). Adaptive management allows policies to be adjusted as additional data are collected and understanding improves of the relationships between stressors and species. The Etowah HCP employs a passive adaptive management approach (Walters and Hilborn 1978) that specifies the annual collection of biological monitoring data at fixed and floating sites within the range of target species, including locations where development is occurring. These data will be used to evaluate the relative support for model predictions. Subsequently, these new data will be added to the existing sets used to run the models. With each addition, parameter estimates should increase in precision and decrease in bias (assuming model assumptions are met), which in turn will increase the accuracy of model predictions. Therefore, predictions that are initially based primarily on assumptions will gradually acquire a solid foundation in empirical knowledge, while policies are adjusted to reflect the improved understanding. In addition to biological monitoring, the Etowah HCP includes compliance monitoring to assess performance in governments' implementation of HCP policies and to identify provisions that require adjustment. Other potential adaptive management actions range from minor corrections of ordinance language to addition of new (but previously identified) policy provisions. A separate program of research planned for selected subwatersheds aims to improve understanding of the mechanisms by which stormwater runoff impacts fish species by studying the effects of different development and stormwater management practices. These studies can be viewed as a form of "active" adaptive management conducted on a small scale, and their results have potential to improve the estimates of buildout EIA and otherwise improve both predictive modeling and mechanistic understanding of the system.

DISCUSSION

We have presented an integrative approach to predictive modeling and stressor management that can be applied to imperiled species in the absence of extensive ecological and population parameter data. The approach can be used even in the face of considerable uncertainty in relationships between stressors and species occupancy or abundance, provided that adaptive management measures are incorporated into the policy to allow improvement as additional data are collected. As demonstrated with the case study of the Etowah HCP, this approach permits management action to begin immediately, without waiting for certainty in the understanding of relationships. In our example, the predictive modeling provides an empirical justification for managing land use impacts and for restricting development nodes to designated locations in the Etowah. Based on the modeling, a ‘no action’ scenario would lead to large declines in occupancy and abundance of the target species, while under the HCP scenario declines are considerably lower and not expected to jeopardize the survival of the species. The approach is general and applicable to other species, provided a stressor can be identified and managed with an appropriate policy.

As demonstrated, the method can be applied even when a species’ response to a stressor is not only uncertain, but unknowable from field observation data. This could be due to limited distribution (as was the case with the amber darter), extreme rarity, difficulty in sampling, or a simple lack of data. In such cases, we believe it is reasonable to assume that a species will respond similarly to a related surrogate that occupies similar habitats and has broadly similar characteristics. Bayesian methods provide a quantitative means of incorporating data from a surrogate, while still allowing the data from the species itself to be expressed. Under this approach the assumptions are clearly stated and the model becomes a hypothesis that can potentially be refuted by further data collection.

Our approach relies heavily on passive adaptive management, “a scientifically rigorous process of formulating predictive models, making policy decisions based on those models, and revising the models as monitoring data become available” (Wilhere 2002). This is different from active adaptive management, in which management actions are conducted as deliberate experiments (Walters and Hilborn 1978). Although active management has been promoted as superior to passive management (Wilhere 2002), we argue that it is not terribly well suited to a land use regulation context. In the case of the Etowah HCP, it was a difficult challenge for local governments to agree to HCP policies, and both government officials and the regulated community wanted assurance that the policies were necessary, appropriate, and likely to be effective. They fully understood the need to adjust policies as new information was collected, but they were not willing to consider implementing regulations on an “experimental” basis. We suggest that a passive adaptive management approach is entirely appropriate and necessary in such a circumstance. However, we do recommend supplementing the approach with small-scale manipulations and experiments to answer questions that can reduce uncertainty in assumptions and improve mechanistic understanding.

In order for the adaptive management approach to work, there must be flexibility and room for adjustment in the parameters of the policy. In the case of the runoff limits policy, there are two major points of flexibility. The first is the ability of local governments to add new development nodes, which could either be constrained or relaxed if new data show that species are more or less sensitive than currently thought. The second is the runoff limits performance standards themselves. For example, the "5%" Runoff Limit standard for development in Priority 2 areas could be tightened to 3%, or relaxed to 7% (or other appropriate values), as the responses of the fish species are better defined.

In our example, we treated the identification of development nodes as a relatively simple problem. However, our initial dichotomous management question—how to maximize imperiled fish population viability while minimizing costs to developers and local governments—is really an optimization problem. Indeed, considering all theoretical combinations of potential EIA limits and differing spatial arrangements for applying those limits, this is quite a challenging problem when approached abstractly. In reality, however, the comprehensive plans and zoning codes of local governments, coupled with existing locations of infrastructure, combine to severely constrain potential options. For example, property zoned for commercial development cannot realistically be downzoned to low density uses in most cases. Of necessity, we adopted a management approach (a stormwater performance standard) that did not directly limit land use, densities or imperviousness, and we structured it so that the standards were less strict for designated development nodes. Thus, the difficult optimization problem became a simple minimization problem: what are the minimum changes to development node boundaries necessary to meet the take limits established by the Etowah HCP scientific advisory committee?

This is not to imply, however, that it was an easy matter to achieve consensus among multiple jurisdictions on acceptable development node locations. Rather, our point is that managing stressors via local government regulation is subject to constraints that may limit the latitude for optimization relative to other more familiar forms of management.

Alternative modeling approaches

Our modeling was conducted within a general linear modeling (GLM) framework. In recent years, simple generalized linear models of species occurrence and abundance have been extended to address incomplete detection of species (MacKenzie et al. 2002, MacKenzie et al. 2006, Royle 2004, Royle and Nichols 2003), which can otherwise be a source of bias (Gu and Swihart 2004). These models are now becoming common in conservation applications (e.g., Ball et al. 2005). A typical characteristic of these types of models is an emphasis on hypothesis-driven model development and evaluation in an information-theoretic framework (Burnham and Anderson 2002). At the same time as these models have achieved common currency, a parallel field of research known as “niche modeling” or “species distribution modeling” (SDM) (Araujo and Guisan 2006, Guisan and Zimmermann 2000) has developed, with a focus on the application of innovative pattern-recognition modeling methods to species distribution predictions, often using presence-only data (Araujo and Guisan 2006, Elith et al. 2006). A recent review (Elith et al. 2006) compared 16 such methods, and the list was far from comprehensive. These approaches promise greater flexibility in dealing with nonlinear relationships, and have been promoted as providing superior predictive performance than traditional methods (Elith et al. 2006, Oakes et al. 2005, Olden and Jackson 2002). Although we opted for a GLM framework because we wished to account for incomplete detectability, the general approach we describe here would work equally well with any of the various species distribution models.

Many HCPs focus on the acquisition and protection of conservation areas; the Etowah HCP is somewhat unusual in its focus on avoidance and minimization of habitat degradation through land use regulation. Although significant portions of the Etowah are currently in permanent protection (Figure 6.5), the majority of the ranges of the federally listed species lie outside of these conservation lands on private property. Acquiring sufficient land to provide equivalent benefits to the HCP policy would cost billions of dollars, given current land prices, making the focus on land use regulation a necessity. It was this regulatory focus, and the need to guide and justify regulatory policies, that motivated the development of the approach described in this paper. We believe the approach is also transferable to cases where land acquisition and conservation are the central focus of species management efforts. Since such cases constitute more traditional reserve design problems, however, there are numerous alternative, analogous methods to arrive at appropriate management decisions (an excellent example is Drechsler et al. 2003) .

We initially presented this method as an alternative to be used when there are insufficient data to parameterize a PVA. In fact, both approaches could be combined. A PVA, should the data become available to perform one, could provide a sound basis for setting minimum levels of occupancy or abundance to be maintained, reducing the reliance on expert opinion (although expert opinion may still be required for setting the threshold for quasi-extinction). For species such as the Cherokee darter, which are widely distributed in semi-isolated patches, a metapopulation modeling approach (e.g., Schtckzelle and Baguette 2004) might be preferable to a classical PVA. A portion of the monitoring to be conducted as part of the Etowah HCP is intended to collect the sort of time-series data that can eventually be used for PVA parameterization.

CONCLUSIONS

Managing for imperiled species on private lands is a challenging proposition that demands a defensible basis for proposed regulatory policies. Based on our experience with the Etowah HCP, we believe the approach outlined in this paper can provide such a basis, even in cases where data are scarce. This is the first study we are aware of that uses a species predictive model to guide and predict the outcomes of land use regulatory policy. However, with continued growth in urban and suburban land uses, there will be an increasing need to manage stressors of species on private lands through regulatory policies. Deferring action until stressor-species relationships are precisely known will not be an option in many cases. Simple, defensible model-driven approaches that allow management action to proceed despite uncertainty will be essential for effective imperiled species conservation.

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APPENDIX 6A. TAKE LIMITS FOR THE AMBER DARTER AND CHEROKEE DARTER

The Amber Darter Take Limit requires that the projected occurrence probability of the amber darter meets the following criteria, as indicated by modeling using the most current version of the Etowah HCP Species Occurrence and Abundance Model:

a. At least 30% of stream miles in which probability of occurrence is greater than or equal to 80% under 2006 conditions must maintain a predicted probability of occurrence greater than or equal to 80% under the buildout scenario ("30/80/80 rule").
b. At least 50% of stream miles in which the 2006 probability of occurrence is greater than or equal to 80% must maintain a predicted probability of occurrence greater than or equal to 50% under the buildout scenario ("50/80/50 rule").

The preceding requirements apply to each of four designated populations areas that have known occupation (see Figure 6.9):

1. Upper Etowah R. mainstem and lower Amicalola
2. Middle Etowah R. mainstem and lower Long Swamp Cr.
3. Lower Etowah R, mainstem and lower Sharp Mt. Cr.
4. Lower Shoal Cr (Cherokee Co.).

The Cherokee darter take limit requires that the projected abundance of the Cherokee darter meets the following criteria, as indicated by modeling using the most current version of the Etowah HCP Species Occurrence and Abundance Model:

1. The total decline in Cherokee darter abundances under the buildout scenario, relative to 2006 conditions, must not exceed 30%.
2. At least 90% of stream miles in which estimated mean abundance is greater than or equal to 70 per 100m reach under 2006 conditions must maintain an estimated abundance of at least 33 per 100m reach under the buildout scenario.

The preceding requirements apply to occupied ranges of each of the three recognized evolutionarily significant units (ESUs) of the Cherokee darter (see Figure 6.10):

   a. Upper ESU: tributaries to the Etowah upstream of the confluence of Amicalola Creek and the Etowah River mainstem
   b. Middle ESU: tributaries of the Etowah upstream from the confluence of Stamp Creek and the Etowah River mainstem (in Lake Allatoona) and downstream from Amicalola Creek

Lower ESU: tributaries to Lake Allatoona and the Etowah mainstem downstream from and including Stamp Creek.

APPENDIX 6B. METHODS FOR CALCULATING EIA FOR ETOWAH HCP SCENARIO AND NO ACTION ALTERNATIVE SCENARIO

Our estimates of TIA at buildout (Table 6.3) were based on mean published values for commercial, industrial, institutional, and various residential land use categories (Capiella and Brown 2001). For conservation areas, we assumed that minor improvements such as access roads and entrance parking lots would result in a TIA of approximately 1.5%. For Priority 1 and 2 areas not in development nodes, we consulted with local planning officials, examined existing zoning maps, and considered actual developments constructed in the region to estimate a mean density of future growth. We estimated this to be 1-2.5 units per acre for most areas. Under the HCP scenario, we assumed that the strict runoff limits for Priority 1 areas would constrain densities to the lower bound of this estimate (1 unit per acre, or 11% TIA based on Capiella and Brown 2001), while in Priority 2 areas densities would average somewhat higher (2.5 units per acre, or 20% TIA extrapolated from Capiella and Brown 2001). For the no-action scenario, we assumed that without constraints of stormwater management, densities in both Priority 1 and Priority 2 areas would tend toward 2.5 units per acre.

Estimates of EIA were premised on the idea that stormwater management would effectively reduce the impact of TIA by removing contaminants and reducing hydrologic alteration. For the HCP scenario, we made the following assumptions:

- We assumed that 50% of the impacts of runoff to fish were from contaminants and 50% were from hydrologic alteration. This assumption reflects the state of science on the topic: both impacts are known to be significant (Wenger and Freeman 2006), and in the absence of data to suggest that one was more significant than the other, we assumed they were equal.

- For development nodes, we assumed that the hydrologic alteration portion of the impact was 0.50*TIA, based on the requirement of the Runoff Limits program that the runoff from nodes be reduced to the volume expected from half of the impervious area actually present. We assumed the hydrologic alteration impacts were equivalent to 5% TIA in Priority 2 areas and zero TIA in Priority 1 Areas and conservation areas, again based on the requirements of the Runoff Limits program.
- For the contaminant portion of impact, we assumed a mean 75% contaminant removal rate for development nodes, 85% for Priority 2 areas, and 90% for Priority 1 areas and conservation areas. A minimum of 68% contaminant removal rate is required under the stormwater management policy (Wenger et al. in preparation). However, the ordinance also effectively requires that at least a portion of runoff be managed by infiltration, which has contaminant removal rates approaching 100% (Barraud et al. 1999, Davis et al. 2003, Walsh et al. 2004). The range of contaminant removal values reflects the increasing proportion of infiltration expected in development nodes, Priority 2 areas and Priority 1 areas, respectively.

As an example, take a cell in a Priority 2 area that is not in a node or conservation area. Based on the assumptions described previously, this cell has a TIA of 20%. Under the runoff limits program, hydrologic alteration impacts are estimated to be equivalent to 5% TIA, and contaminant impacts are estimated to be reduced by 85% to 3% TIA. The final EIA is the mean of these two values, or 4% (Table 6.3).

For the no-action scenario, we assumed that without the runoff limits management policy of the HCP, stormwater management would be less effective in reducing both contaminant and hydrologic alteration impacts, and thus EIA values would be higher. We did assume that stormwater regulations typical for the region (Metropolitan North Georgia Water Planning District 2004) would be applied across all jurisdictions. These regulations call for mean contaminant removal rate of 68%, and the use of extended stormwater detention to manage hydrologic alteration. The benefit of extended detention in reducing hydrologic alteration is unknown; in fact, this uncertainty was an impetus in the development of the runoff limits policy (Wenger et al. in preparation). However, in order to estimate EIA under the no action alternative, we were forced to make the assumption that the reduction in hydrologic alteration was greater than zero but less than 50% (the assumed reduction in hydrologic alteration attributed to the standard for development nodes), and therefore adopted 25% as an estimate. We retained the assumption that impacts of hydrologic alteration and contaminants were equivalent. All of these assumptions were combined to convert the TIA values to EIA values for the no action alternative (Table 6.3).

APPENDIX 6C. PREDICTIVE MODELING RESULTS FOR AMBER DARTER AND CHEROKEE DARTER

The amber darter is predicted to have moderate declines in occupied habitat under the Etowah HCP scenario, and much greater losses under the no action scenario (Table C1). Greater losses are predicted in the Lower Etowah and Shoal Creek population areas under the HCP scenario (Table 6.7, Figures 6.11 and 6.12), while under the no action scenario loss is high throughout the range, but lowest in the Lower Etowah.

The Cherokee darter is predicted to decline moderately in abundance under the HCP and more significantly under the no action scenario (Table 6.8, Figures 6.13 and 6.14). Declines are predicted to be greater for the lower and middle ESUs than for the upper ESU.

Table 6.1. Fish species covered under the Etowah HCP. Status refers to federal (Fed.) or state (GA) listing as endangered (E) or threatened (T). Federally listed species are the subjects of this study.

Scientific Name	Common Name	Family	Status
Macrhybopsis sp. cf. *aestivalis*[1]	Coosa chub	Cyprinidae	GA E
Noturus sp. cf. *munitus*[1]	Coosa madtom	Ictaluridae	GA E
Percina antesella (Williams and Etnier)	amber darter	Percidae	Fed. E / GA E
Percina lenticula (Richards and Knapp)	freckled darter	Percidae	GA E
Percina sp. cf. *macrocephala*[1]	bridled darter	Percidae	GA E
Etheostoma etowahae (Wood and Mayden)	Etowah darter	Percidae	Fed. E / GA E
Etheostoma scotti (Bauer, Etnier and Burkhead)	Cherokee darter	Percidae	Fed. T / GA E
Etheostoma sp. cf. *brevirostrum* A[1]	holiday darter	Percidae	GA E
Etheostoma sp. cf. *brevirostrum* B[1]	holiday darter	Percidae	GA E

[1] Undescribed species assumed most closely related to *Macrhybopsis aestivalis, Noturus munitus, Percina macrocephala, and Etheostoma brevirostrum*, respectively.

Table 6.2. Covariates of the predictive models used for the three species. "Modifying" indicates which model response variable is modified by the covariate. "Direction" indicates whether the covariate is positively (+) or negatively (-) associated with the response variable. Asterisk (*) indicates that the effect is positive or negative, depending on the collector identity.

Species	Covariate	Modifying	Direction
E. etowahae	Watershed area	Occupancy	+
	Dlink (downstream link magnitude)	Occupancy	+
	$Dlink^2$	Occupancy	-
	Mean slope of tributary system	Occupancy	+
	% of tributary system inundated by impoundments	Occupancy	-
	% EIA within 1.5km radius	Occupancy	-
E. scotti	Micaceous saprolite	Abundance	+
	% EIA within 1 km radius	Abundance	-
	Elevation	Occupancy	-
	Gneiss bedrock	Occupancy	-
	Metagraywacke bedrock	Occupancy	+
	In Little River tributary system	Occupancy	-
	Collector	Detection	*
P. antesella	Watershed area	Occupancy	+
	% EIA within 1.5km radius	Occupancy	-

Table 6.3. Estimated total and effective impervious area (TIA and EIA, respectively) for cells in different categories under the HCP scenario ("HCP") and no-action alternative ("No Act"). Values for TIA are based on Capiella and Brown (2001). See Appendix B for assumptions underlying EIA values.

Category	TIA- HCP	EIA- HCP	TIA- No Act	EIA- No Act
Node: Commercial	72.2%	27.1%	72.2%	33.6%
Node: Industrial	53.4%	20.0%	53.4%	24.8%
Node: Multifamily	44.4%	16.7%	44.4%	20.6%
Node: ¼ acre lots	27.8%	10.4%	27.8%	12.9%
Node: Institutional	34.4%	12.9%	34.4%	15.6%
Conservation Area	1.5%	0.1%	1.5%	0.7%
Priority 1 (~1 unit/acre)	11%	0.6%	20%	9.3%
Priority 2 (~2.5 units/acre)	20%	4.0%	20%	9.3%

Table 6.4. Predicted length of habitat occupied by Etowah darters (in km) in eight watersheds under 2006 conditions, the HCP scenario and the no-action scenario. Proportional losses in habitat from 2006 conditions are also shown for the HCP scenario and no action scenario.

Watershed/area	2006 Habitat	Remaining Habitat: Etowah HCP	Loss: HCP	Remaining Habitat: No Action	Loss: No Action
Amicalola Cr.	61.1	52.2	15%	1.8	74%
Etowah Headwaters	59.2	49.8	16%	1.8	88%
Etowah Mainstem-Shoal Cr.	27.4	16.1	41%	0.8	88%
Long Swamp Creek	39.5	26.4	33%	1.2	78%
Raccoon Cr.	32.1	30.1	6%	1.0	92%
Etowah Middle Mainstem	9.6	3.3	65%	0.3	87%
Smithwick Cr.	1.4	0.3	75%	0.0	97%
Stamp Cr.	1.7	0.7	59%	0.1	96%
Total	**231.9**	**178.9**	**23%**	**36.9**	**84%**

Table 6.5. Parameter estimates and standard errors for Etowah darter model. The uncertainty in model predictions of occupancy is proportional to these errors.

Parameter	Estimate	Standard Error
Intercept	-4.63946	3.141905
Watershed area	2.840578	1.534279
Dlink (downstream link magnitude)	4.382269	2.180286
$Dlink^2$	-4.4792	2.424152
Slope of tributary system	1.472488	1.105672
% of tributary system inundated by impoundments	-1.32945	0.879284
% EIA within 1.5km radius	-6.86954	6.15484

Table 6.6. Results of sensitivity analysis for EIA predictions for the Etowah darter model. Adjustment in EIA represents a test that EIA values for each modeled stream reach are 10% or 25% greater or lower than predicted based on the assumptions for the HCP scenario. Remaining habitat means the estimated length of occupied Etowah darter habitat at buildout under the HCP scenario, after accounting for adjustment in EIA. Decline from 2006 indicates the proportional loss of habitat from 2006 conditions (see Table 6.4). The final column indicates whether the scenario meets the take limits, after accounting for adjustment in EIA.

Adjustment in EIA	Remaining Habitat	Decline from 2006	Meets Take Limits?
- 25%	193.08	17%	Yes
- 10%	184.11	21%	Yes
0	178.86	23%	Yes
+ 10%	173.61	25%	Yes
+ 25%	166.32	28%	No

Table 6.7. Predicted length of habitat occupied by amber darters (in km) in four population areas under current conditions, the Etowah HCP and the No Action Alternative. Proportional losses in habitat are also shown.

Population Area	Current Habitat	Remaining Habitat-Etowah HCP	Loss - HCP	Remaining Habitat-No Action	Loss - No Action
Upper Etowah	23	22	4%	7	68%
Middle Etowah	14	14	1%	5	62%
Lower Etowah	27	21	20%	12	53%
Shoal Cr.	1	1	29%	0	100%
Total	**64**	**57**	**11%**	**25**	**61%**

Table 6.8. Predicted populations (in 1000s) of the three ESUs of Cherokee darters under current conditions, the Etowah HCP and the No Action Alternative. Proportional losses are also shown.

ESU	Current Abundance	Abundance- Etowah HCP	Loss - HCP	Abundance- No Action	Loss - No Action
Lower	321	252	22%	184	43%
Middle	313	244	22%	172	45%
Upper	44	40	9%	28	37%
Total	**678**	**536**	**21%**	**384**	**43%**

Figure 6.1. The Etowah Basin, showing tributary systems and watersheds used to clip stream reaches.

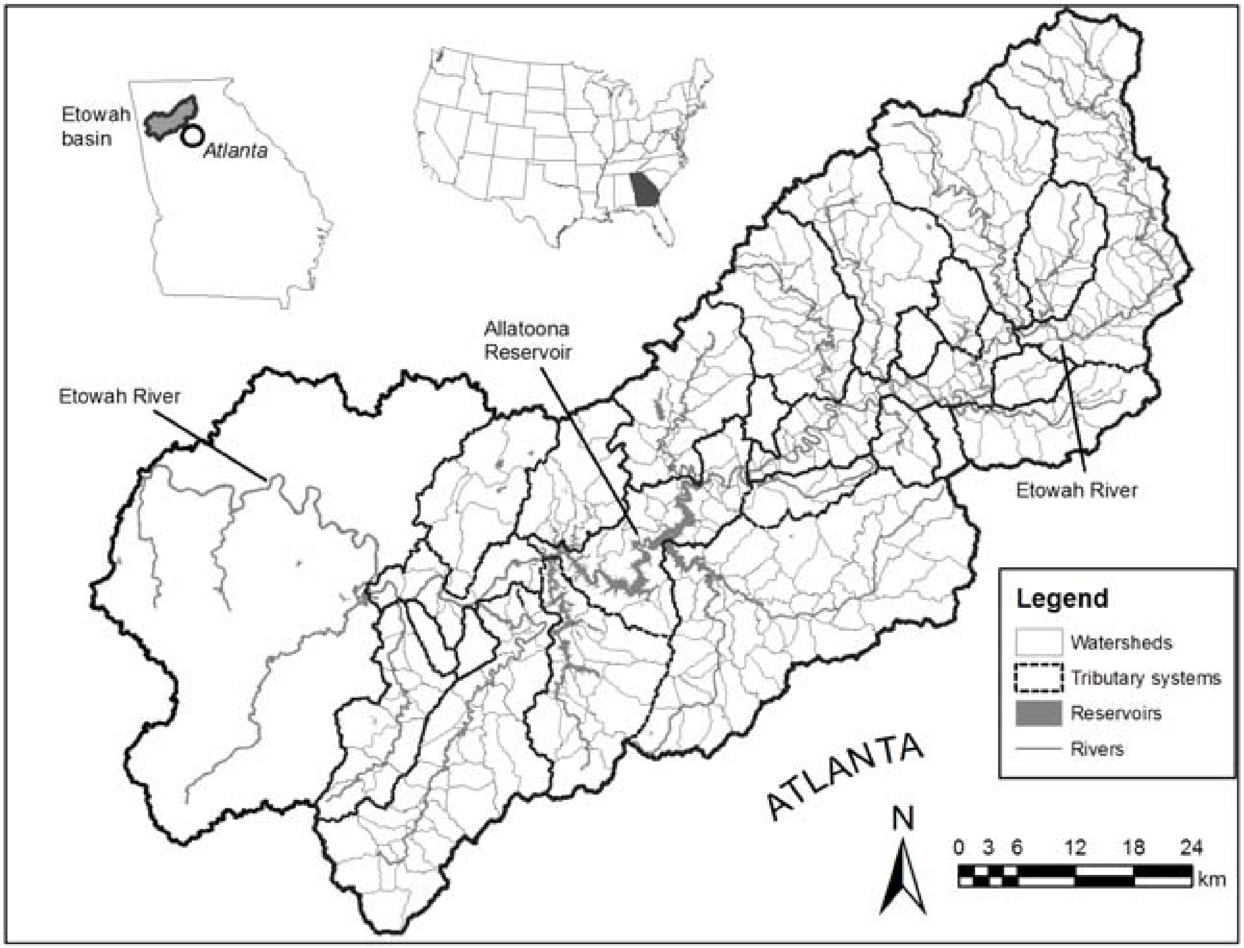


Figure 6.2. Priority Areas and known occurrences of threatened and endangered fish species in the Etowah basin.

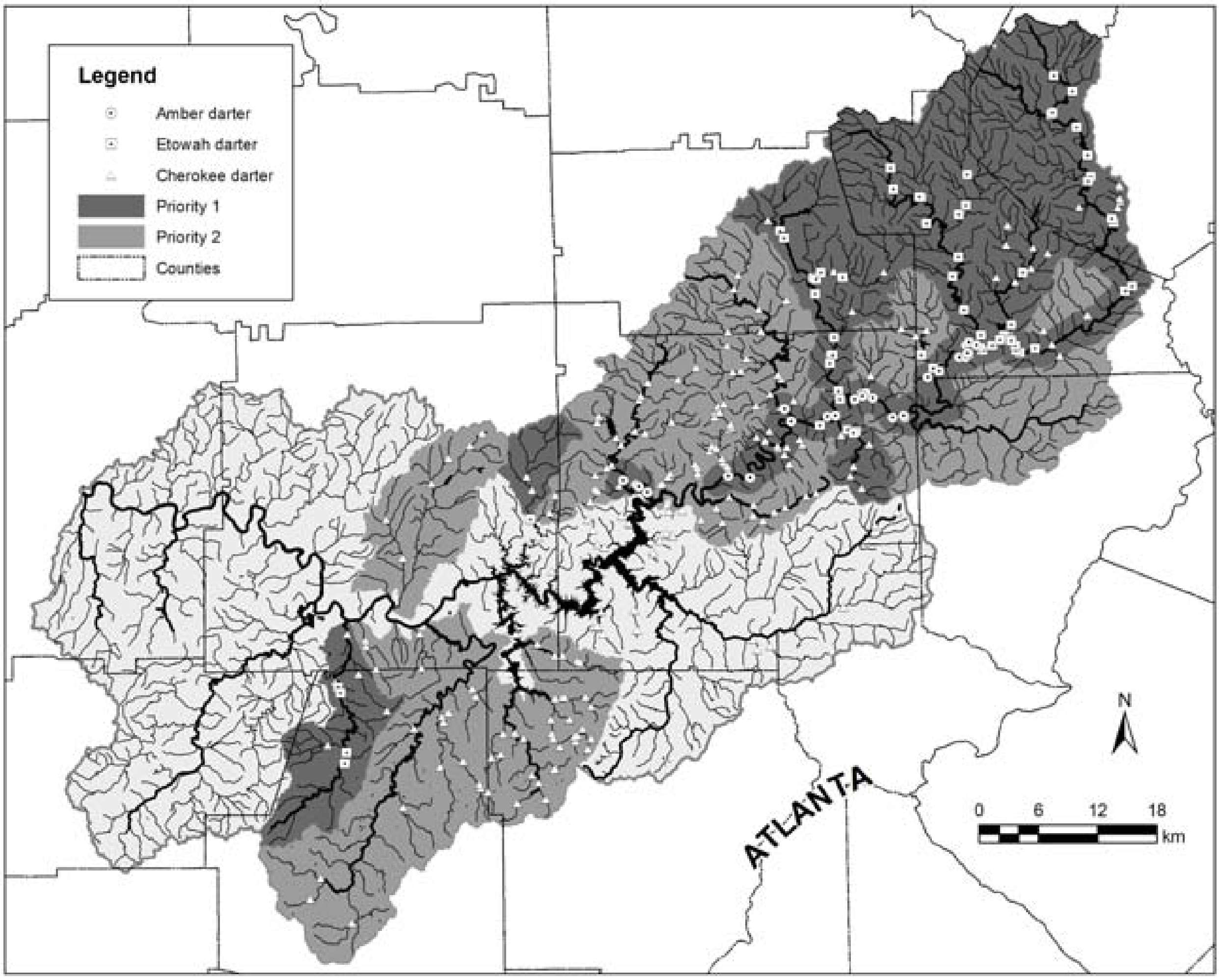


Figure 6.3. Population areas for Etowah darters under the take limits. Population areas are shown in heavy black or grey, with bold labels. Other streams with Etowah darter collections (not considered major population areas) are labeled in normal font.

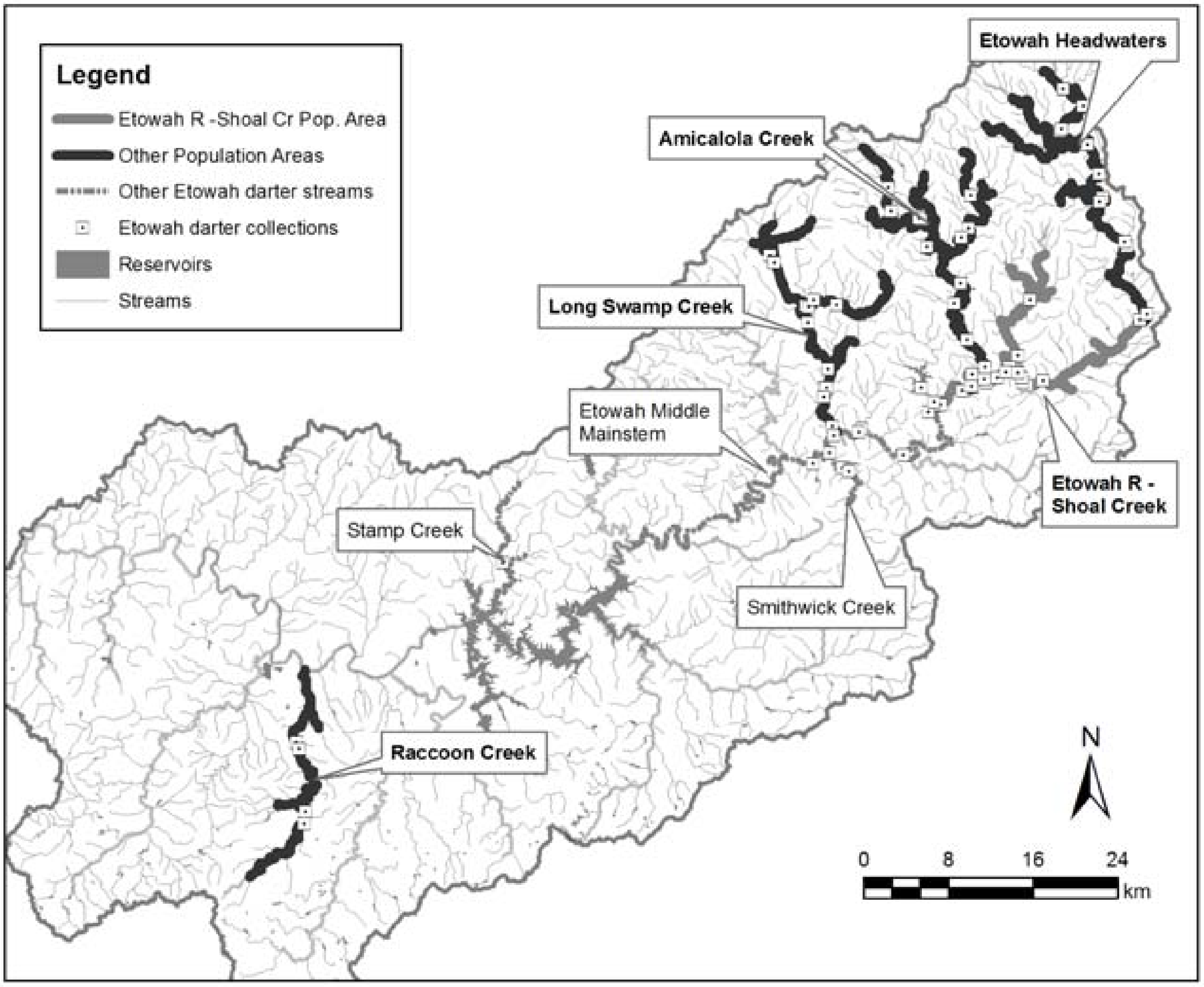


Figure 6.4. Predicted length of habitat occupied by Etowah darters in eight watersheds under current conditions, the HCP scenario and the no-action scenario. The first five watersheds are the five population areas covered by the take limits.

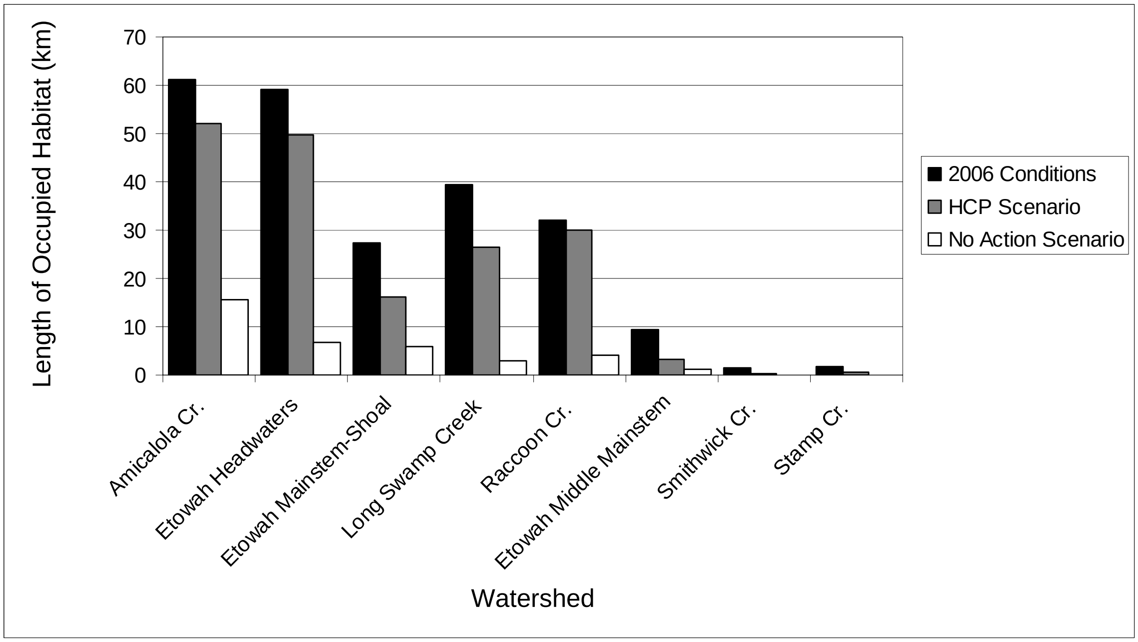


Figure 6.5. Modeled Etowah darter occupancy (occurrence probability) under 2006 conditions. Color-coded stream labels indicate mean of the posterior occurrence probability for each reach. Results only shown for streams with known occupation.

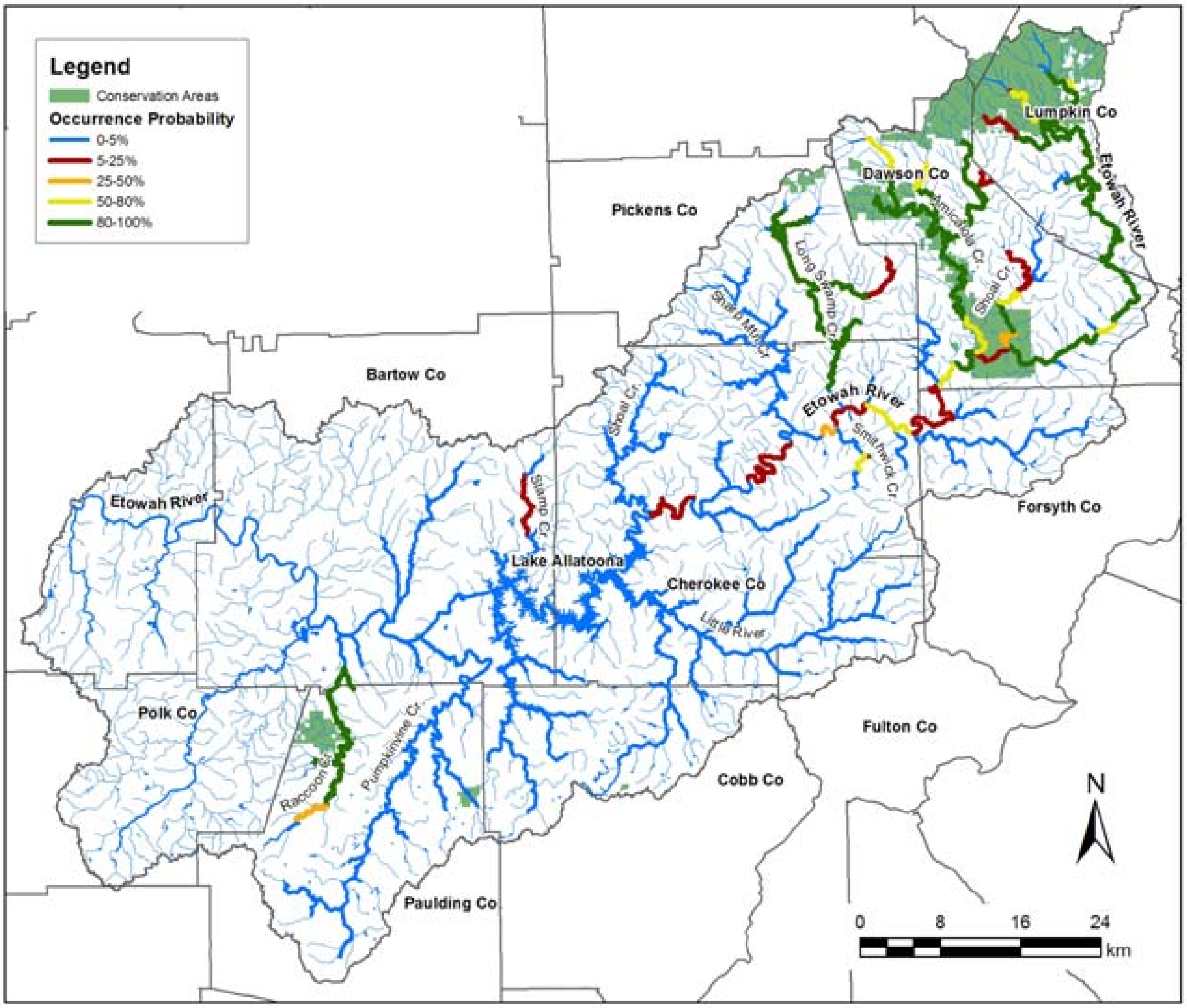


Figure 6.6. Modeled Etowah darter occupancy (occurrence probability) under the HCP scenario. Color-coded stream labels indicate mean of the posterior occurrence probability for each reach. Inset shows full posterior distribution of occurrence probability for an individual reach. Results only shown for streams with known occupation.

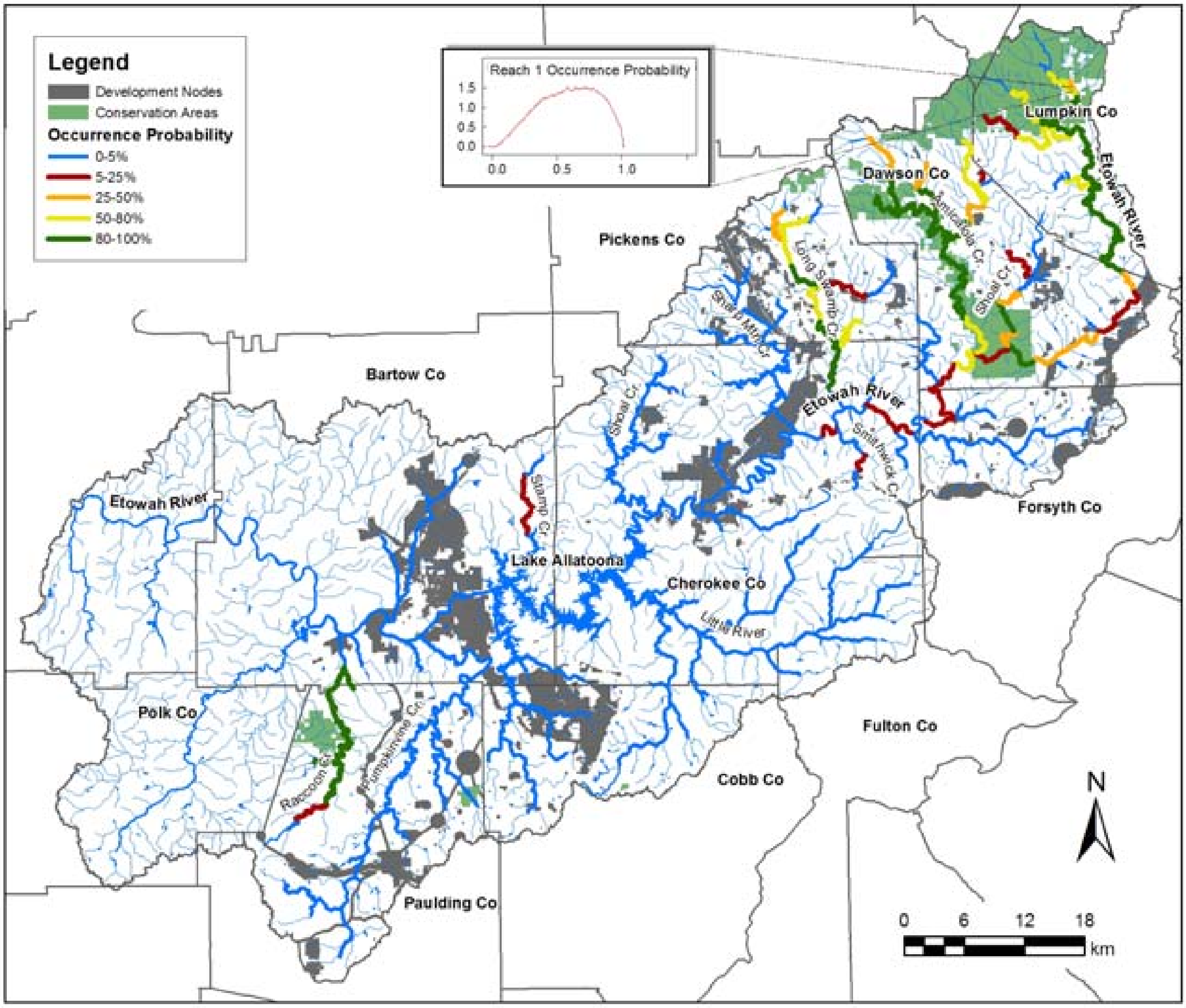


Figure 6.7. Predicted length of habitat occupied by amber darters in four population areas under current conditions, the HCP scenario and the no-action scenario.

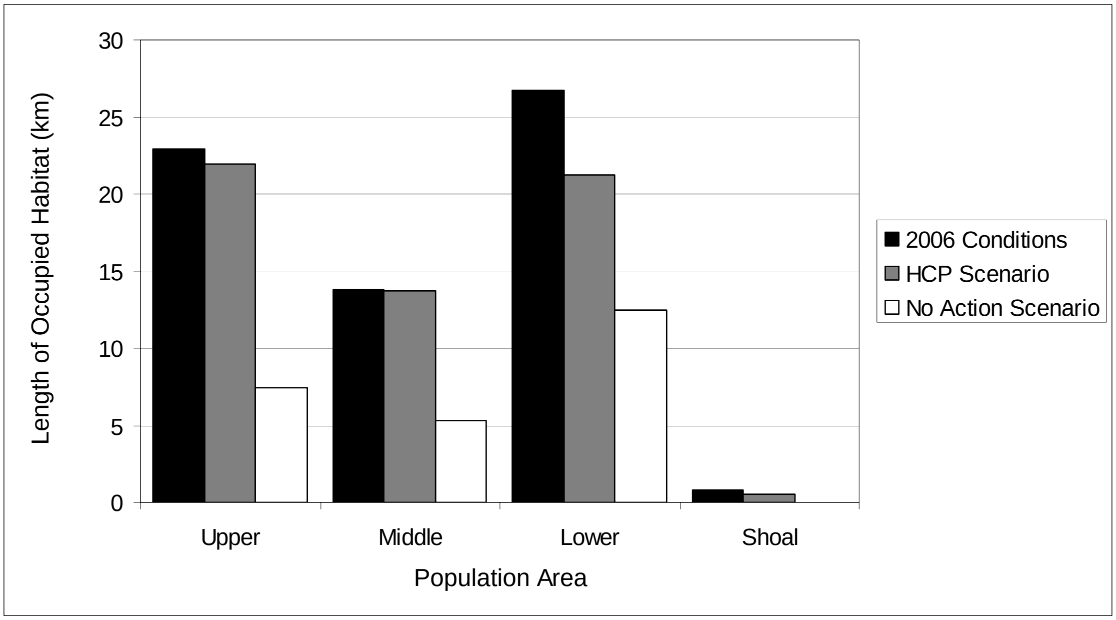


Figure 6.8. Predicted populations of the three ESUs of Cherokee darters under current conditions, the HCP scenario and the no-action scenario.

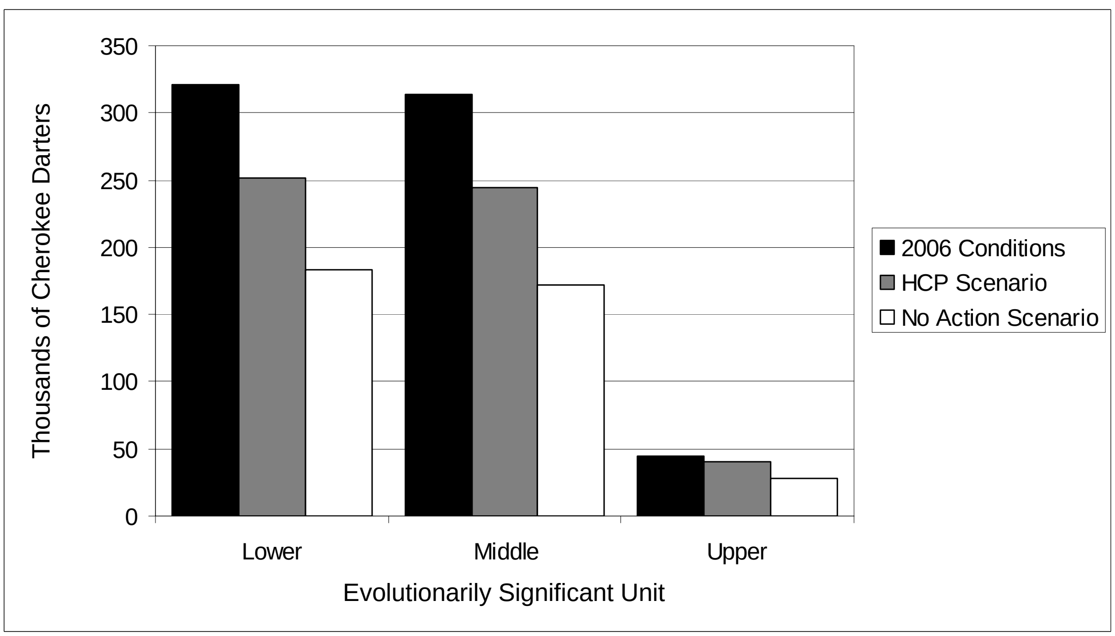


Figure 6.9. Amber darter population areas.

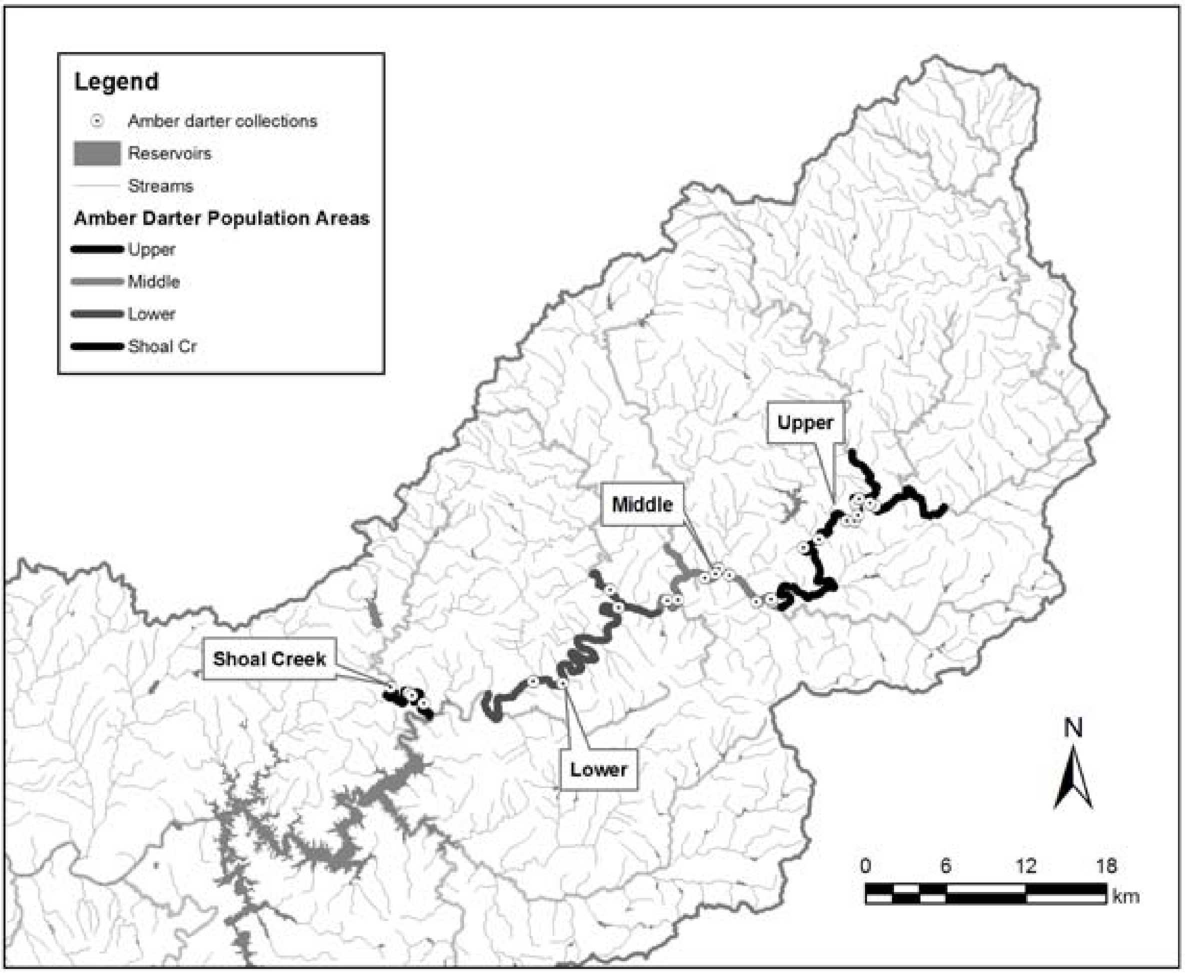


Figure 6.10. Cherokee darter ESUs.

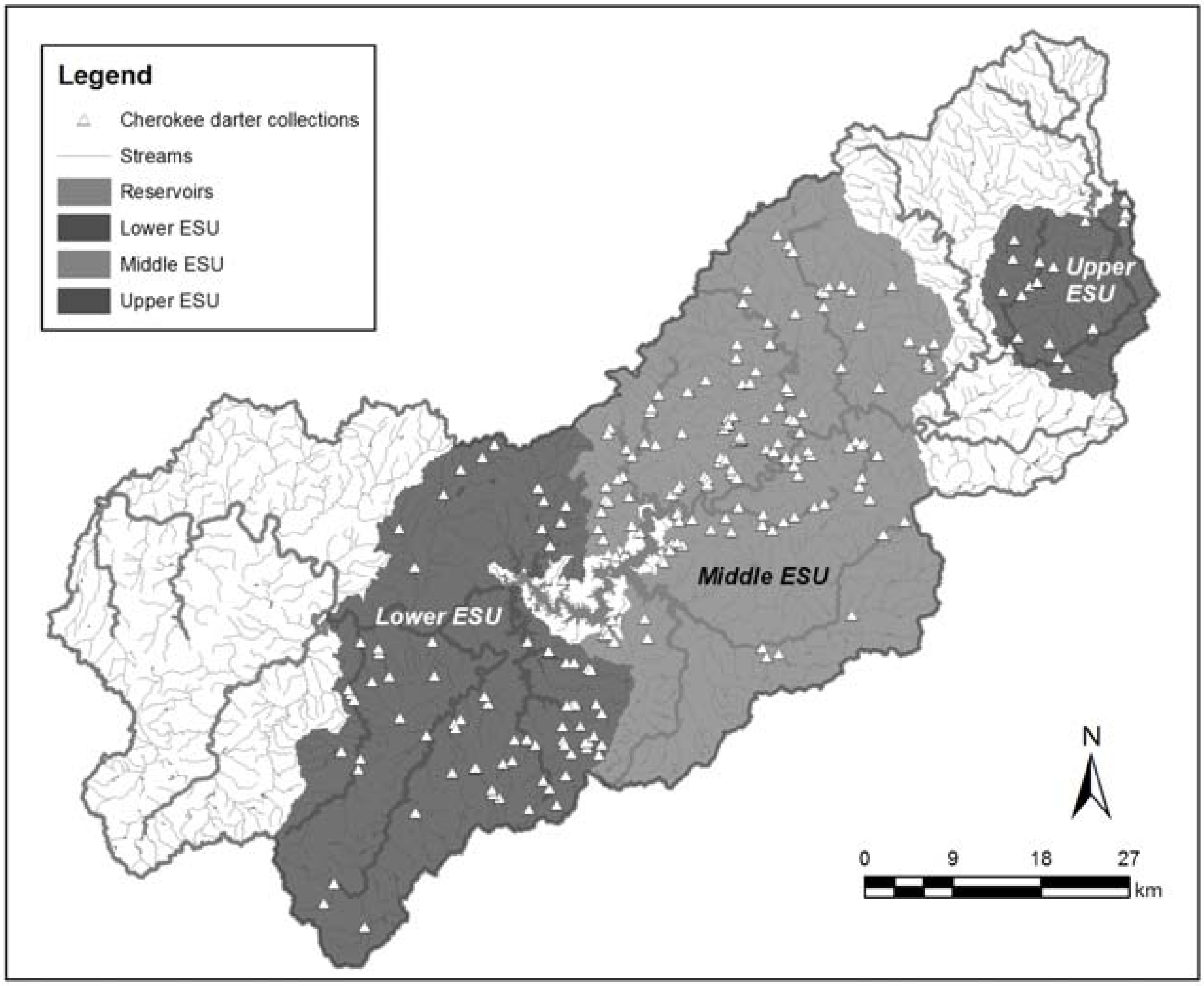


Figure 6.11. Modeled amber darter occupancy (occurrence probability) under 2006 conditions. Color-coded stream labels indicate mean of the posterior occurrence probability for each reach. Results only shown for streams with known occupation.

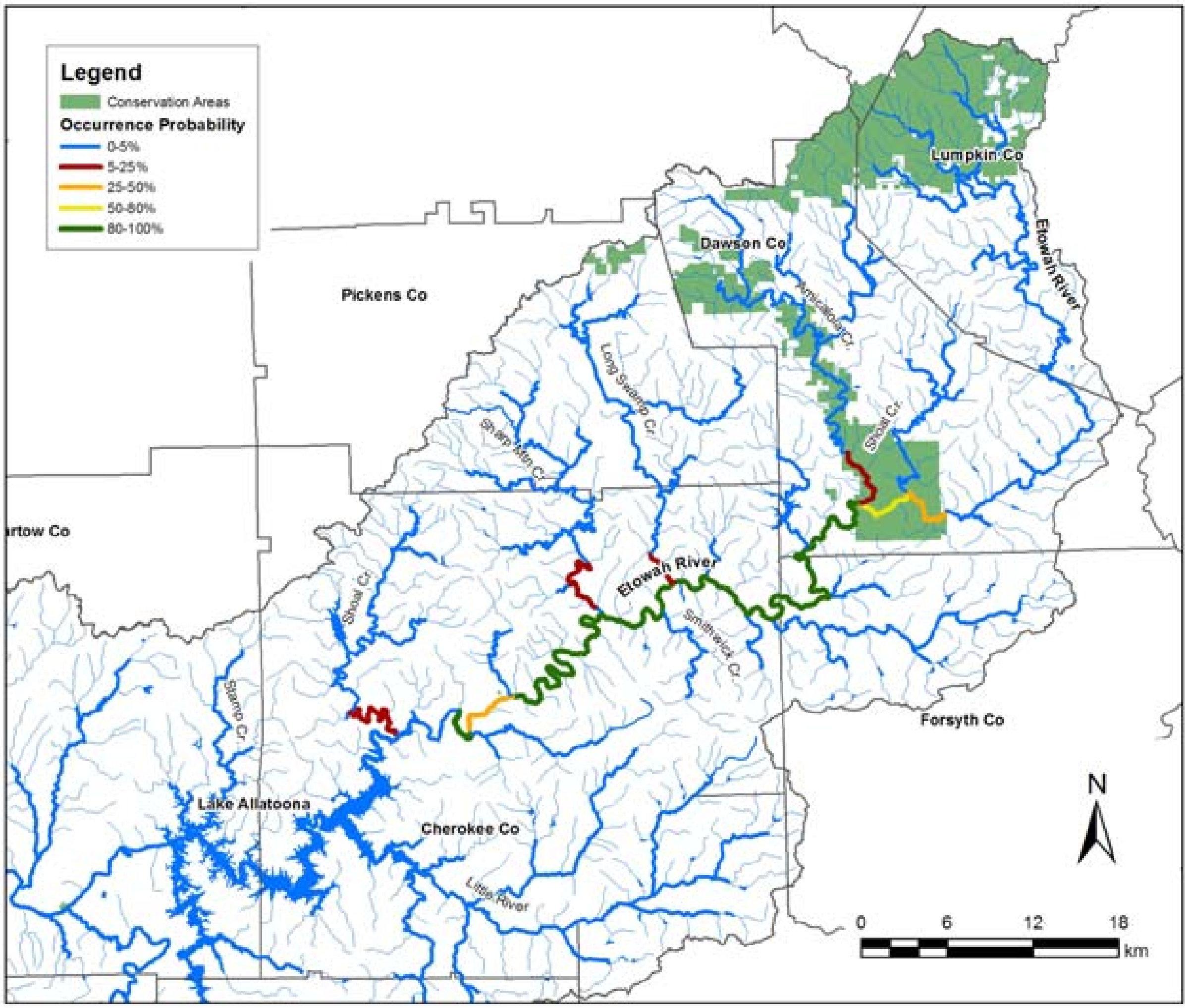


Figure 6.12. Modeled amber darter occupancy (occurrence probability) under the HCP scenario. Color-coded stream labels indicate mean of the posterior occurrence probability for each reach. Results only shown for streams with known occupation.

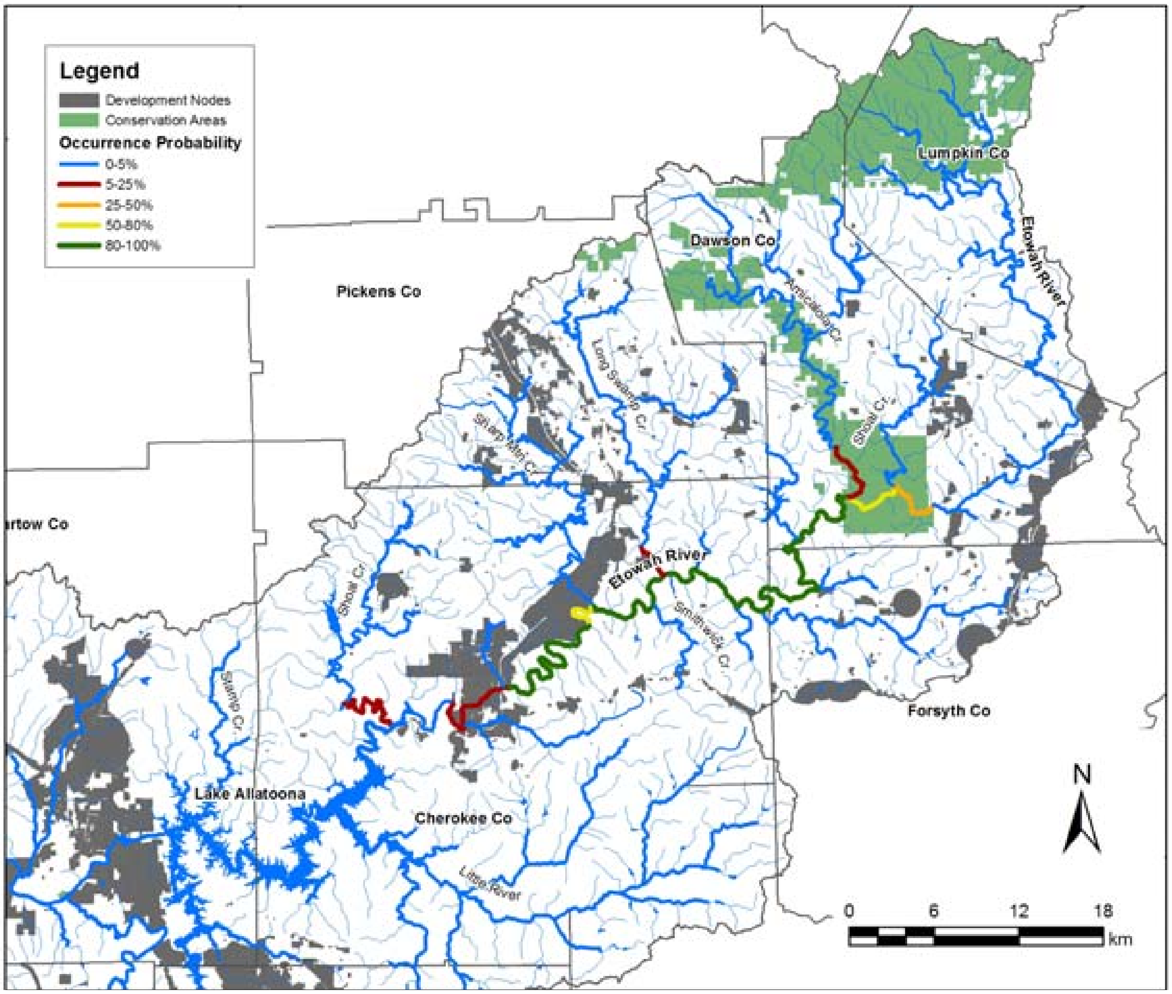


Figure 6.13. Modeled Cherokee darter abundance under 2006 conditions.

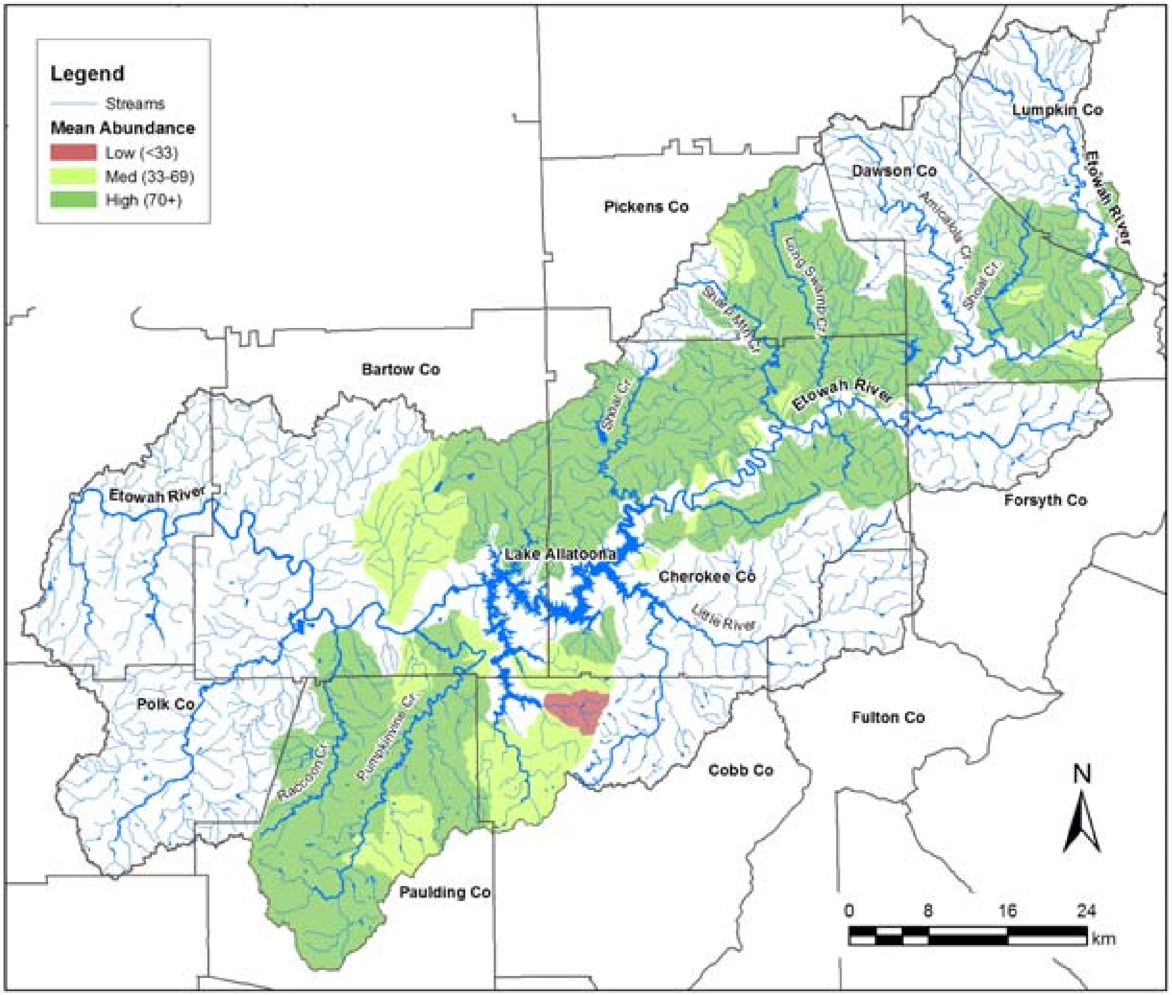
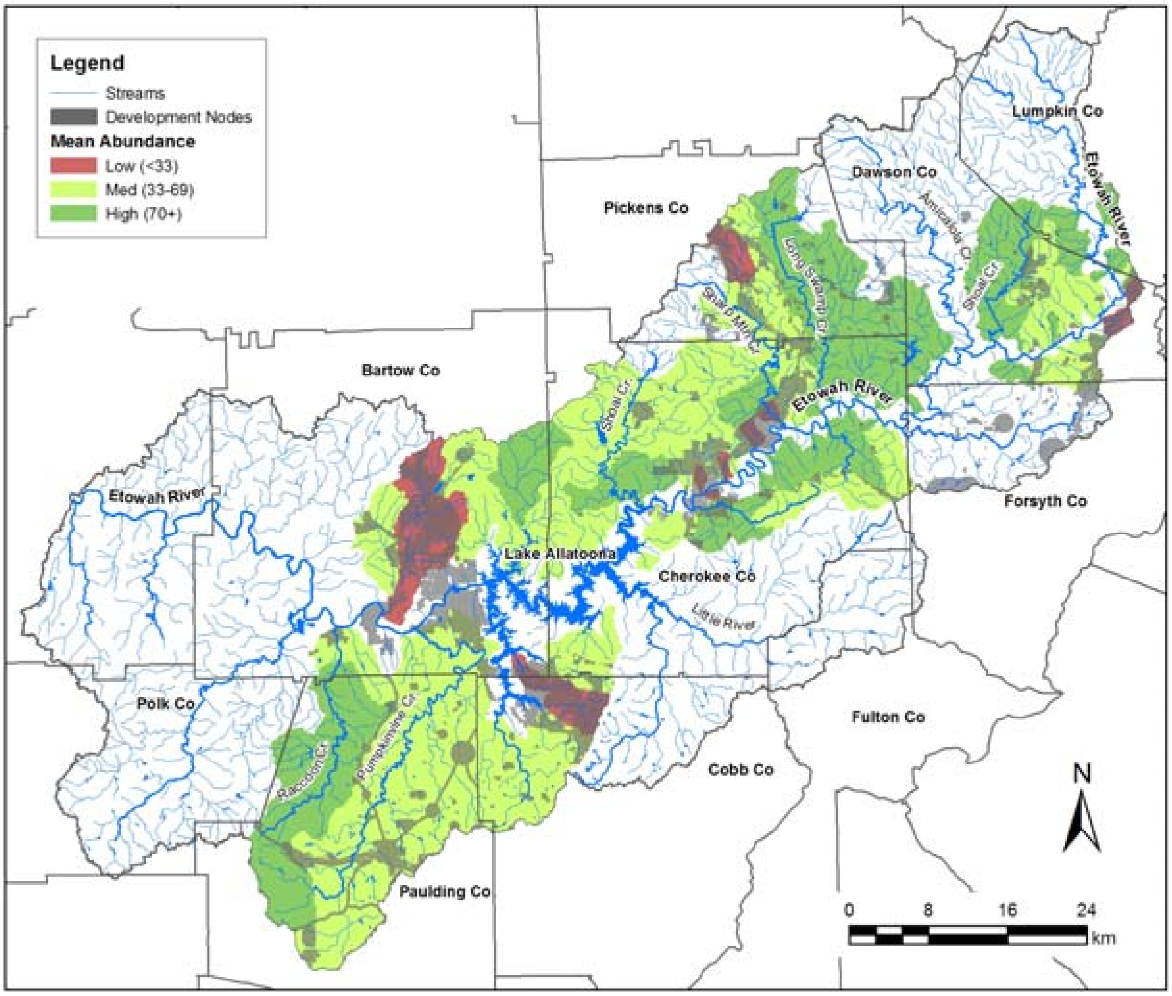


Figure 6.14. Modeled Cherokee darter abundance under the HCP scenario.



Chapter 7

Conclusions

As of this writing, we are finalizing the Etowah Habitat Conservation Plan (HCP) document for submission to the U.S. Fish and Wildlife Service (FWS). Staff of FWS will review the proposed plan to determine whether it provides adequate avoidance, minimization and mitigation measures, and to ensure that it does not jeopardize the survival of the species. This review process might take up to a year, according to FWS officials (Aaron Valenta, US FWS, pers. com.). At the end of the process the FWS will recommend changes to the plan, which will be considered by the Etowah HCP Steering Committee. Our hope is that such changes will be minimal, considering the lengthy stakeholder participation process that was required to achieve the fragile consensus on the draft provisions. Assuming that FWS and the Steering Committee come to agreement on the final form of the plan, local governments who wish to participate will implement Etowah HCP policies and receive an incidental take permit.

Much of Chapter 2 of this dissertation is included in the Etowah HCP, almost verbatim, as the justification for the management policies included in the plan. The full contents of Chapter 2 will also be published on the HCP website (www.etowahhcp.org) as a stand-alone report. The runoff limits program, which is the topic of Chapter 5, is incorporated into the plan as part of the stormwater management policy. Limited material from Chapters 3 and 4 appears in much reduced form in the background for the stormwater management policy and in the "Take

Statement," which explains the methods and results for calculating take of the federally listed species. The bulk of the Take Statement, which will be attached to the HCP, is substantially reworked from the contents of Chapter 6. In the final version of the Etowah HCP, the published or to-be-published versions of all of these chapters will be referenced.

Assuming that the HCP is implemented, the participating jurisdictions will support programs to monitor compliance with plan requirements and the status of the HCP-targeted fish species, as described briefly in Chapter 6. Responsibility for reviewing proposed changes to development nodes will fall to an HCP implementation organization, a small oversight body created by agreement of the participating counties and municipalities. We will turn over the HCP predictive model to this organization for use in testing whether proposed node changes meet the take limits. With the assistance of Dave Homans, the steps required to run the model have been scripted into a utility that is relatively simple to use, so that specialized GIS and statistical expertise are not required for operation.

While we have high hopes for the approval and successful implementation of the Etowah HCP, it remains possible that local governments could ultimately reject the agreement. Even if this happened, most of this work would remain relevant to management and conservation. Georgia staff of the FWS already regard the proposed policies of the HCP as representing the best available science, which means that in their review of individual projects they make the recommendation (which is effectively a requirement in most cases) that these policies be implemented on an individual site basis. The forecasting model of the Etowah HCP could be used by FWS as a guide to whether proposed development projects constituted jeopardy of listed species.

We also think the general approach we describe in Chapter 6 can be of use in other HCPs. It is likely that other conservation scientists face a similar situation to that of the Etowah: one or more species threatened by rapid land use change, but with limited data to predict the outcome of that change on species populations. Our approach provides a rational basis for decision making that maximizes use of limited data. We think this method could be profitably adapted to many other species of conservation interest.

However, I wouldn't want to oversell the contribution of this dissertation to the success of the Etowah HCP. If the project works, it will be primarily due to the relationships initially built by Laurie Fowler with local officials and other key stakeholders. Curt Gervich, Outreach Coordinator for the Etowah HCP from 2003 to 2006, worked full time within the basin to extend these relationships into a large network of support for the plan. The success of a large conservation project with considerable economic implications ultimately rests on individuals, who must feel personally invested and committed. The biggest lesson from the Etowah HCP that I would pass on to other such projects is to invest in an outreach program that provides meaningful involvement of stakeholders and decision makers. While the resources invested in the science influence the quality of an HCP, the resources invested in outreach and public involvement ultimately determine whether the project results in failure or success.