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WATER STRESS PROJECTIONS FOR THE NORTHEASTERN AND MIDWESTERN UNITED STATES IN 2060: ANTHROPOGENIC AND ECOLOGICAL CONSEQUENCES[1]

Brian G. Tavernia, Mark D. Nelson, Peter Caldwell, and Ge Sun[2]

ABSTRACT: Future climate and land-use changes and growing human populations may reduce the abundance of water resources relative to anthropogenic and ecological needs in the Northeast and Midwest (U.S.). We used output from WaSSI, a water accounting model, to assess potential changes between 2010 and 2060 in (1) anthropogenic water stress for watersheds throughout the Northeast and Midwest and (2) native fish species richness (i.e., number of species) for the Upper Mississippi water resource region (UMWRR). Six alternative scenarios of climate change, land-use change, and human population growth indicated future water supplies will, on average across the region, be adequate to meet anthropogenic demands. Nevertheless, the number of individual watersheds experiencing severe stress (demand > supplies) was projected to increase for most scenarios, and some watersheds were projected to experience severe stress under multiple scenarios. Similarly, we projected declines in fish species richness for UMWRR watersheds and found the number of watersheds with projected declines and the average magnitude of declines varied across scenarios. All watersheds in the UMWRR were projected to experience declines in richness for at least two future scenarios. Many watersheds projected to experience declines in fish species richness were not projected to experience severe anthropogenic water stress, emphasizing the need for multidimensional impact assessments of changing water resources.

(KEY TERMS: fish; climate variability/change; surface water hydrology; land-use/land-cover change; planning; water supply; water use; water stress; GIS.)

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INTRODUCTION

Growing human populations require adequate water supplies to avoid social strife and economic hardships, such as income loss due to crop failure (Morehart *et al.*, 1999), and to meet biodiversity conservation goals, such as providing aquatic habitat (Spooner *et al.*, 2011). For areas in the Northeast and Midwest (U.S.), there is concern that demands could exceed water supplies in the future (U.S. General Accounting Office, 2003), and at the same time, there is uncertainty about the future abundance of water supplies (Bates *et al.*, 2008; Karl *et al.*, 2009). Increasing temperatures and associated increases in rates of evapotranspiration have the potential to reduce water

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[2]Respectively, Research Associate (Tavernia), Department of Biology, North Carolina State University, 127 David Clark Labs, Raleigh, North Carolina 27695; Research Forester (Nelson), USDA Forest Service, Northern Research Station, St. Paul, Minnesota 55108; and Research Hydrologist (Caldwell, Sun), USDA Forest Service, Southern Research Station, Raleigh, North Carolina 27606 (E-Mail/Tavernia: btavernia@gmail.com).

abundance in the region (Gleick, 2000; National Assessment Synthesis Team, 2000; Levin *et al.*, 2002). Nevertheless, the ultimate influence of increased temperatures will depend on changes in the amount, timing, and variability in regional precipitation, as well as the frequency and intensity of storm events (Gleick, 2000; National Assessment Synthesis Team, 2000; Levin *et al.*, 2002). Climate change projections depend on the general circulation model (GCM) used to model future climate conditions and on assumptions about future greenhouse gas emissions used to drive the GCM (USDA Forest Service, 2012); for this reason, assessments of the future balance between water demands and supplies should consider projections from multiple GCMs and emission scenarios to encapsulate a range of potential future climate conditions (Gleick, 2000; Pierce *et al.*, 2009; Mote *et al.*, 2011). Regionally, population growth and land-use change might be important determinants of the balance between water supplies and demand (Vörösmarty *et al.*, 2000; Alcamo *et al.*, 2003; Sun *et al.*, 2008). The demographic and economic trajectories of human populations will shift the proportions of land-use types throughout the Northeast and Midwest (Wear, 2011). Given that land-use types differ in their hydrologic properties, land-use change may influence the balance of water supply and demand. For example, higher demands for surface and groundwater withdrawals may result from an increase in the proportion of croplands irrigated due to drying climate conditions (Eheart and Tornil, 1999).

Beyond concerns regarding anthropogenic water stress, changing surface water supplies and growing anthropogenic water demands may reduce the discharge volume of streams and rivers, negatively affecting the species richness (i.e., number of species) of fish assemblages (Xenopoulos and Lodge, 2006; Spooner *et al.*, 2011). Fish species richness responds in a positive, logarithmic fashion to discharge volume (Xenopoulos and Lodge, 2006; Spooner *et al.*, 2011), possibly because larger rivers support a greater diversity of habitats and increased energy availability (Eadie *et al.*, 1986; Guégan *et al.*, 1998), or harbor larger and less extinction-prone populations of diverse species which receive a larger and more diverse pool of immigrants (Hugueny, 1989; Lévêque *et al.*, 2008), or a combination of these factors (cf., Connor and McCoy, 1979). Accordingly, reductions in discharge volume can reduce fish species richness by altering these factors (Kanno and Vokoun, 2010). Researchers have used relationships between fish species richness and discharge volume, or species richness-discharge relationships (SDRs), to assess potential effects of reduced discharge levels on fish assemblages (e.g., Xenopoulos and Lodge, 2006; Spooner *et al.*, 2011).

Long-range planning has the potential to reduce socioeconomic hardships and the ecological impacts that would follow from water shortages (Easterling, 1996; Mawdsley *et al.*, 2009). Broadly, our goal was to assess the balance between future water supplies and demands throughout the Northeast and Midwest to aid water resource managers and policy makers in planning for potential water scarcities. Our specific objectives were to: (1) use a water accounting model to assess the balance between water supplies and anthropogenic demands under different scenarios of climate change, land-use change, and human population growth over the next 50 years, and (2) use SDRs to project potential reductions in fish species richness as a result of reduced discharge volumes under the same scenarios.

METHODS

Region of Interest

We used water stress and discharge output from a water accounting model, the Water Supply Stress Index (WaSSI) model (Sun *et al.*, 2008, 2011; Caldwell *et al.*, 2012), to assess potential future changes in anthropogenic water stress and fish species richness for watersheds in the Northeast and Midwest (49°23′ to 35°56′N, and from 97°18′ to 66°53′W) (Figure 1). The Northeast and Midwest contain or intersect 551 eight-digit Hydrological Unit Code watersheds (hereafter HUCs) as represented by the Natural Resources Conservation Service's (NRCS) Watershed Boundary Dataset GIS layer (Accessed April 2011, http://datagateway.nrcs.usda.gov/) (Figure 1). This area encompasses the U.S. Forest Service's Eastern Region, and as part of the 2010 Forest and Rangeland Renewable Resources Planning Act (RPA) Assessment, the U.S. Forest Service has created future scenarios of climate change, land-use change, and human population growth to project 2060 forest and rangeland conditions for this region (USDA Forest Service, 2012). We used these scenarios (described below) to project potential changes in anthropogenic water stress (via future WaSSI, or water stress, values) and fish species richness (via future net discharge volume rates) in 2060.

Future Scenarios

IPCC Storylines and General Circulation Models. The RPA Assessment used climate (Coulson *et al.*, 2010a, b), land-use (Wear, 2011), and population (Zarnoch *et al.*, 2010) projections consistent with greenhouse gas emission scenarios developed by the Intergovernmental Panel on Climate Change (IPCC) (USDA Forest Service, 2012). Emission scenarios

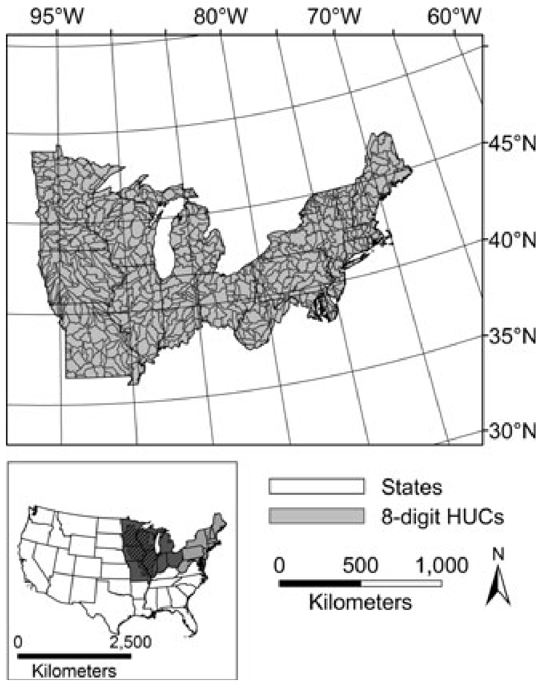

FIGURE 1. Map Showing the Location of Eight-Digit Hydrologic Unit Code Basins Across the Northeast and Midwest, U.S. and Map Showing the Location of the Study Region Within the Conterminous U.S. (locator map). Light gray states are in the Northeast and dark gray states are in the Midwest. Northeastern states include: Connecticut, Delaware, Massachusetts, Maryland, Maine, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Vermont, and West Virginia. Midwestern states include: Iowa, Illinois, Indiana, Michigan, Minnesota, Missouri, Ohio, and Wisconsin. Hatching indicates the Upper Mississippi water resources region.

resulted from IPCC storylines that made different assumptions about changing global populations and gross domestic product (Table 1). For the RPA Assessment, the U.S. Forest Service elected to use IPCC's A1B, A2, and B2 storylines because these captured a range of potential futures likely to drive variation in natural resources and because marker scenarios had been developed for these storylines (USDA Forest Service, 2012). Marker emission scenarios used common assumptions about driving forces in storylines, were

TABLE 1. Projections of Global Population and Global Gross Domestic Product (GDP) Associated with Intergovernmental Panel on Climate Change Storylines.

Storyline	2010	2020	2040	2060
Global population (millions)				
A1	6,805	7,493	8,439	8,538
A2	7,188	8,206	10,715	12,139
B2	6,891	7,672	8,930	9,704
Global GDP (2006 trillion US$)				
A1	54.2	80.6	181.8	336.2
A2	45.6	57.9	103.4	145.7
B2	67.1	72.5	133.3	195.6

Source: USDA Forest Service (2012).

intended to illustrate their respective storylines, and were subjected to greater scrutiny. Recent observations of greenhouse gas emissions (Raupach *et al.*, 2007) suggest that projected emissions under the B2 storyline may underestimate actual emissions, and for that reason, we chose to use the A1B and A2 storylines for our water resource assessments.

There are many sources of uncertainty when assessing future changes in natural resources conditions (Beaumont *et al.*, 2008). By representing a range of likely future climate, land-use, and population conditions, use of A1B and A2 storylines captures some uncertainty involved in assessing future water resource conditions. Greenhouse gas emissions associated with these storylines produce different climate change projections depending on the general circulation model (GCM) used to simulate future climate. For the RPA, the U.S. Forest Service projected future climate change using projections from three GCMs: CGCM 3.1 developed by the Canadian Centre for Climate Modeling and Analysis; CSIRO MK 3.5 developed by Australia's Commonwealth Scientific and Industrial Research Organization; and MIROC 3.2 developed jointly by Japan's National Institute for Environmental Studies, Center for Climate System Research, University of Tokyo, and Frontier Research Center for Global Change. The U.S. Forest Service selected these models because they had average or above average sensitivity to greenhouse gas emissions (Randall *et al.*, 2007), showed a reasonable degree of accuracy when simulating present-day mean climate conditions (Reichler and Kim, 2008), and produced a range of future climate conditions (see Results section).

To address uncertainty resulting from choice of IPCC storylines and GCMs, we used the WaSSI model to assess potential changes in WaSSI values, or anthropogenic water stress, and net discharge volume for 2060 under six scenarios representing unique combinations of A1B and A2 IPCC storylines and CGCM, CSIRO, and MIROC GCMs (Table 2). Changes in net discharge can be used in combination with SDRs to project potential declines in fish species richness (described below). Baseline water stress and net discharge volumes were determined by assigning current values to population (2006), land use (2010), and climate (1996-2006). Available baseline data varied slightly in year of origin, but all baseline data were assumed to represent 2010. Future scenarios used population (2060) and land-use (2060) conditions consistent with either A1B or A2 IPCC storylines and climate conditions (2055-2065) based on the aforementioned GCMs. For baseline and future scenarios, WaSSI values and net discharge volumes were averaged across 11-year periods (1996-2006 for baseline; 2055-2065 for future) to avoid drawing conclusions based on climate conditions for a single year.

TABLE 2. Scenarios of Climate Change, Land-Use Change, and Population Growth Used for Water Resource Assessments.

Storyline	GCM	Climate	Land Use	Population
Baseline	Historical	1996-2006	2010	2006
A1B	CGCM	2055-2065	2060	2060
	CSIRO	2055-2065	2060	2060
	MIROC	2055-2065	2060	2060
A2	CGCM	2055-2065	2060	2060
	CSIRO	2055-2065	2060	2060
	MIROC	2055-2065	2060	2060

Notes: Future scenarios defined using unique combinations of Intergovernmental Panel on Climate Change storylines and general circulation models (GCM). Future climate data from Coulson *et al.* (2010a, b), land-use from Wear (2011), and population projections from Zarnoch *et al.* (2010).

Climate Projections. Historical (1940-2006) and future (2001-2100) climate datasets (Coulson and Joyce, 2010; Coulson *et al.*, 2010a, b) created for the U.S. Forest Service's RPA are archived by the Rocky Mountain Research Station (RMRS) (Accessed April 2011, http://www.fs.fed.us/rm/data_archive/). We used five arc-minute grid datasets from RMRS representing historical and future monthly values for average daily minimum and maximum temperatures as well as monthly total precipitation. Because of their coarse spatial resolution, future climate projections produced by GCMs may fail to capture important subregional climatic patterns caused by local influences (e.g., orographic effects). RMRS future climate projections maintained local signals by overlaying GCM projected changes in temperature and precipitation onto local climate conditions simulated by the Parameter-elevation Regressions on Independent Slopes Model (PRISM Climate Group, Oregon State University, prism.oregonstate.edu) (USDA Forest Service, 2012). Detailed descriptions of methods used to produce RMRS climate datasets can be found in Coulson and Joyce (2010) and Coulson *et al.* (2010a, b).

There were 32,240 five arc-minute grid cells overlapping HUCs in the Northeast and Midwest with an average area of 64 km^2 whereas the 551 eight-digit HUCs had an average area of 3,432 km^2. We needed to aggregate climate data from the county scale to HUCs, a process which could influence the outcome of our analyses (Jelinski and Wu, 1996). We used an area-weighted approach to upscale climate projections from grid cells to HUCs. In brief, where multiple grid cell boundaries intersect a HUC boundary, the climate value from each cell is multiplied by its areal fraction (proportion) of the HUC, and summed with area-weighted values from all other cells that intersect the HUC, resulting in area-weighted mean monthly temperature and precipitation values for each HUC. The area-weighted approach assumes that temperature and precipitation values are homogenous throughout grid cells. Violations of this assumption may bias mean climate values for HUCs, although not in a systematic fashion. Nevertheless, overall mean climate values pre- and post-area weighting are mathematically identical, and we assumed that our area-weighting strategy was a valid approach, consistent with previous water resource assessments and tools (e.g., Price *et al.*, 2010).

Land-Use Projections. For the Forest Service's RPA, Wear (2011) used econometric models to project land-use conditions at the county level for the conterminous United States (U.S.) from 2010 to 2060. Land-use change was modeled only for nonfederal lands because land-use patterns on federal lands were assumed to be constant over this time frame and data were readily available for nonfederal lands. The paucity of federal lands in the eastern U.S. further suggests that nonfederal land-use projections appropriately represent land-use trends for our water resource assessments. Projected land-use types included forest, cropland, pasture, rangeland, and urban areas. The WaSSI model incorporates additional subcategories of these broad land-use categories to account for differing evapotranspiration rates. Consequently, we used a geographic information system (GIS) to produce county-level estimates of current (2001) land-use subcategories based on the MODerate Resolution Imaging Spectroradiometer (MODIS) land-use product MOD12Q1 (Accessed April 2011, http://modis-land.gsfc.nasa.gov/). The forest category was divided into deciduous, coniferous, and mixed forest types, and the rangeland and pasture categories were divided into shrubland, savanna, and grassland cover types. Because Wear's (2011) land-use projections pertained only to nonfederal lands, we excluded federal land areas from our MOD12Q1-based estimates of land use by using the Protected Areas Database for the U.S. (PADUS_v1) (Accessed April 2011, http://www.protectedlands.net/) to mask out federal lands. For modeling purposes, we assumed that proportions of land-use subcategories for forest (deciduous, coniferous, and mixed forest) and rangeland/pasture (shrubland, savanna, and grassland) on nonfederal lands will remain constant across future projections within each county, equal to the estimates of current proportions based on MOD12Q1, regardless of whether total area of forest or rangeland/pasture was projected to change. For forests, we believed this to be a reasonable assumption because distribution patterns for tree species may show a delayed response to changing climate patterns (Iverson *et al.*, 2004), especially over the 50-year time span we considered. Land-use change projections did not include information on the area of water bodies, so we obtained estimates of water coverage for each county from the U.S. Census Bureau (U.S. Census Bureau, 2001).

There were 1,035 counties in the region with an average surface area of 1,659 km^2 based on the U.S. Census Bureau's 2000 County and County Equivalent Areas GIS layer (Accessed September 2011, http://www.census.gov/geo/www/cob/co2000.html). As was done for climate projections, we used an area-weighted approach to rescale county land-use projections to eight-digit HUCs.

Population Projections. In support of the Forest Service's RPA, Zarnoch *et al.* (2010) created county-level population projections for the years 2006-2060 at five-year increments (Accessed April 2011, http://www.treesearch.fs.fed.us/pubs/35892). They produced the population dataset by disaggregating U.S. population projections consistent with A1B and A2 IPCC storylines to counties. Disaggregation for the period from 2010 to 2035 was based on county shares of national growth (Woods & Poole Economics Inc., 2006. The 2006 Complete Economic and Demographic Data Source. www.woodsandpoole.com), whereas a recursive approach was used to project county growth to 2060 (Zarnoch *et al.*, 2010). We used linear interpolation to produce annual population projections from semi-decadal projections provided by Zarnoch *et al.* (2010) and an area-weighted approach to scale county projections to eight-digit HUCs.

Water Balance Model and Water Supply Stress Modeling

Quantitative comparisons of total annual water demand (WD) and water supply (WS) volumes were used to determine the level of water stress for each HUC, a quantity referred to as the Water Supply Stress Index (WaSSI) (Sun *et al.*, 2008):

$$\mathrm{WaSSI} = \mathrm{WD}/\mathrm{WS} \tag{1}$$

WaSSI values greater than one indicate a watershed where demand exceeds supply and suggests that endogenous water supplies need to be supplemented (e.g., by water transfers from surrounding HUCs).

To calculate WaSSI values, we used a water accounting model (Sun *et al.*, 2011; Caldwell *et al.*, 2012) capable of assessing monthly WD, WS, and net discharge at the scale of eight-digit HUCs. In addition to climate, land-use, and population data (see above), the WaSSI model requires information on ground and surface water withdrawals as well as return flow rates. We determined baseline ground and surface water withdrawals using information in the U.S. Geological Survey 2005 nationwide survey of water use (Kenny *et al.*, 2009). This survey reports information for eight different sectors of water users across the Northeast and Midwest: thermoelectric power (71.0% of freshwater withdrawal volume), public supply (14.1%), industrial (7.1%), irrigation (3.4%), domestic (1.5%), mining (1.2%), aquaculture (1.2%), and livestock (0.5%). Return flow, or water that returns to surface water sources after nonconsumptive use, is an important consideration in calculating water supplies. We determined return flow rates for water use sectors using information contained in the U.S. Geological Survey 1995 water use survey (Solley *et al.*, 1998); this is the most recent report with return flow rate information included. Return flow rates vary substantially among sectors (e.g., as high as 97.5% for some thermoelectric power users *vs.* 39.3% for irrigation).

For each month of a simulation, flow accumulation, human consumptive use, and routing calculations were performed in order from the most to the least upstream watershed of the ocean or an international border. In each watershed, human consumptive surface water use that takes place in that watershed was subtracted from the routed flow to the inlet of the next downstream watershed (Caldwell *et al.*, 2012). Each watershed was assumed to have one outlet, but may have zero or more than one watershed flowing into it. It was assumed that all return flows, whether originating from surface or groundwater sources, was returned to surface water. This assumption was necessary because the relative proportion of return flow returning to groundwater or surface water is not noted in the U.S. Geological Survey water use data (Solley *et al.*, 1998).

The discharge (D) out of a given watershed was computed as:

$$D = \sum \mathrm{Q_{up}} + \mathrm{Q_{gen}} - \sum \mathrm{CU} + \sum \mathrm{GW_{ret}} \tag{2}$$

where $\sum \mathrm{Q_{up}}$ is the sum of the flows from upstream watersheds after accounting for human consumptive use in those upstream watersheds; $\mathrm{Q_{gen}}$ is the flow derived from water yield generated in the given watershed due to natural water balance processes (i.e. in the absence of withdrawals and returns); $\sum \mathrm{CU}$ is total consumptive surface water use across the eight water use sectors in the given watershed; and $\sum \mathrm{GW_{ret}}$ is total groundwater return flows across the eight water use sectors in the given watershed. If there are no watersheds upstream of the given watershed, $\mathrm{Q_{up}} = 0$.

The flow derived from water yield generated in a given watershed due to natural water balance processes ($\mathrm{Q_{gen}}$) was computed as:

$$\mathrm{Q_{gen}} = (\mathrm{PPT} - \mathrm{ET} \pm \Delta S) \times A \tag{3}$$

where PPT is precipitation, ET is computed evapotranspiration, ΔS is computed change in soil water storage, and A is watershed area. Estimates of ET are a function of potential evapotranspiration, land-use type, leaf area index, soil moisture content, and precipitation (Sun *et al.*, 2011; Caldwell *et al.*, 2012). Thus, Q_{gen} can vary as a function of land use because land-use types are dominated by different vegetation types that have unique transpiration rates.

Total consumptive surface water use over the eight water use sectors in a given watershed ($\sum$CU) was computed as:

$$\sum \mathrm{CU} = \sum[(1 - R_{\mathrm{frac}}) \times \mathrm{WD_{sw}}] \tag{4}$$

where R_{frac} is fraction of total water withdrawn that is not consumed (Solley *et al.*, 1998), and WD_{SW} is total freshwater withdrawals from surface water sources (Kenny *et al.*, 2009).

Total groundwater return flows assumed to return to surface water across the eight water use sectors ($\sum GW_{ret}$) were computed as:

$$\sum \mathrm{GW_{ret}} = \sum[\mathrm{R_{frac}} \times \mathrm{WD_{GW}}] \tag{5}$$

where WD_{GW} is total freshwater withdrawals from groundwater sources (Kenny *et al.*, 2009). For this study, groundwater withdrawals were assumed to remain at 2005 levels for all future scenarios.

In months where $\sum$CU exceeded the sum of Q_{up}, Q_{gen}, and GW_{ret} (i.e., discharge would be negative), discharge was set to zero and the remaining water withdrawal was assumed to be supplied by an infinite water supply reservoir (e.g., deep water well).

For each eight-digit HUC, we defined WS as the total potential amount of water available for withdrawals at the outlet of the HUC, represented as:

$$\mathrm{WS} = \mathrm{SS} + \mathrm{WD_{GW}} \tag{6}$$

where SS is the total surface water supply, and WD_{GW} is the total groundwater withdrawal. SS was defined as:

$$\mathrm{SS} = \mathrm{Q_{gen}} + \mathrm{Q_{up}} \tag{7}$$

Defined in this manner, SS accounts for consumptive water use in upstream watersheds, but does not include the impact of consumptive use in the watershed in question. Note that SS is discharge, D, without the local consumptive surface water use ($\sum$CU) and groundwater return flows ($\sum GW_{ret}$) included. In this study, we did not attempt to link changes in ecosystem or human evaporative water use to changes in precipitation.

WD was determined by summing total gross water use (WU) from both surface and groundwater withdrawals for each sector (Kenny *et al.*, 2009):

$$\mathrm{WD} = \sum \mathrm{WU}_i, i = 1 - 8 \text{ water use sectors} \tag{8}$$

For future water stress projections, we developed an equation to project levels of domestic WD. Domestic WD was considered equivalent to the sum of total water use in the domestic sector and the portion of public supply water use that was delivered to meet domestic demands (Kenny *et al.*, 2009). Over the conterminous U.S., we correlated 2005 domestic WD (millions of gallons per day) (Kenny *et al.*, 2009) to 2006 population levels (in thousands of persons) (Zarnoch *et al.*, 2010):

$$\begin{gathered}\text{Domestic WD} = 0.0931 \times \text{population}, R^2 = 0.93, \\ n = 2099 \text{ watersheds}\end{gathered} \tag{9}$$

R^2 is the coefficient of determination and represents the proportion of variation in water use explained by population levels. We assumed that WD in all other sectors would remain constant based on past trends (Brown, 2000; Kenny *et al.*, 2009); this assumption included the portion of the public supply water use sector not delivered for domestic use. Projected changes in domestic WD can also influence future WS via changes in return flow volumes.

Fish Species Richness

To assess the potential for changes in fish species richness, we developed a relationship between fish species richness and net discharge volume, referred to as a species richness-discharge relationship (SDR). Our assessments of anthropogenic water stress focused on the sociopolitical boundaries that identify the Northeast and Midwest, but the same extent would not have been appropriate for our assessment of changes in fish species richness. Instead, we focused our assessment on the Upper Mississippi water resources region (WRR) (Figure 1) because WRRs represent an ecologically meaningful scale at which to create SDRs (Spooner *et al.*, 2011). We constrained our SDR to this single WRR because the Mississippi valley region is known as a native fish species richness hotspot (Stohlgren *et al.*, 2006) and because other WRRs in the Northeast and Midwest (1) have a large portion of the region extending beyond the Northeast and Midwest, or (2) lack significant relationships between species richness and discharge (Spooner *et al.*, 2011). All HUCs in the Upper Mississippi WRR overlap our larger study area (Figure 1), and fish species richness is related to net discharge volume in this region (see below).

To develop our SDR, we obtained distribution status data for 865 native freshwater fish species across the conterminous U.S. from NatureServe (NatureServe,

2010. Digital Distribution Maps of the Freshwater Fishes in the Conterminous United States, Version 3.0. Arlington, Virginia). We used this dataset to determine a species richness value (i.e., number of species present) for each HUC in the Upper Mississippi WRR. Richness values included species assigned to big river, medium river, creek, and spring/spring brook habitats by NatureServe. NatureServe distribution records for native fish are restricted to those HUCs where species are naturally occurring, i.e., not introduced, and for this reason, our richness values were not inflated by introduced species. Some native species have been widely introduced into HUCs beyond their native range (e.g., northern pike, *Esox lucius*). The NatureServe dataset omits those records from introduced HUCs, but in doing so, sometimes inadvertently removes fish species records from HUCs within their native range (Margaret Ormes and Jason McNees, NatureServe, July 27, 2011, personal communication). However, we assumed that such anomalies would not affect the interpretation of our results because very few native fish species have been omitted from only a small fraction of HUCs in which they occur, and these species are typically not of conservation concern. Fish distribution records corresponded to HUC boundaries in the U.S. Geological Survey's Hydrologic Unit Boundaries GIS layer (Accessed August 2011, http://www.nationalatlas.gov/). For the Upper Mississippi WRR, a small number of these boundaries did not exactly correspond with the NRCS Watershed Boundary Dataset HUC boundaries used by the WaSSI model (2 of 131 WaSSI HUCs). Where mismatches occurred, we used an area-weighted approach (as described above) to produce species richness estimates for HUCs in the WaSSI model. To develop our SDR, we also used average annual net discharge (km^3/yr) data computed by the WaSSI model run under baseline conditions (Table 2).

We created our SDR using the statistical program R v 2.15.0 (R Development Core Team, 2012). Prior to fitting statistical models, we log-transformed species richness and net discharge to linearize the relationship between these two variables. Initially, we used the lm function (STATS library; R Development Core Team, 2012) to implement a general linear model relating richness to net discharge. The effect of discharge was significant ($p < 0.01$), but a Moran's I test (Fortin and Dale, 2005) of model residuals showed evidence of significant ($p < 0.01$) positive spatial autocorrelation, i.e., residuals with similar values tended to be clustered in geographic space. If spatial autocorrelation is ignored, there is an increased risk of committing a Type I error (Fortin and Dale, 2005), i.e., falsely concluding that richness is related to discharge. The eight-digit HUCs used by the WaSSI model are nested within larger, six-digit HUCs, and based on a visual inspection, we felt this nesting may have contributed to the observed autocorrelation. As a result, we used the lme function (NLME library) (Pinheiro *et al.*, 2009) to fit a mixed effects model that included a fixed component relating richness to net discharge and a random component with a random intercept to account for the nested nature of the HUCs. Based on a likelihood ratio test (Zuur *et al.*, 2009), the mixed effects model provided a significantly ($p < 0.01$) better fit than a simple general linear model, and the model's residuals lacked evidence of significant autocorrelation based on a Moran's I test ($p > 0.10$).

From this final model, our SDR for the Upper Mississippi WRR was:

$$\begin{aligned} \mathrm{LR} &= 0.1565 \times \mathrm{LD} + 1.6327, \\ R^2 &= 0.40, n = 131 \text{ watersheds} \end{aligned} \tag{10}$$

where LR is log of species richness, and LD is log of net discharge. Our R^2 value was calculated using log-likelihoods (Kramer, 2005) and indicated the amount of variation in species richness explained by net discharge. Using this SDR, we determined species richness using average annual net discharge output from the WaSSI model for baseline conditions and for each future scenario (Table 2). For each HUC, the difference between richness values for a future scenario and baseline conditions was the projected change in richness. We limited our projections for species richness to those HUCs projected to experience a decrease in net discharge and hence fish species richness because there is uncertainty about the effect of an increase in discharge on fish species richness (Xenopoulos *et al.*, 2005). If projected declines in fish species richness included fractional fish species, we rounded estimates down to the nearest whole number if fractions were <0.5 and up to the nearest whole number if fractions were ≥ 0.5. We rounded declines <0.5 species to 0 and assumed these represented no change in richness values. We assumed that this approach was consistent with other studies that did not report fractional declines in fish species richness (e.g., Spooner *et al.*, 2011). We noted that projected potential declines in fish species richness may not be fully realized by 2060 because of time lags between habitat loss (i.e., reductions in discharge) and species extirpations (Tilman *et al.*, 1994). Nevertheless, we assumed that projections of potential richness declines were useful for comparing future scenarios.

RESULTS

Climate, Land-Use, and Population Projections

Under baseline conditions, HUCs across the Northeast and Midwest displayed an average annual

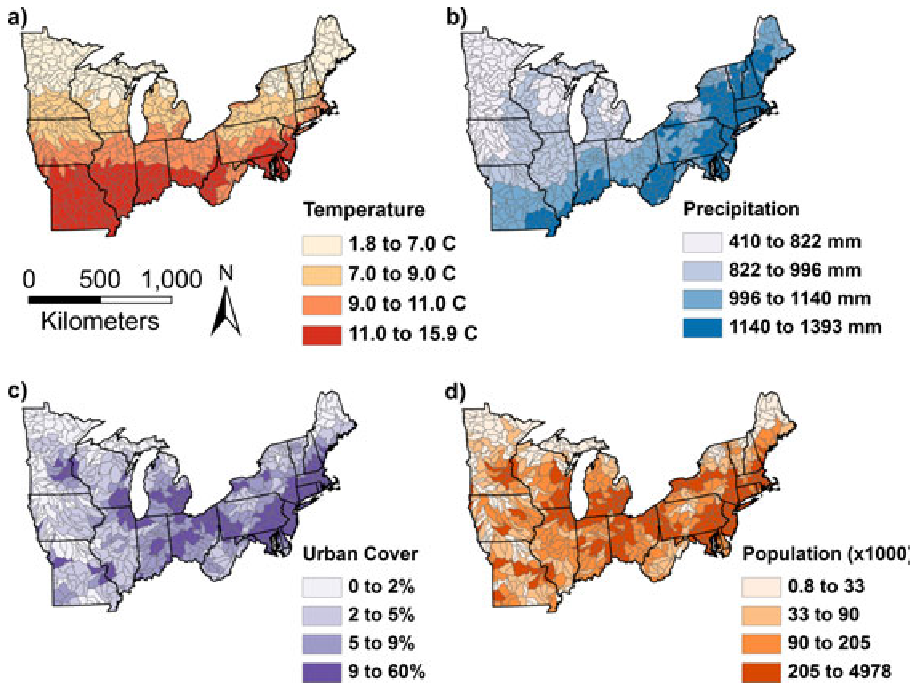

FIGURE 2. (a) Baseline Conditions for Average Annual Temperature (1996-2006), (b) Average Annual Precipitation (1996-2006), (c) Percent Urban Cover (2010), and (d) Human Population Size (2006) for Eight-Digit Hydrologic Unit Code Basins Across the Northeast and Midwest.

temperature of 9.1°C (range: 1.8-15.9°C); temperatures increase from north to south across the region (Figure 2). Average annual precipitation was 977.0 mm (409.8-1,392.5 mm), and precipitation tended to be lower in the northern half of the Midwest (Figure 2). Percent cover of urban land-use averaged 7.9% (0.3-59.3%) with concentrations of urban HUCs located throughout the region (Figure 2). Percent covers for forest, cropland, pasture, and rangeland averaged 40.9% (0.5-94.3%), 30.5% (0-89.3%), 11.1% (0.1-45.3%), and 0.4% (0-23.9%), respectively. The average HUC had 237 thousand people (750-4.9 million), and patterns of population size approximately paralleled those observed for percent urban cover (Figure 2).

For each future scenario, we summarized projected changes in climate, land-use, and human population by averaging changes across HUCs. With the exception of precipitation, future scenarios displayed consistent directions of change in climate, land-use, and population, but magnitudes of change differed (Table 3). Average projected increases in temperature ranged from 2.2 to 4.0°C. Half of the future scenarios projected decreases in precipitation, ranging from an average of −17.6 to −104.5 mm, and the other half projected increases, ranging from an average of 20.3 to 48.5 mm (Figure 3). Growth of urban areas drove losses of other land-use types for all scenarios, but IPCC's A1B storyline projected greater urban growth (average 5.0%) than the A2 storyline (average 3.9%). Population projections indicated that the total regional population will increase from approximately 131 million in 2006 to 169 million in 2060 under A1B and to 190 million under A2. HUCs were projected to add an average of 68 thousand people for A1B and an average of 108 thousand people for A2.

Water Supply Stress

Under baseline conditions, the average WaSSI value across HUCs in the Northeast and Midwest was 0.07, a value associated with low levels of anthropogenic water stress (Figure 4; Table 4). Nevertheless, 11 of 551 HUCs had moderate to high WaSSI values ($1 \geq$ WaSSI ≥ 0.50) and 2 HUCs were water-stressed (WaSSI > 1). Severely stressed HUCs were characterized by a heavy dependence on water withdrawals to meet thermoelectric power demands. The average WaSSI value for the region increased under all future scenarios and ranged from 0.08 to 0.19 (Table 4). The number of HUCs with moderate to high WaSSI values also increased, ranging from 13 to 31, and the number of water-stressed HUCs ranged from 2 to 18. Locations of severely stressed HUCs varied across future scenarios, but 13 HUCs were severely stressed in ≥ 2 future scenarios (Figure 4).

We considered the independent effects of climate change, land-use change, and human population growth on WaSSI values in 2060. Projected population increases for both A1B and A2 led to an average regional WaSSI value of 0.08, and land-use change led to an average WaSSI value of 0.07 for both storylines. The

TABLE 3. Mean Projected Changes in Temperature (T), Precipitation (PPT), Land-Use, and Population Under Six Future Scenarios.

Storyline	GCM	T (°C)	PPT (mm)	% Urban	% Forest	% Crop	% Pasture	% Rangeland	Population (thousands)
A1B	CGCM	2.3 (0.7, 3.2)	36.0 (−115.4, 176.8)	5.0 (0.1, 32.2)	−2.3 (−14.6, 0)	−2.1 (−12.3, 0)	−0.6 (−13.2, 2.7)	0.0 (−0.4, 0)	68.4 (−171.1, 1409.6)
	CSIRO	2.5 (0.5, 3.5)	−17.6 (−197.3, 166.1)	5.0 (0.1, 32.2)	−2.3 (−14.6, 0)	−2.1 (−12.3, 0)	−0.6 (−13.2, 2.7)	0.0 (−0.4, 0)	68.4 (−171.1, 1409.6)
	MIROC	4.0 (0.9, 5.2)	−104.5 (−304.3, 63.1)	5.0 (0.1, 32.2)	−2.3 (−14.6, 0)	−2.1 (−12.3, 0)	−0.6 (−13.2, 2.7)	0.0 (−0.4, 0)	68.4 (−171.1, 1409.6)
A2	CGCM	2.8 (0.8, 3.6)	20.3 (−86.0, 138.9)	3.9 (0, 31.9)	−1.9 (−13.8, 0)	−1.6 (−14.6, 0)	−0.5 (−13.1, 1.8)	0.0 (−0.5, 0)	108.1 (−15.6, 2064.4)
	CSIRO	2.2 (0.6, 3.0)	48.5 (−186.9, 240.0)	3.9 (0, 31.9)	−1.9 (−13.8, 0)	−1.6 (−14.6, 0)	−0.5 (−13.1, 1.8)	0.0 (−0.5, 0)	108.1 (−15.6, 2064.4)
	MIROC	3.3 (0.8, 4.6)	−76.8 (−308.7, 124.5)	3.9 (0, 31.9)	−1.9 (−13.8, 0)	−1.6 (−14.6, 0)	−0.5 (−13.1, 1.8)	0.0 (−0.5, 0)	108.1 (−15.6, 2064.4)

Notes: Future scenarios were defined using unique combinations of Intergovernmental Panel on Climate Change storylines and General Circulation Models (GCM). Means were calculated by averaging values across 551 eight-digit Hydrologic Unit Code watersheds located across the Northeast and Midwest, U.S. The range for each variable is reported in parentheses.

effects of projected climate change depended on the unique combination of IPCC storyline and GCM considered; WaSSI values ranged from 0.07 (A1B-CGCM) to 0.18 (A1B-MIROC).

Fish Species Richness

Under 2010 baseline conditions, the average annual net discharge for the 131 HUCs in the Upper Mississippi WRR was 11.47 km^3/yr (0.3-248.7 km^3/yr) (Figure 5). Fish species richness averaged 53.6 species (15-123 species) (Figure 5). Across future scenarios, the number of HUCs expected to experience declines in net discharge and consequent potential declines in fish species richness varied from 16 to 131 (Table 5). For these HUCs, the average magnitude of projected declines in discharge and fish species richness ranged from 9.9 to 56.1% and from 1 to 6.2 species, respectively, across scenarios. For all 131 HUCs in the Upper Mississippi WRR, at least two future scenarios projected declines in discharge and fish species richness (Figure 5).

When we considered the independent effects of climate change, land-use change, and population growth, we found that mean reductions in net discharge volume and fish species richness were almost entirely driven by projected climate changes. Climate change led to average net discharge declines ranging from 9.9 (A1B-CGCM) to 56.1% (A1B-MIROC) and average richness declines of 1 (A1B-CGCM) to 6.2 (A1B-MIROC) species. While land-use change and population growth were projected to cause net discharge to decline for some HUCs, neither led to consequent losses of fish species richness (i.e., richness declines <0.5 species).

DISCUSSION

We assessed consequences of changing water resources for both anthropogenic water stress and fish species richness in the Midwest and Northeast, U.S. Our study required the development of separate, but related models and geographic extents to address multiple potential impacts of changing water resources. Acknowledging uncertainty about future conditions, we assessed potential changes using six alternative realizations of future climate, land-use, and population consistent with unique combinations of IPCC's A1B and A2 storylines and CGCM, CSIRO, and MIROC GCMs. For all six scenarios, the average WaSSI value (demand/supply), our measure of anthropogenic water stress, increased for HUCs in the Northeast and Midwest, but these values also indicated that supplies

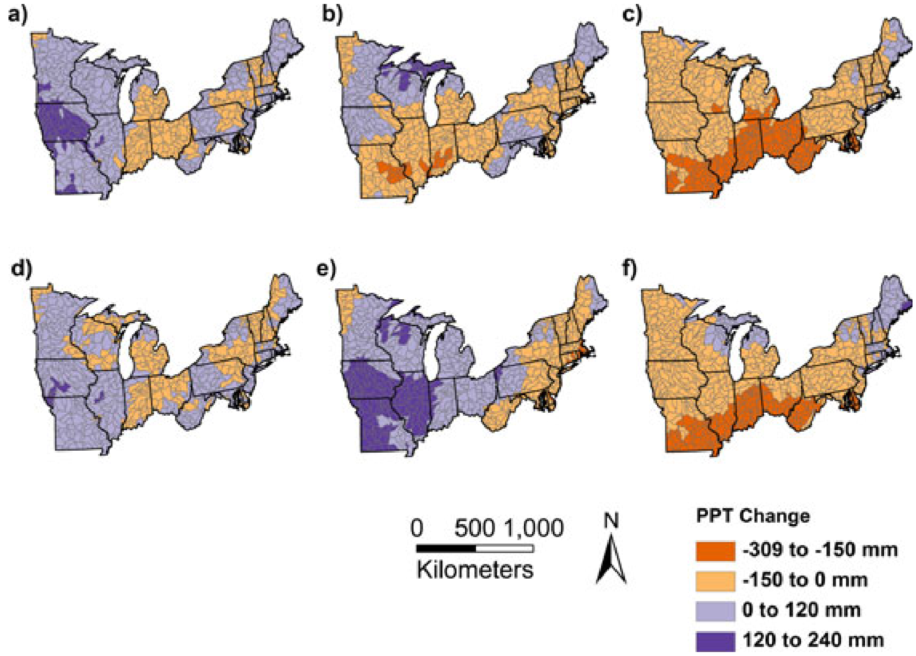

FIGURE 3. Projected Changes in Precipitation for Eight-Digit Hydrologic Unit Code (HUC) Basins Across the Northeast and Midwest. Changes were projected based on unique combinations of two Intergovernmental Panel on Climate Change storylines (A1B, A2) and three General Circulation Models (CGCM 3.1, CSIRO MK 3.5, MIROC 3.2): (a) A1B-CGCM 3.1, (b) A1B-CSIRO MK 3.5, (c) A1B-MIROC 3.2, (d) A2-CGCM 3.1, (e) A2-CSIRO MK 3.5, (f) A2-MIROC 3.2.

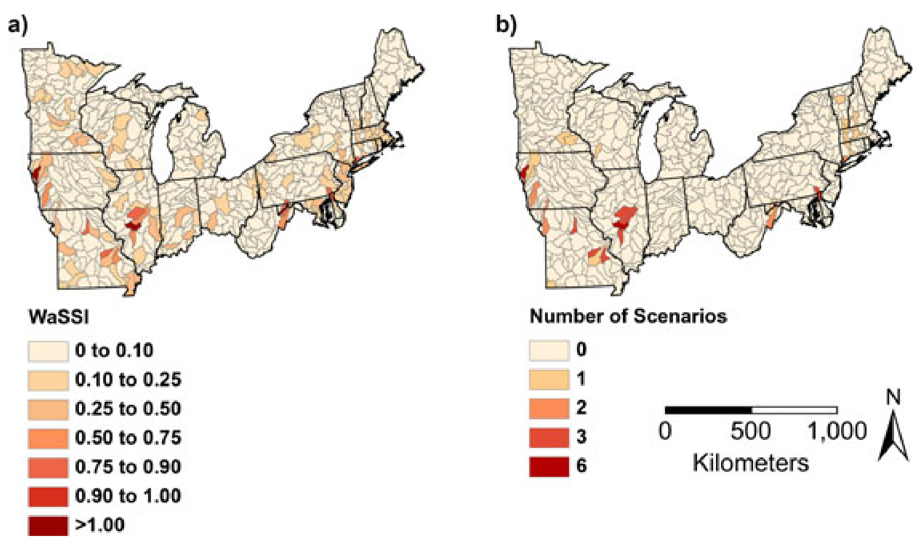

FIGURE 4. (a) Baseline (2010) WaSSI Values and (b) the Number of Future (2060) Scenarios in Which WaSSI Values Were >1 for Eight-Digit Hydrologic Unit Code (HUC) Basins Across the Northeast and Midwest. WaSSI values represent the proportion of a HUC's water supply that is being used to meet anthropogenic demands. WaSSI values >1 indicate that demands exceed supplies and that a HUC is potentially water-stressed. Future WaSSI values were projected based on six scenarios representing unique combinations of two Intergovernmental Panel on Climate Change storylines (A1B, A2) and three General Circulation Models (CGCM 3.1, CSIRO MK 3.5, MIROC 3.2).

exceeded demands for the average HUC as is true for baseline conditions. Despite water supplies being adequate to meet demand on average, our projections also indicated an increase in the number of water-stressed HUCs (WaSSI > 1) across all but one scenario. Locations of water-stressed HUCs varied across scenarios, but there were some HUCs that were stressed under multiple future scenarios. The next step in evaluating the vulnerability of these HUCs is to assess the abilities of social, political, economic, and water delivery

TABLE 4. Water Supply and Stress Index (WaSSI) Values for Eight-Digit Hydrologic Unit Code (HUCs) Basins Across the Northeast and Midwest, U.S.

Storyline	GCM	WaSSI	Number of HUCs: 1 > WaSSI > 0.5	Number of HUCs: WaSSI > 1
Baseline	Historical	0.07 (0, 2.15)	11	2
A1B	CGCM	0.08 (0, 1.36)	13	2
	CSIRO	0.13 (0, 5.21)	21	9
	MIROC	0.19 (0, 7.83)	29	18
A2	CGCM	0.18 (0, 9.99)	31	15
	CSIRO	0.09 (0, 1.90)	15	4
	MIROC	0.14 (0, 6.42)	24	11

Notes: WaSSI values represent the proportion of a HUC's water supply that is being used to meet anthropogenic demands. The mean WaSSI value across 551 HUCs, the value range (in parentheses), number of HUCs with moderate to high values (1 > WaSSI > 0.5), and number of water-stressed HUCs (WaSSI > 1) are reported. Results are reported for 2010 baseline conditions and for six 2060 scenarios defined using unique combinations of Intergovernmental Panel on Climate Change storylines and General Circulation Models (GCM).

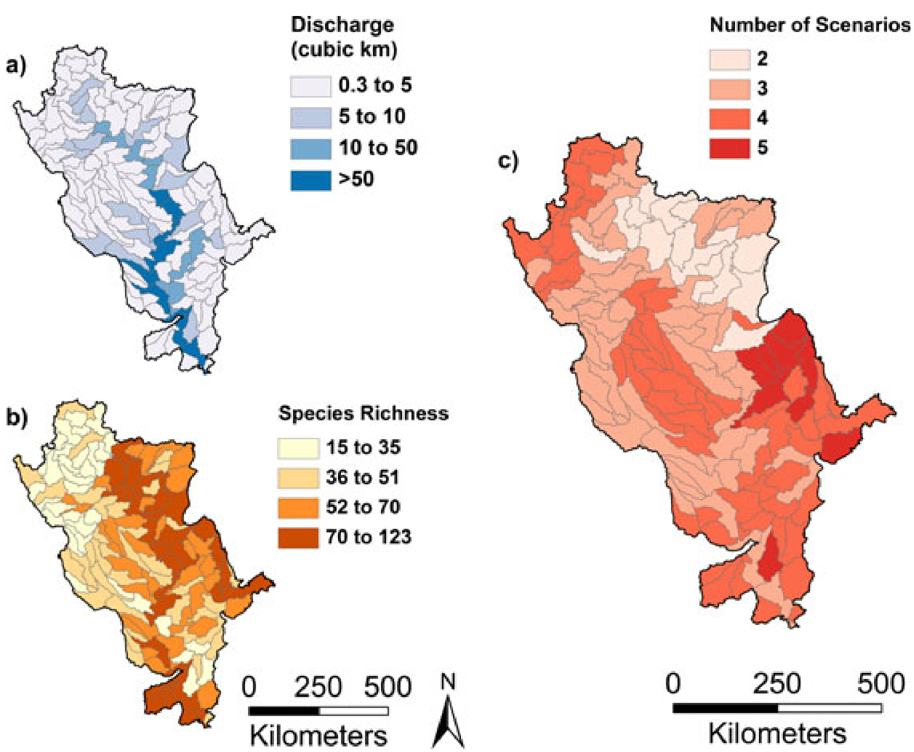

FIGURE 5. (a) Baseline (2010) Net Discharge, (b) Native Fish Species Richness, and (c) the Number of Future (2060) Scenarios in Which Native Fish Species Richness Was Projected to Decline for Eight-Digit Hydrologic Unit Code (HUC) Basins Across the Northeast and Midwest. Species richness is the total number of native fish species inhabiting riverine habitats within each HUC. Declines in fish species richness were projected based on declines in net discharge levels for six future scenarios. Future scenarios represented unique combinations of two Intergovernmental Panel on Climate Change storylines (A1B, A2) and three General Circulation Models (CGCM 3.1, CSIRO MK 3.5, MIROC 3.2).

systems to cope with and adapt to water stress (Lettenmaier *et al.*, 1999; Kelly and Adger, 2000).

Similar to assessments for other regions and spatial extents (Sun *et al.*, 2008; Fung *et al.*, 2011; Caldwell *et al.*, 2012), we found that the magnitude of projected increases in regional anthropogenic water stress was sensitive to changes in water supply due to climate change; projected increases in demand due to population growth or changes in supply due to land-use change had relatively small effects on regional water stress. For example, the average HUC-level change in WaSSI value due to climate change under IPCC's A1B storyline ranged from 0 (CGCM) to + 0.11 (MIROC) whereas land-use change and population growth led to 0 and + 0.01 change values, respectively. The MIROC GCM under IPCC's A1B storyline projected relatively large declines in precipitation and increase in temperature, leading to the

TABLE 5. Mean and Range (in parentheses) for Projected Changes in Net Discharge and Fish Species Richness Across Six Future Scenarios.

Storyline	GCM	Number of HUCs	Discharge (%)	Species Richness
A1B	CGCM	16	−9.9 (−6.4, −17.2)	−1 (−1, −1)
	CSIRO	111	−21.4 (−5.4, −43.0)	−1.9 (−1, −4)
	MIROC	131	−56.1 (−34.8, −66.1)	−6.2 (−3, −13)
A2	CGCM	49	−18.2 (−7.6, −39.2)	−1.6 (−1, −4)
	CSIRO	10	−13.8 (−8.3, −23.8)	−1.2 (−1, −2)
	MIROC	131	−46.6 (−19.7, −67.9)	−4.7 (−2, −11)

Notes: Numbers were summarized for eight-digit Hydrologic Unit Code (HUCs) in the Upper Mississippi water resources region projected to experience a decline in net discharge and richness under future conditions. Results are reported for six 2060 scenarios defined using unique combinations of Intergovernmental Panel on Climate Change storylines and General Circulation Models (GCM).

largest projected WaSSI values due to decreases in water supply. Of the changes in climate, McCabe and Wolock (2011) found that water yield was more sensitive to changes in precipitation than changes in temperature. While water stress depended more on climate projections than land-use or population projections at a regional scale, there were exceptions for individual HUCs. For example, projected increases in population led to greater increases in WaSSI values than did climate projections for HUC 02060004, a watershed near Washington, D.C., regardless of the IPCC storyline or GCM considered.

Biodiversity within aquatic ecosystems is highly dependent on the magnitude, frequency, duration, timing, and rate of change in water moving through the system (Poff *et al.*, 1997). In the Upper Mississippi WRR, our SDR indicated that native fish species richness is positively related to net discharge volumes of HUCs. We did not explicitly test the mechanism behind this relationship, but presume that it follows predominant theory — that watersheds with higher discharge levels offer a greater diversity of habitats and more energy (Eadie *et al.*, 1986; Guégan *et al.*, 1998) and/or support larger and less extinction-prone populations of diverse species which receive a larger and more diverse pool of immigrants (Hugueny, 1989; Lévêque *et al.*, 2008). Our projections suggested that discharge and species richness may decline for HUCs in the Upper Mississippi WRR, but the magnitude of declines and the number and spatial distribution of HUCs experiencing declines differed across future scenarios. Across all scenarios, projected declines in fish species richness for individual HUCs ranged from 1 to 13 species, and every HUC in the region was projected to experience declines in discharge and fish species richness in ≥ 2 future scenarios. Spooner *et al.* (2011) also assessed potential declines in fish species richness for this region using a SDR and projected declines in discharge with a single future scenario, a combination of IPCC's A2 storyline and the HADCM3 (Hadley Center, UK) GCM. We visually inspected Figure 4 in Spooner *et al.* (2011) and concluded that they projected a loss of between 2 and 42 species for HUCs in the Upper Mississippi WRR by 2070. Our projections and those of Spooner *et al.* (2011) are consistent in magnitude with presumed historical losses of fish species (range 0 to 51 species) from HUCs in this WRR (NatureServe, 2010. Digital Distribution Maps of the Freshwater Fishes in the Conterminous United States, Version 3.0). Interestingly, many HUCs not projected to experience anthropogenic water stress were projected to experience declines in fish species richness, stressing the importance of assessing a diverse array of potential impacts resulting from future changes in water resources.

Our assessments of potential changes in WaSSI values and fish species richness might be affected by a number of assumptions underlying the WaSSI model and the SDR. Related to the WaSSI model, there is no nationwide dataset documenting the occurrence and management of water infrastructure related to interbasin transfers (e.g., pipelines) or water storage (e.g., reservoirs). Consequently, although the WaSSI model includes natural water transfers among basins, it does not currently account for anthropogenic water transfer or storage. If a nationally consistent dataset becomes available for incorporation into the WaSSI model, the sensitivity of our assessments to interbasin transfer and water storage infrastructure should be investigated. The WaSSI model also assumes that groundwater withdrawals will remain consistent with observed 2005 levels (Kenny *et al.*, 2009). While this may be a safe assumption for many areas within the Northeast and Midwest (U.S. Geological Survey, Groundwater Depletion. Accessed August 2012, ga.water.usgs.gov/edu/gwdepletion.html), groundwater depletion in some areas may reduce the quantity or quality of groundwater available in the future (Konikow and Kendy, 2005). Thus, groundwater depletion may lead to increased anthropogenic water stress and reduced discharge in areas where streams and rivers are hydraulically connected to groundwater (Konikow and Kendy, 2005).

The WaSSI model uses WaSSI values (demand/supply) as an indicator of potential anthropogenic water stress because it is easily understood. In this study, we assumed that HUCs exceeding a WaSSI value of 1.0

may be water stressed, and we assume that this threshold was useful in identifying HUCs with the potential to experience anthropogenic water stress in the future, allowing for more detailed assessments. Others have used WaSSI values of 0.4 as a threshold for watersheds becoming water stressed (Raskin *et al.*, 1997; Alcamo *et al.*, 2000; Cosgrove and Rijsberman, 2000; Vörösmarty *et al.*, 2000), but in fact the WaSSI value at which a watershed becomes water stressed depends on local water management strategies. For example, a WaSSI value of 0.8 may be common in watersheds of the Colorado River basin and not lead to significant hardship because water management infrastructure is in place to accommodate those conditions (e.g., reservoirs with multiple years of storage available, interbasin transfers via canals, pipelines, etc.). However, a WaSSI value of 0.8 in the Chesapeake Bay area of the Atlantic Coast would indicate severe water shortage and stress because these conditions are rare and water management strategies are not in place to deal with them. We assumed that the entire water supply volume within a HUC is available to be used for anthropogenic purposes, and given that a number of factors may limit water availability (e.g., inadequate infrastructure), our assessments of potential water stress may be conservative.

We also considered five critical assumptions from Olden *et al.* (2010) that, if violated, may limit the utility of SDRs for assessing potential declines in fish species richness. Although the first assumption — that species richness is in equilibrium with discharge — likely varies between glaciated and unglaciated portions of our region over geological time frames, we assumed that rates of projected loss of species richness will be relatively consistent over the short time frame associated with our projections (50 years). The second assumption — that models are valid when extrapolating beyond the range of empirical data — was untested in this study. We did, however, produce our SDR model for HUCs within a single WRR to prevent cross-regional extrapolation. We also noted that our projections of fish species richness declines were consistent with presumed historical losses for HUCs in this WRR. The third assumption — that mean annual discharge is a valid simplification of variable runoff — leads to models without additional flow regime parameters (e.g., seasonality of flow) that affect aquatic communities (Poff *et al.*, 1997). Climate change may act on these additional parameters (e.g., by changing seasonal flow patterns) (Hayhoe *et al.*, 2007), and future efforts should assess the potential importance of including these effects on projections of fish species richness. Climate change may also have more direct effects on fish species by potentially causing stream temperatures to exceed species' thermal tolerances, and this may affect fish assemblages, especially in areas where many species are at or near their thermal tolerances (e.g., the southern Great Plains) (Matthews and Zimmerman, 1990). Nevertheless, our model, relating richness to mean annual discharge, showed relatively high explanatory power (Møller and Jennions, 2002), and we assumed our assessments were useful for comparing alternative scenarios. The fourth assumption — pertaining to variability in spatial scales of basins — likely is of minor concern in this study because only one scale of HUCs was used for modeling SDRs and projecting future species richness. The fifth assumption — that total species richness is adjusted for nonnative species — was addressed in this study by using only native species for modeling SDRs, and projecting future fish species richness only for native species. In aggregate, we felt that these five assumptions were adequately addressed, or were likely to have only minor potential influence on 50-year projections of native fish species richness in the Upper Mississippi WRR, but this conclusion cannot be extrapolated to other regions, scales, time frames, or taxa.

Uncertainties about future climate change, land-use change, and human population growth make it difficult to assess the potential effects of future water resource changes in the Northeast and Midwest. We identified HUCs projected to experience severe anthropogenic water stress and declines in fish species richness under multiple scenarios of climate change, land-use change, and human population growth. These are potential areas of interest for intensive, location-specific evaluations of the vulnerability of anthropogenic systems and fish species to future changes in water resources. Given that projections of anthropogenic water stress and fish species declines were not always spatially congruent, efforts to identify areas vulnerable to changing water resource conditions would benefit from multi-metric impact assessments.

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