THE EFFECTS OF GLOBAL CLIMATE CHANGE ON THE FISHES OF THE SOUTHEASTERN UNITED STATES

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Introduction

The southeastern United States includes twelve states that stretch from the eastern border of Texas to the Atlantic Coast. This region ranges southward to Florida and northward to Kentucky and West Virginia (figure 1). This region is characterized by warm summers and mild winters. Average January temperatures generally exceed 5° C, and average July temperatures generally range from 25-30° C (Mulholland et al. 1997). Precipitation, which mostly occurs as rainfall, varies from 1000 mm per year in the northwestern extremes of the region to 1400-1600 mm per year on the Gulf Coast and in southeastern Florida (Poff 1992). This region is geologically diverse: despite its small size, it contains three of the six major physiographic provinces of the United States. These three provinces are divided into eight subunits: the Gulf-Atlantic Coastal Plain, Piedmont, Blue Ridge, Newer Appalachians, Appalachian Plateau, Interior Lowlands, Ozark Plateau, and the Ouachitas (deBlij and Muller 1998). The abundant rainfall and diverse geography of this area interact to form a wide variety of aquatic habitats. This variety of habitats and the long growing season of the Southeast have given rise to the most diverse fish community in North America (Warren et al. 2000). For example, a square quadrat (150 miles on a side) containing eastern Tennessee and western Virginia contains 226 fish species, whereas the entire Great Lakes region (which is nearly 20 times larger) contains only 176 species (Ross-Flanagan 2003).

The southeastern United States is home to numerous economically important fisheries.

The high-gradient streams in the southern Appalachian Mountains support populations of popular game fishes such as brook trout (*Salvelinus fontinalis*) and smallmouth bass (*Micropterus dolomieui*) (Wallace et al. 1992). Lower-gradient streams , lakes, and ponds contain sport fish such as chain pickerel (*Esox niger*) and multiple species of sunfishes (such as largemouth bass, *Micropterus salmoides*; bluegill sunfish, *Lepomis macrochirus*; and white crappie, *Pomoxis annularis*) (Felley 1992; Smock and Gilinsky 1992). These fishes attract a large number of anglers. Spending by anglers (resident and out-of-state visitors) in each southeastern U. S. state ranged from 102 million (West Virginia) to 1.6 billion (Florida) dollars in 2001. Freshwater recreational fishing contributed an average of 564 million dollars to each state's economy in this region (Table 1) (USDOI et al. 2001). In addition to these valuable game species, the medium-sized rivers of the Atlantic Coast also contain shad and herring (*Alosa spp.*) populations. Shad and herring existed in sufficient numbers to support a small commercial fishery in the late seventies and early eighties (Garman and Nielsen 1992).

Channel catfish (*Ictalurus punctatus*) are also native to this region (Fuller, Nico et al. 1999). Though wild and stocked channel catfish are considered game fish, they are much more economically important as a commercially raised species. Channel catfish culture occurs primarily on southeastern U.S. fish farms, and most of the fish raised are sold as food (NASC 2005). In 2004, producers from the Southeast sold approximately 315,000 tons of catfish to processing plants: the total value of this product was 451 million dollars (NASC 2005). Channel catfish will be discussed in more detail in a following section.

Given that commercial and recreational capture and the large-scale culture of fish for food and sport (Avault 1996; Banner 2000) produce significant amounts of revenue in this region, the fisheries of the Southeast are certainly worth protecting. However, the freshwater aquatic habitats of the Southeast have been degraded by impacts associated with land use such as urbanization, mining, forestry, water management, and agriculture. The effects of these activities have been the subject of many studies (e.g. Nagel 1991; Ross et al. 2001). However, few studies have examined the possible impacts of climate change on the fishery resources of the Southeast.

In recent years, atmospheric concentrations of heat-trapping gas have been increasing, causing some concern as they can affect the global climate (IPCC 2001). Surprisingly, the southeastern United States has shown a cooling trend over the past 50 years. This is thought to be a result of increased cloudiness, which reduces daily temperature fluctuations and the daily temperature maximum (Mulholland et al. 1997). Although more cloud cover could partially offset climate change in this area, the fact remains that heat-trapping gas concentrations continue to increase. As the concentrations of these “greenhouse gases” continue to rise, they will eventually overwhelm the protective effects of the cloud cover. This is why climate models still predict a warmer, drier climate in the southeastern United States in the future (Ward et al. 1992). Furthermore, the increase in cloud cover is probably a cyclic event. It is possible that cloud cover will decrease again over time (Mulholland et al. 1997), which would exacerbate the warming trend in this region.

Climate change is not just restricted to temperature; weather patterns are also expected to change. Although predictions concerning precipitation changes in the southeastern United States vary, many of them point to an increase in precipitation with the exception of the northernmost portion of the region (Mulholland et al. 1997).

The flow regimes of large river systems in the southeastern United States depend in part on the behaviors of their headwaters, which for many rivers are located outside the boundaries of this region. Current trends show a decrease in snowpack and ice cover. In the high latitudes of the Northern Hemisphere, snowpack has decreased by approximately 10% since the late 1960's (Albritton and Meira Filho 2001). These changes in flow regimes and thermal regimes will impact the majority of aquatic ecosystems, including those that support freshwater fisheries.

The purpose of this review is to address the question of how much the freshwater fisheries of the southeastern United States will be affected by global climate change.

Brook trout populations in the Appalachians

Since fish are cold-blooded, every aspect of their physiology is controlled by temperature. Their metabolism, or the amount of energy they need to survive and grow, is also temperature-dependent. Every species of fish is adapted to live within a certain thermal range.

Temperatures at the high end of this range cause metabolic inefficiency: in effect, fish living at these higher temperatures simply cannot eat enough to meet their energy demands. On the other hand, temperatures at the lower end of this range cause decreased activity and appetite (which decreases growth and weight gain). This makes it difficult for a fish to build a large enough energy reserve to survive the winter (when food supplies are severely limited).

Brook trout (*Salvelinus fontinalis*) have a narrow temperature tolerance range and cannot live in warm water. Even sublethal increases in summer temperature would cause an increased demand for food (Shuter and Meisner 1992).

Ries and Perry (1995) estimated the increase in food demand for Appalachian brook trout living at higher temperatures. At temperatures that were 2° C higher than the current average, they would need 15-20% more prey, and at temperatures that were 4° C higher than average, they would need 30-40% more prey (Ries and Perry 1995). Since trout are often food-limited in the summer months (Morgan et al. 1999), climate change will probably cause decreases in trout populations. Furthermore, cold water fishes such as trout often need low overwinter temperatures in order to spawn successfully (Langford 1983; Gerdaux 1998). To the author’s knowledge, this has not been experimentally tested for Appalachian brook trout, but it is possible that increased temperatures could cause further population decreases by hindering or preventing successful reproduction.

The effects of altered stream flows could also impact brook trout populations via two mechanisms. First, increasing groundwater temperatures may decrease refuge availability and quality. Coldwater fishes such as brook trout rely on cool groundwater discharge for a refuge from high summer water temperatures (Meisner et al. 1988), especially in lower elevation streams and streams near the southern edge of their range (Meisner 1990). The availability of these cold-water refugia will be decreased as groundwater temperatures are expected to increase with an increase in mean global temperatures (groundwater temperatures closely approximate average annual temperatures in temperate zones) (Meisner et al. 1988). Some authors argue that an average temperature increase of about 4°C could dramatically reduce the range of brook trout in the southeastern United States (Flebbe 1993; Mulholland et al. 1997).

Second, a change in stream flow may greatly affect local populations. Aquifers, or groundwater reservoirs, discharge water into streams and keep them flowing through the dry season. Since geologic features limit aquifer size in this region, and rainfall is expected to occur in more extreme, less frequent events (Mulholland et al. 1997), small to medium-sized streams may no longer flow continuously throughout the year (Ward et al. 1992; Poff et al. 1996). The Blue Ridge and Appalachian provinces already contain streams with highly variable flows. The effects of altered precipitation patterns on individual streams of these two provinces would depend upon presence of springs and local aquifers respectively. However, an overall increase in intermittent (discontinuous in time) flow has been predicted (Ward et al. 1992).

Channel catfish

The channel catfish (*Ictalurus punctatus*) can tolerate a wide variety of water temperatures (0-35° C), but its optimal temperature range spans only a few degrees (21-27° C) (Stickney 2000). Though channel catfish grow fastest at 28° C, they digest and utilize their food most efficiently at 24-26° C (Buentello et al. 2000). This implies some loss of metabolic efficiency at higher temperatures. Channel catfish require several weeks of water temperatures below 15°C for successful spawning (Boyd and Tucker 1998). Therefore, these fish are also vulnerable to climate change. Because of the warm summer temperatures in the Southeast, a dramatic shift in the native range of this fish is possible: McCauley and Beitinger (McCauley and Beitinger 1992) suggested that a 2° C increase in mean annual temperature would shift the range of this fish 250 km northward. Since channel catfish , in their native range, can only be found as far south as northern Mexico (McCauley and Beitinger 1992), the northward retreat of the species from this area has some serious implications for the catfish farming industry.

Climate change will alter water chemistry and quality, which could also reduce production of catfish, especially those without benefit of pond aeration systems. When fish “breathe in water” they are actually using the dissolved oxygen gas trapped in the water. The dissolved oxygen concentration in water is temperature-dependent: as water temperature increases, oxygen levels decrease (Goldman and Horne 1983). Water temperature is the most important predictor of dissolved oxygen concentrations in shallow ponds (Hargreaves and

Tucker 2003). Changes in dissolved oxygen can reduce productivity of catfish. Catfish are usually grown in ponds, and the management of waste from uneaten feed, fecal matter, and fish metabolites is largely left to nature. Most of these natural decomposition processes require dissolved oxygen. Therefore, the maximum stocking capacity of a catfish pond receiving feed and fertilizer is determined by available dissolved oxygen. Since current stocking densities represent the maximum possible production in these systems (Hargreaves and Tucker 2003), a drop in dissolved oxygen levels could lower the productive capacity of catfish ponds. Current aquaculture practices employ aeration at night to maintain ponds in a super-saturated state. However wild catfish populations lack such technological support. This may be a significant issue for the subsitance catfish fishery. While there are no readily available numbers on this group, the catfish fishery is substantial. Figures for Alabama alone, estimate 230,000 catfish anglers (USDOI, USFWS, USDOC, and U.S. C. Bureau 2001).

Production rates in aquaculture facilities could be decreased due to an increase in disease outbreaks. The immune function of fish is compromised in stressful situations, including those that involve crowding and high temperatures. Therefore, culture systems that raise channel catfish are inherently disease-prone (Plumb 2001). It is quite possible that climate change could worsen this situation by affecting rates and probabilities of disease and parasite outbreaks, especially in crowded catfish ponds. Increasing water temperatures will make for a more stressful environment for fish in general which will put them in a more vulnerable state for diseasen transmission and outbreaks.

On one hand, higher temperatures associated with milder winters may lower infection rates by decreasing the stress experienced by fishes surviving a long winter. For example, "winterkill" in channel catfish results from a rapid temperature drop and a subsequent long period where water temperatures do not exceed 10°C (Bly et al. 1993). Diseases such as this would become less common with shorter and/or milder winters. On the other hand, warmer water temperatures may increase virulence of certain diseases or create additional stresses that increase the probability of infection. For example, rates of bacterial disease in aquaculture systems often peak at high temperature (Hefer and Pruginin 1981; Wedemeyer 1996). This may also be attributable to low dissolved oxygen levels, which have been shown to slow recovery from bacterial diseases in channel catfish (Mqolomba and Plumb 1992). Warmer winters may also allow for increased survival of bacteria and parasites, outweighing any advantages of such a seasonal change. There are no concrete answers to questions about climate change and disease transmission. However shifts in when diseases occur, year-round prevalence and introduction of new pathogens due to range expansions all pose additional challenges to fish in this region.

Striped bass and sunfishes

Most of the warmwater game fishes in this region, such as bluegill sunfish (*Lepomis macrochirus*) and largemouth bass (*Micropterus salmoides)*, tolerate a wide range of temperatures. They may be more affected by the secondary effects of climate change such as water quality, stream flow, and water level changes than by a small temperature increase.

Physical changes in lake systems may negatively affect reservoir dwellers such as the prized striped bass (*Morone saxitilis*) (Soballe et al. 1992). In most cases, the sun lacks sufficient energy to heat all of the water in a lake. Instead, the majority of the sun's heat only penetrates into the first few meters of the water column. Because warmer water is less dense than colder water (down to a temperature of 4° C), the water in the upper "layer" of the lake does not mix with that in the lower layer. This results in the formation of a stratified or "two-story" (Moyle and Cech 1988) lake. Because the temperatures in the two layers (the epilimnion on top and the hypolimnion on bottom) are quite different, many fish cannot occupy both of these habitats. Coolwater and coldwater fish such as striped bass and trout seek refuge in the cooler hypolimnion, and warmwater fish such as largemouth bass occupy the epilimnion. Since there is no mixing between the two layers of the lake, the hypolimnion is basically a closed compartment with a limited supply of oxygen. Most temperate zone lakes mix or "destratify" once or twice a year (Goldman and Horne 1983), and this mixing of the two layers replenishes the oxygen supply in the hypolimnion. Global climate change will probably decrease the frequency of this mixing by increasing the strength and duration of stratification (Topping and Bond 1988; Gaedke et al. 1998).

Longer periods of stratification would force fishes living in the hypolimnion to survive for longer periods at low oxygen levels. Increased solar heating will thicken the top layer of warm water (the epilimnion), making the hypolimnion smaller. In addition, since higher temperatures will increase fish metabolism and oxygen usage, the available oxygen in the hypolimnion will be depleted sooner. Therefore, fishes that are confined to this thermal compartment face a "temperature-oxygen squeeze" (Matthews et al. 1985) that severely limits their available spring and summer habitat (Christie and Regier 1988; Gerdaux 1998). Crowding in the smaller hypolimnion would also subject fish to low prey availability, stress, and the probability of increased disease transmission (Cheek et al. 1985; Coutant 1985). Since juvenile striped bass are more tolerant of high temperatures than large adults, current climatic conditions only threaten the trophy fishery for this species (Coutant 1985). Even though smaller striped bass still provide a recreational fishery in lakes and ponds, climate change could eliminate wild striped bass populations from the Gulf Coastal region, which represents the southern margin of their native range (Coutant 1985; Moss 1985). The probability of this occurrence will increase if groundwater temperatures increase. Like brook trout, striped bass rely upon groundwater flow as a cold-water refuge in warmer months (Cheek et al. 1985; Coutant 1985). Managers have begun to stock hybrid sunshine bass (white bass x striped bass) in an attempt to sustain this fishery (Moss 1985). Despite resilience of these hybrids to higher temperatures (hybrids have a lower oxygen consumption rate than striped bass at 30°C), they experience similar stress responses when exposed to hypoxic (low-oxygen) conditions (Mitchell and Cech 1994).

Changes in timing and quantity of precipitation will affect water levels in ponds and some lakes in the region, and the nature of annual floods in stream systems. Small changes in water levels of ponds will likely have minimal impacts on freshwater fishes inhabiting the open-water zone of these systems. Instead, they will have more serious consequences for species with fairly specific depth ranges, such as fishes that spawn in the shallow, vegetated zone near the shoreline. For example, an increase pond water levels before, during, and shortly after the annual spawning season produces high spawning success and high survival of larval largemouth bass (Keith 1975).

This phenomenon is not limited to lakes and ponds. Late winter and spring floods that increase water level and submerge seasonal floodplains of the Cache River, Arkansas contribute to increased abundance of larval darters (Percidae), minnows (Cyprinidae), and sunfish (Centrarchidae) (Killgore and Baker 1996). Obviously, precipitation is a major driving factor in aquatic systems; changes in annual rain and snowfall totals can drastically affect aquatic ecosystems. In most major river systems worldwide, the size of the seasonal flood is determined by precipitation.

Changes in temperate rivers will be driven by changes in precipitation in the form of rain and snow. Reduced snowpacks will decrease spring flows in large systems such as the Mississippi (Nijssen et al. 2001). Without high spring flows, temperate stream systems may experience lower minimum flows (Nijssen et al. 2001) and a decrease in the magnitude of the flood pulse. The medium-sized rivers of the Atlantic and Gulf Coastal Plains and the blackwater rivers of this region are pulse-dominated systems, and many of their native fishes such as sunfish and pickerel migrate into the submerged floodplain to forage and spawn (Welcomme 1979; Felley 1992; Smock and Gilinsky 1992; Harper and Mavuti 1996). During the seasonal flood, debris such as leaf litter and insects from the surrounding areas falls into the stream. This input is a vital addition to the food supply in these systems (Felley 1992; Mulholland and Lenat 1992; Smock and Gilinsky 1992).

Finally, like the coldwater streams that support brook trout, warmwater streams in the Southeast could cease flowing for part of the year. Again, there is a risk that altered weather patterns may cause intermittent or discontinuous flow in small to medium-sized streams (Ward et al. 1992; Poff et al. 1996).

Native communities and nonnative invaders

Because fish are cold-blooded, their range can be determined by temperatures as well as by physical barriers to migration. Tropical species will likely experience a northward expansion as increasing global temperatures shift these thermal barriers (Mulholland et al. 1997). Resource managers should be concerned because a small increase in temperature could allow further invasions by tropical species into the southeastern United States. This has particularly important implications for the southernmost states in this region, especially Florida.

Because of its warm climate and mild winters, introduced tropical species survive and sometimes thrive in Florida (Austin 1993). Many tropical fishes such as tilapia species (*Tilapia spp.* and *Sarotherodon spp.*), peacock bass (*Cichla ocellaris*), and the jaguar guapote (*Cichlasoma managuense*) have established reproducing populations in peninsular Florida (Shafland 1996; Courtenay 1997; Fuller et al. 1999). However, they are confined to this area because it is surrounded on three sides by ocean and they are unable to tolerate the low temperatures in the more northern portions of the state (Austin 1993). For example, the lower lethal temperature limit for two predatory fishes, the jaguar guapote (*Cichlasoma managuense*) and the butterfly peacock (*Cichla ocellaris*) is 12° C and 15° C respectively (Shafland 1996). There is some evidence that introduced fishes negatively affect native species. For example, introduced tilapias also utilize the vegetated shoreline habitats in lakes and ponds. They therefore compete with native sunfishes for food and spawning habitat (Crisman 1992; Shafland 1996; Courtenay 1997), and prey upon sunfish eggs and juveniles (Crisman 1992; Fuller et al. 1999). The spotted tilapia (*Tilapia mariae*) and the Rio Grande cichlid (*Cichlasoma cyanoguttatum*) are two such species, and both have been introduced to Florida. Temperature tolerance tests indicate that they are capable of inhabiting waters throughout Florida (Siemien and Stauffer 1989). Although the Rio Grande cichlid, with its increased cold tolerance, can be found as far north as the southern edge of Lake Ponchatrain, Louisiana, spotted tilapia populations do not occur north of Polk and Hillsborough counties in Florida (Commission 2003).

An increase in mean annual temperatures could allow an expansion of the realized niches of these species by increasing the low temperatures which limit their ability to compete and survive in novel environments. Some species of tilapia also eat the aquatic vegetation upon which native species lay their eggs (Courtenay 1997). Other larger introduced fishes such as the jaguar guapote have a highly predatory nature and could negatively affect native sunfish communities. Upon its introduction into the Rio Usumacinta basin in Mexico, this fish began to prey heavily upon the native species (Shafland 1996). Species that are too small to affect native gamefish populations can still have detrimental effects. For example, the introduced pike killifish (*Belonesox belizanus*) prefers to prey upon small native species such as the eastern mosquitofish (*Gambusia holbrooki*) (Courtenay 1997), which is a voracious predator of mosquito larvae. Heavy predation on this beneficial species could lead to increased insect pests.

On the other hand, some of these introduced species have been beneficial to some degree. The introduced blue tilapia is abundant enough to support a small commercial fishery (Courtenay 1997). Peacock bass support a recreational fishery in the urban canals on southern Florida. This fishery is worth an estimated 15.5 million dollars annually (Shafland and Stanford 1999). Despite all of their potential benefits and/or drawbacks, few studies have been conducted to determine the ecological effects of these myriad introduced species (Courtenay 1997). However, though there is no documented case of an introduced fish species causing the extinction of any of Florida's native species, this is not the case in other locations (Courtenay

1997). One of the few cases in which pre and post-introduction data were obtained involved the stocking of peacock bass into Lake Gatun, Panama. The expanding peacock bass population has been implicated in the virtual elimination (reduction in numbers caught by 90% or more) of 6 of the 8 species commonly found in the lake (families Characinidae, Cichlidae, Elotridae, and Poeciliidae) (Zaret and Paine 1973). A study of several drainages in the northern Appalachians found that successful invasion of aquatic communities by nonnative fish species is facilitated by human modification of water or habitat quality (Ross et al. 2001). Climate change certainly falls into this category, and its alteration of native fish habitat to favor invading exotic species cannot be ignored.

Increasing temperatures could also contribute to the reproductive success and range expansions of nonnative nuisance plant species. Two of the most notorious invasive plants in the Southeast are water hyacinth (*Eichhornia crassipes*) and hydrilla (*Hydrilla verticillata*). Water hyacinth can be found throughout the Southeast, but it is most prevalent in Florida, Alabama, and Louisiana. The state of Louisiana treats about 200,000 acres of water annually to reduce water hyacinth populations, but nearly 1,000,000 acres remain untreated due to lack of funding (Saunders and Johnson 2004). Although it is established in isolated drainages in southern Virginia, North and South Carolina, and southern Arkansas, most of the populations of this plant are nonpermanent in these states (Jacono and Richerson 2003). Hydrilla has also invaded most of the Southeast, but it is most prevalent in the southernmost states in this region.

Approximately 70% of Florida's river basins contain hydrilla: it is the most common aquatic plant in the state (Jacono and Richerson 2003). This aquatic weed is also problematic in the Atchafalaya Basin in Louisiana, the Mobile Delta in Alabama, and in parts of southwest Georgia (Jacono and Richerson 2003).

Both of these plants are tropical species intolerant of cool temperatures. Before intensive management programs were implemented in Florida, winter temperatures were the only factor impeding their range expansion (Joyce 1992). Water hyacinth production ceases below 10°C (Schmitz, et al. 1991).

Although it can be found throughout much of the United States, climate change could greatly increase the success of the hydrilla strain that currently plagues the Southeast. In a 1986 laboratory study, germination of hydrilla tubers was 3% at 15° C (Steward and Van 1987). This indicates that low winter temperatures currently suppress the propagation and spread of this aquatic pest.

The ecological effects of these plants are profound. Both plants form dense mats of vegetation, thereby shading the water column; this suppresses native vegetation and inhibits photosynthesis in the water column (Schmitz et al. 1997). In addition, large floating rafts of water hyacinth can physically damage native plant beds (Schmitz et al. 1997). These plants are also capable of enormous production of plant detritus – one hectare of water hyacinth can produce between 404 and 1186 metric tons of plant detritus on an annual basis (Schmitz et al.

1991). As a result, hydrilla and water hyacinth can have a profound effect on water chemistry. Dense mats of these plants can cause a drop in dissolved oxygen levels, a decrease in pH, and a decrease in bicarbonate alkalinity (Reddy et al. 1983). These factors are very important in shaping fish communities, so an increase in the success of these plants is likely to have a detrimental effect on the native fishes of the southeastern U. S.

Sufficient climatic warming could allow invasive aquatic species that currently have limited distributions in this area to invade the remainder of the southeastern United States, and greatly expand their ranges. At best, this expansion would cause an increase in the cost of control programs, and, at worst, it could cause multiple extinctions.

Socioeconomic effects and summary

Changes in fishery productivity will in turn affect the human populations and economies that are reliant on those resources. Fisheries contributes significantly to the economy of the southeastern United States. Including satellite businesses (such as feed producers and processing plants), channel catfish culture in the United States produces 300,000 full-time jobs and 8 billion dollars in revenue annually (Avault 1996). Because most channel catfish culture occurs in the Southeast, most of the employment opportunities and revenues associated with this industry are centered around this region. In the Southeast, culture of trout (for food and for recreational and conservation purposes) occurs in the southern Appalachians. North Carolina is the largest producer of cultured trout in the southeast U. S.. This state is second only to Idaho in its production volume: total sales of trout in North Carolina in 2004 reached 5.9 million dollars (NASC 2005). Trout culture in North Carolina is vulnerable to climate change: production is already limited by warm summer temperatures (Banner 2000). Trout production in Georgia, Tennessee, Virginia, and West Virginia ranged from 54,000 to 400,000 pounds in 2004; revenue produced from sales of these fish ranged from $181,000 to $727,000 (NASC 2005). Although these numbers are likely much higher as they do not include the U.S. Fish and Wildlife Service raised “mitigation” fish which are used for stocking and for which the agency receives no compenstaion. Since high summer temperatures currently limit production of both catfish and trout, global warming would decrease production rates in both culture systems.

The freshwater recreational fisheries of this region also produce an impressive amount of revenue. North Carolina and Florida are both in the top five states in the United States in terms of out-of-state angler visitation, with Florida receiving almost 4 million anglers and North Carolina receiving about 1.6 million anglers in 1996 (Ditton et al. 2002). Therefore, loss of productivity in or collapse of these recreational fisheries poses a threat to the regional economy of the Southeast. A 2°C rise in temperature on the North American continent may result in a major northward shift in the ranges of economically important fish species. As a result, the southeastern United States. may lose a large portion of its recreationally important fishes.

Furthermore, the increased success of tropical aquatic weeds will further affect the southeastern United States.   Not only will these nuisance species continue to affect economically important fisheries, but their increased productivity and distribution may require increasingly expensive control programs. The state of Florida spent 13.2 million dollars on hydrilla control from 2000-2001 (Management 2001). If winter temperatures become too warm to limit production of invasive plants, the already high cost of aquatic weed control could skyrocket.

Given that the effects of climate change will likely affect fish physiology, water chemistry and quality, and flow regimes, fish populations will probably undergo changes that are difficult to predict on the population or species level. Consequently, fisheries of the southeastern United States could potentially suffer as a result of global climate change. In light of these possibilities, we should make efforts to better manage our heat-trapping gas emissions in order to avoid lessening the productivity of an important part of the regional economy of the Southeast.

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Literature Cited

Albritton, D. L., and L. G. Meira Filho. 2001. Technical Summary: a report accepted by Working Group I of the IPCC but not approved in detail.

Austin, D. F. 1993. Displacement of native ecosystems by invasive alien plants—the Florida experience, or how to destroy an ecosystem. Symposium on Invading Species - Their Threat and Potential. Florida Exotic Pest Plant Council, Miami, Fla.

Avault, J. W. J. 1996. Fundamentals of Aquaculture: a Step-by-Step Guide to Commercial Aquaculture. AVA Publishing Company Inc., Baton Rouge, La.

Banner, K. 2000. From the mountains to the sea: aquaculture in North Carolina. Aquaculture Magazine March/April 2000:36-46.
Bly, J. E., L. A. Lawson, A. J. Szalai, and L. W. Clem. 1993. Environmental factors affecting outbreaks of winter saprolegniosis in channel catfish, *Ictalurus punctatus* (Rafinesque). Journal of Fish Diseases 16:541-549.
Boyd, C., and C. Tucker. 1998. Pond Aquaculture Water Quality Management. Kluwer Academic Publishers, Norwell, Mass.
Buentello, J. A., D. M. I. Gatlin, and W. H. Neill. 2000. Effects of water temperature and dissolved oxygen on daily feed consumption, feed utilization, and growth of channel catfish (*Ictalurus punctatus*). Aquaculture 182:339-352.
Cheek, T. E., M. J. Van Den Avyle, and C. C. Coutant. 1985. Influences of water quality on distribution of striped bass in a Tennessee River impoundment. Transactions of the American Fisheries Society 114:67-76.
Christie, G. C., and H. A. Regier. 1988. Measures of optimal thermal habitat and their relationship to yields for four commercial fish species. Canadian Journal of Fisheries and Aquatic Science 45:301-314.
Commission, G. S. M. F. 2003. Nonnative aquatic species summaries. Ocean Springs, MS.
Courtenay, W. J. J. 1997. Nonindigenous fishes. Pages 109-122 *in* D. Simberloff, D. C. Schmitz, and T. C. Brown, editors. Strangers in Paradise: Impact and Management of Nonidigenous Species in Florida. Island Press, Washington, D. C.
Coutant, C. C. 1985. Striped bass, temperature, and dissolved oxygen: a speculative hypothesis for environmental risk. Transactions of the American Fisheries Society 114:31-61.
Crisman, T. L. 1992. Natural lakes of the southeastern United States: origin, structure, and function. Pages 475-538 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.
deBlij, H. J., and P. O. Muller. 1998. Physical Geography of the Global Environment. John Wiley and Sons, New York.
Ditton, R. B., H. S. M., and D. K. Anderson. 2002. Recreational fishing as tourism. Fisheries 27(3):17-24.
Felley, J. D. 1992. Medium—low-gradient streams of the Gulf Coastal Plain. Pages 233-269 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.
Flebbe, P. A. 1993. Comment on Meisner (1990): Effect of climatic warming on the southern margins of the native range of brook trout, *Salvelinus fontinalis*. Canadian Journal of Fisheries and Aquatic Science 50:883-884.
Fuller, P. L., L. G. Nico, and J. D. Williams. 1999. Nonindigenous Fishes Introduced into Inland Waters of the United States, volume special publication 27. American Fisheries Society, Bethesda, MD.
Gaedke, U., D. Ollinger, P. Kirner, and E. Bauerle. 1998. The influence of weather conditions on the seasonal plankton development in a large and deep lake (L. Constance) III: The impact of water column stability on spring algal development. Pages 71-84 *in* D. G. e. a. George, editor. Management of Lakes and Reservoirs during Global Climate Change. Kluwer Academic Publishers, The Netherlands.
Garman, G. C., and L. A. Nielsen. 1992. Medium-sized rivers of the Atlantic Coastal Plain. Pages 315-349 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity

of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.
Gerdaux, D. 1998. Fluctuations in lake fisheries and global warming. Pages 263-272 *in* D. G. e. a. George, editor. Management of Lakes and Reservoirs during Global Climate Change. Kluwer Academic Publishers, The Netherlands.
Goldman, C. E., and A. J. Horne. 1983. Limnology. McGraw-Hill, Inc.
Hargreaves, J. A., and C. S. Tucker. 2003. Defining loading limits of static ponds for catfish aquaculture. Aquacultural Engineering 28:47-63.
Harper, D., and K. Mavuti. 1996. Freshwater wetlands. R. R. McClanahan, and T. P. Young, editors. East African Ecosystems and Their Conservation. Oxford University Press, New York.
Hefer, B., and Y. Pruginin. 1981. Commercial Fish Farming: with special reference to fish culture in Israel. Wiley and Sons, New York.
Hill, D. K., and J. J. Magnuson. 1990. Potential effects of global climate warming on the growth and prey consumption of Great Lakes fish. Transactions of the American Fisheries Society 119:265-275.
Jacono, C. C., and M. M. Richerson. 2003a. Eichhornia Crassipes. U. S. Geological Survey.
Jacono, C. C., and M. M. Richerson. 2003b. Hydrilla Verticillata. U. S. Geological Survey.
Joyce, J. C. 1992. Impact of *Eichhornia* and *Hydrilla* in the United States. Pages 106-109 *in* ICES Marine Science Symposia: Introductions and Transfers of Exotic Species. International Council for Exploration of the Sea, Halifax, Nova Scotia.
Keith, W. E. 1975. Management by water level manipulation. H. Clepper, editor Black bass biology and management. National Symposium on the Biology and Managemet of the Centrarchid Basses. Sport Fishing Institute, Washington, D. C., Tulsa, Oklahoma.
Killgore, K. J., and J. A. Baker. 1996. Patterns of larval fish abundance in a bottomland hardwood wetland. Wetlands 16(3): 288-295.
Kling, G. W., and coauthors. 2003. Confronting Climate Change in the Great Lakes Region: Impacts on our Communities and Ecosystems. Union of Concerned Scientists, Cambridge, Massachusetts, and Ecological Society of America, Washington, D.C.
Langford, T. E. 1983. Electricity Generation and the Ecology of Natural Waters. Liverpool University Press, Liverpool.
Management, F. D. o. I. P. 2001. Controlling Invasive Exotic Plants on Florida's Public Lands and Waters. Florida Department of Environmental Protection.
Matthews, W. J., L. G. Hill, and S. M. Schellhaass. 1985. Depth distribution of striped bass and other fihs in Lake Texoma (Oklahoma-Texas) during summer stratification. Transactions of the American Fisheries Society 114:84-91.
McCauley, R., and T. Beitinger. 1992. Predicted effects of climate warming on the commercial culture of the channel catfish *Ictalurus punctatus*. Geojournal 28(1):61-66.
Meisner, J. D. 1990. Potential loss of thermal habitat for brook trout, due to climatic warming, in two southern Ontario streams. Transactions of the American Fisheries Society 119:282-291.
Meisner, J. D., J. S. Rosenfeld, and H. A. Regier. 1988. The role of groundwater in the impact of climate warming on stream salmonines. Fisheries 13(3):2-8.
Mitchell, S., and J. J. Cech. 1994. Comparison of metabolic rates of striped bass, white bass, and their hybrid at selected temperatures and dissolved oxygen levels. Pages 275-279 *in* D. MacKinlay, editor High Performance Fish: Proceedings of an International Fish

Physiology Symposium at the University of British Columbia in Vancouver, Canada, July 16-21, 1994. Fish Physiology Association; Vancouver, B.C., Vancouver, BC (Canada).

Morgan, I. J., L. M. L'Cruz, J. J. Dockray, T. K. Linton, and C. M. Wood. 1999. The effects of elevated summer temperature and sublethal pollutants (ammonia, pH) on protein turnover in the gill and liver of rainbow trout (Oncorhynchus mykiss) on a limited food ration. Comparative Biochemistry and Physiology Part A 123:43-53.

Moss, J. L. 1985. Summer selection of thermal refuges by striped bass in Alabama reservoirs and tailwaters. Transactions of the American Fisheries Society 114:77-83.

Moyle, P. B., and J. J. Cech. 1988. Fishes: an Introduction to Ichthyology, 2nd edition. Prentice Hall, Englewood Cliffs, N.J.

Mqolomba, T. N., and J. A. Plumb. 1992. Effect of temperature and dissolved oxygen concentration on *Edwardsiella ictaluri* in experimenatlly infected channel catfish. Journal of Aquatic Animal Health 4:215-217.

Mulholland, P. J., and coauthors. 1997. Effects of climate change on freshwater ecosystems of the South-eastern United States and the Gulf Coast of Mexico. Hydrological Processes 11:949-970.

Mulholland, P. J., and D. R. Lenat. 1992. Streams of the southeastern piedmont, Atlantic Drainage. Pages 193-231 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.

Nagel, J. W. 1991. Is the decline of brook trout in the southern Appalachians resulting from competitve exclusion and/or extinction due to habitat fragmentation? Journal of the Tennessee Academy of Science 66:141-143.

NASC. 2005a. Catfish Processing: 2005. National Agricultural Statistics Center, U. S. Department of Agriculture, Washington, D. C.

NASC. 2005b. Trout Production. National Agricultural Statistics Center, Washington, D. C.

Nijssen, B., G. M. O'Donnell, A. F. Hamlet, and D. P. Lettenmaier. 2001. Hydrologic sensitivity of global rivers to climate change. Climatic Change 50:143-175.

Plumb, J. A. 2001. Overview of warm-water fish diseases. C. Lim, and C. D. Webster, editors. Nutrition and Fish Health. Haworth Press, New York.

Poff, N. L., S. Tokar, and P. Johnson. 1996. Stream hydrological and ecological responses to climate change assessed with an artificial neural network. Limnology and Oceanography 41(5):857-863.

Reddy, K. R., P. D. Sacco, D. A. Graetz, and e. al. 1983. Effect of aquatic marcrophytes on physico-chemical parameters of agricultural drainage water. journal of Aquatic Plant Management 21:1-7.

Ries, R. D., and S. A. Perry. 1995. Potential effects of global climate warming on brook trout growth and prey consumption in central Appalachian streams, USA. Climate Research 5(3):197-206.

Ross, R. M., W. A. Lellis, R. M. Bennett, and C. S. Johnson. 2001. Landscape determinants of nonindigenous fish invasions. Biological Invasions 3:347-361.

Ross-Flanagan, N. 2003. Researcher Identifies North American Hotspots for Sish Conservation. Pages Electronic press release *in* University of Michigan News and Information Services. University of Michigan, Ann Arbor, Michigan.

Saunders, D. E., and S. Johnson. 2004. Keeping invasive aquatic plants at bay. Louisiana Agriculture 47(2).

Schmitz, D. C., and coauthors. 1991. The ecological impact and management history of three invasive alien aquatic plant species in Florida. B. N. McKnight, editor Biological pollution : the control and impact of invasive exotic species : proceedings of a symposium held at the University Place Conference Center, Indiana University-Purdue University at Indianapolis. Indiana Academy of Sciences, Indianapolis.

Schmitz, D. C., D. Simberloff, R. H. Hofsteter, W. Haller, and D. Sutton. 1997. The ecological impact of nonindigenous plants. Pages 39-62 *in* Strangers in Paradise: Impact and Management of Nonindigenous Species in Florida. Island Press, Washington, D. C.

Shafland, P. L. 1996. Exotic fishes of Florida — 1994. Reviews in Fisheries Science 4(2):101-122.

Shafland, P. L., and M. S. Stanford. 1999. The introduced butterfly peacock (*Cichla ocellaris*) in Florida. IV. Socioeconomic analyses. Reviews in Fisheries Science 7(2):127-135.

Shuter, B. J., and J. D. Meisner. 1992. Tools for assessing the impact of climate change on freshwater fish populations. Geojournal 28(1):7-20.

Siemien, M. J., and J. R. J. Stauffer. 1989. Temperature preference and tolerance of the spotted tilapia and Rio Grande cichlid. Archiv fur Hydrobiologie 115(2):287-303.

Smock, L. A., and E. Gilinsky. 1992. Coastal plain blackwater streams. Pages 271-313 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.

Soballe, D. M., B. L. Kimmel, R. H. Kennedy, and R. F. Gaugush. 1992. Reservoirs. Pages 421-474 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.

Steward, K. K., and T. K. Van. 1987. Comparative studies of monoecious and dioecious hydrilla (Hydrilla verticillata) biotypes. Weed Science 35(2):204-210.

Stickney, R. R. 2000. Encyclopedia of Aquaculture. John Wiley and Sons, New York.

Topping, J. C., Jr., and J. P. Bond. 1988. The Potential Impact of Climate Change on Fisheries and Wildlife in North America. The Climate Institute, U. S. Environmental Protection Agency.

USDOI, USFWS, USDOC, and U.S. C. Bureau. 2001. 2001 National Survey of Fishing, Hunting, and Wildlife-Based Recreation. U. S. Department of the Interior, U. S. Fish and Wildlife Service, U. S. Department of Commerce, and U. S. Census Bureau, Washington, D. C.

Wallace, J. B., J. R. Webster, and R. L. Lowe. 1992. High-gradient streams of the Appalachians. Pages 133-191 *in* C. T. Hackney, S. M. Adams, and W. H. Martin, editors. Biodiversity of the Southeastern United States: Aquatic Communities. John Wiley and Sons, New York.

Ward, A. K., G. M. Ward, J. Harlin, and R. Donahoe. 1992. Geological mediation of stream flow and sediment and solute loading to stream ecosystems due to climate change. Pages 116-142 *in* P. Firth, and S. G. Fisher, editors. Global Climate Change and Freshwater Ecosystems. Springer-Verlag, New York.

Warren, M. L., and coauthors. 2000. Diversity, distribution, and conservation status of the native freshwater fishes of the Southern United States. Fisheries 25(10):7-31.

Wedemeyer, G. 1996. Physiology of fish in intensive culture systems. Chapman and Hall, New York.

Welcomme, R. 1979. Fisheries Ecology of Floodplain Rivers. Longman, Inc., New York.
Zaret, T. N., and R. T. Paine. 1973. Species introductions in a tropical lake. Science 182:444-455.

Table 1.—Number of anglers fishing in the freshwater habitats of the Southeast in 2001. Data from Alabama and Arkansas are from 2002. Revenue estimates include expenditures on equipment and travel to the fishing site. Adapted from USDOI 2001.

State	Anglers per year		Revenue
	Resident	Nonresident	
Alabama	749,000	558,000	636 million
Arkansas	539,000	243,000	445 million
Florida	819,000	390,000	1.6 billion
Georgia	909,000	133,000	522 million
Kentucky	590,000	190,000	545 million
Louisiana	492,000	138,000	457 million
Mississippi	391,500	118,000	183 million
North Carolina	681,400	358,000	917 million
South Carolina	474,000	200,000	464 million
Tennessee	709,000	194,000	480 million
Virginia	616,000	201,000	419 million
West Virginia	250,000	67,000	102 million

Figure 1.—The southeastern United States and its major rivers.