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Author manuscript

Trans ASABE. Author manuscript; available in PMC 2021 July 28.

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Published in final edited form as:

Trans ASABE. 2020 January 1; 63(3): 753–770. doi:10.13031/trans.13630.

Modeling the Effects of Future Hydroclimatic Conditions on Microbial Water Quality and Management Practices in Two Agricultural Watersheds

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Abstract

Anticipated future hydroclimatic changes are expected to alter the transport and survival of fecally-sourced waterborne pathogens, presenting an increased risk of recreational water quality impairments. Managing future risk requires an understanding of interactions between fecal sources, hydroclimatic conditions and best management practices (BMPs) at spatial scales relevant to decision makers. In this study we used the Hydrologic Simulation Program FORTRAN to quantify potential fecal coliform (FC - an indicator of the potential presence of pathogens) responses to a range of mid-century climate scenarios and assess different BMP scenarios (based on reduction factors) for reducing the risk of water quality impairment in two, small agricultural watersheds - the Chippewa watershed in Minnesota, and the Tye watershed in Virginia. In each watershed, simulations show a wide range of FC responses, driven largely by variability in projected future precipitation. Wetter future conditions, which drive more transport from non-point sources (e.g. manure application, livestock grazing), show increases in FC loads. Loads typically decrease under drier futures; however, higher mean FC concentrations and more recreational water quality criteria exceedances occur, likely caused by reduced flow during low-flow periods. Median changes across the ensemble generally show increases in FC load. BMPs that focus on key fecal sources (e.g., runoff from pasture, livestock defecation in streams) within a watershed can mitigate the effects of hydroclimatic change on FC loads. However, more extensive BMP implementation or improved BMP efficiency (i.e., higher FC reductions) may be needed to fully offset increases in FC load and meet water quality goals, such as total maximum daily loads and recreational water

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quality standards. Strategies for managing climate risk should be flexible and to the extent possible include resilient BMPs that function as designed under a range of future conditions.

Keywords

Climate; Management responses; Microbial water quality; Watersheds; Modeling

Introduction

Waterborne pathogens sourced to fecal waste (including bacteria, viruses, and protozoa) present a risk to human health (through recreation and ingestion) and are commonly identified as the leading cause of waterbody impairments in the U.S. (Coffey et al., 2014; Coffey et al., 2018; Korajkic et al., 2018; Pandey et al., 2018) Approximately 300,000 kilometers of rivers and streams are currently considered impaired (Liao et al., 2016; USEPA, 2019; Wade et al., 2003), however, the actual number of impairments is likely much higher than reported as not all ambient waterbodies are monitored regularly for microbial quality (Pandey et al., 2018; Pandey et al., 2014). Key contributors to recreational water quality impairments are often sewage and agricultural runoff, which are a pervasive problem in many watersheds (Dila et al., 2018).

Fecal indicator bacteria (FIB), such as fecal coliform (FC) and *Escherichia coli* (*E. coli*), are typically measured as a proxy for pathogenic bacteria to assess whether waterbodies can support contact recreational uses (Jeong et al., 2019). When a waterbody is deemed impaired due to excessive levels of FIB (under section 303 d of the U.S. Clean Water Act), a total maximum daily load (TMDL) may be required to restore water quality to support recreational uses (Arnone & Perdek Walling, 2006; Gilfillan et al., 2018; Pandey et al., 2014).

The occurrence of pathogens in waterbodies is influenced by precipitation, which drives transport from upland fecal sources (e.g. humans, livestock and wildlife), and numerous hydro-climatic factors affecting survival (e.g., temperature, ultraviolet radiation, moisture, sediment interactions, pH, nutrient availability and salinity) (Bicudo & Goyal, 2003; Manyi-Loh et al., 2016; Pachepsky & Shelton, 2011; Vermeulen & Hofstra, 2014). FIB loads often strongly correlate with precipitation and runoff which drive more transport from upland sources (Cho et al., 2010a; Dila et al., 2018; Haack et al., 2016; Hong et al., 2017; Lafforgue et al., 2018; Wang et al., 2018). If realized, anticipated future hydroclimatic changes (e.g., more heavy precipitation events and warmer temperatures) could increase the occurrence of fecal pathogens in waterbodies (Hofstra, 2011; Iqbal et al., 2017; Islam et al., 2017; Levy et al., 2016; Vermeulen & Hofstra, 2014) and jeopardize future efforts to meet microbial water quality goals such as TMDLs and recreational water quality criteria (RWQC) (Coffey et al., 2014; Coffey et al., 2018).

Efforts to restore impaired waterbodies usually involve best management practices (BMPs) aimed at reducing the sources and delivery of FIB and other contaminants such as nutrients and sediment (Liu et al., 2017). Many of the BMPs create substantial health co-benefits by improving the safety of water-contact recreation and the ecological condition of waterbodies

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(Richkus et al., 2016). However, the performance of BMPs under future climate conditions is not clear, as estimates of benefits typically are based on past performance (Schmidt et al., 2019). For instance, more intense precipitation can increase transport of fecal waste from agricultural land, increase leaching through subsurface pathways, and reduce contact time in practices that rely on filtration (Coffey et al., 2018). More heavy precipitation could also overwhelm BMPs like riparian buffers (Wagena & Easton, 2018). Future impacts on water quality will depend on the performance of BMPs that may not have been designed to cope with anticipated pressures due to increased precipitation volume and intensity (Paul et al., 2018).

Managing the risk of future impacts therefore requires an understanding of interactions between fecal sources, hydroclimatic conditions and BMP effectiveness at spatial scales relevant to decision makers. The objective of this paper is to better understand the range of potential impacts and build capacity for managing the risk of climate-driven changes in microbial water quality. We used a watershed-scale, process-based, lumped-parameter model, the Hydrologic Simulation Program FORTRAN (HSPF; Bicknell et al., 2005), together with a fecal source characterization tool, to quantify potential FC responses to future climate scenarios and assess the effectiveness of commonly used BMPs to moderate impacts in two watersheds – the Chippewa watershed in Minnesota and the Tye watershed in Virginia. HSPF has been widely applied to quantify FIB TMDLs, and assess the impact of global environmental changes on water quality (Benham et al., 2006; Cho et al., 2016; Coffey et al., 2015; Fonseca et al., 2015; Iudicello & Chin, 2013; Liu et al., 2010). While other studies have examined this problem, there are relatively few in the scientific literature. In addition to projecting future changes, this study is novel in that it also considers the sensitivity of BMP pollutant reduction performance to hydroclimatic change, and the ability the meet recreational water quality goals through management responses. We address the following questions:

1. What are the net impacts of anticipated future hydroclimatic changes on in-stream FC?

2. How will future hydroclimatic changes affect current BMP based strategies for meeting water quality goals?

This study only considers the effects of changes in hydroclimatic conditions. Inclusion of future changes in land use, human activities and other factors affecting upland fecal sources is beyond the scope of this effort. The focus of modeling is on FC rather than *E. coli* [FC were replaced by *E. coli* as the recommended indicator for recreational freshwater in 1986 – see (USEPA, 2012)] as more comprehensive source characterization data was available for FC.

Methods

Study Areas

The Chippewa River watershed in Minnesota (Figure 1) is approximately 5400 km^2 in area and is a major tributary of the Minnesota River (MPCA, 2006). It is situated on the edge of the Corn Belt and contains a mix of agricultural land uses (roughly 75% of total land area)

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and waterbody types (MPCA, 2017). Cultivated crops (primarily corn and soybeans) dominate the southern part of the watershed, while, to the north, there is an increasing mixture of pasture, grassland, and forest, with numerous lakes (LaMotte, 2016). Much of the cropland is ditched and tile-drained. The geomorphology includes a complex mixture of moraines, and till, lacustrine, and outwash plains. The climate is continental, with cold dry winters and warm wet summers. Average monthly temperatures at Benson, MN (located within watershed) range from −12 °C in January, to 23 °C in July. An average of 635 millimeters of precipitation falls in the watershed annually, with approximately 66% of the precipitation occurring from May to October. Annual runoff is estimated to range between 51 to 102 millimeters (spatially) (MPCA, 2017).

The Tye River watershed in Virginia (Figure 1) is approximately 1100 km^2 and is part of the James River Basin which flows east to the Chesapeake Bay. Land use in the watershed is dominated by forest (75%), with significant areas of streamside pasture and hay (15%). Residential areas compose a small portion of the watershed (6%) (LaMotte, 2016; USDA-NASS (USDA, 2009). The most dominant soil group is Clifford loam. Northern and eastern parts of the watershed are mountainous elevations (as high as 900 to 1000 meters) (Benham et al., 2013). The climate is considered warm oceanic/humid subtropical based on Köppen classification, with warm summers and moderately cold winters. Long-term climate data at Montebello station in the north of the watershed shows an average annual precipitation of 1277 mm, with 53% of the precipitation occurring during the growing season (May-October). Average annual daily temperature is 11 °C, with the highest average daily temperature of 22 °C occurring in July, and the lowest average daily temperature of 1.5 °C occurring in December (SERCC, 2012).

Model Development

Fecal Source Characterization—FC loadings to streams in agricultural watersheds results almost exclusively from livestock manure, wildlife and improperly installed or maintained septic systems (Lenhart et al., 2017). We used available information (e.g., state reports, census data, national land cover database, national agricultural statistics service, etc.) about FC sources in both study watersheds to estimate potential loadings (Benham et al., 2013; MPCA, 2006). The information was input to the bacteria source load calculator (BSLC) which helps automate the quantification of FC loading and provides consistency in data development for HSPF (Zeckoski et al., 2005). Information such as land use distributions and livestock/wildlife/human population estimates are inputs to the BSLC and used to generate non-point source (NPS) FC loadings as a monthly variable and direct FC discharges as an hourly variable (Brown et al., 2014; Zeckoski et al., 2005). Table 1 outlines some of the key factors considered in the BSLC when estimating FC loading to agricultural land and streams.

Estimated annual FC loading contributions from key sources are given Table 2. The source characterization identified livestock grazing on pasture, manure application, wildlife and failing individual sewage treatment systems (ISTS) as the main non-point source fecal loadings (transported to water channel via surface runoff) in both watersheds. Key direct sources of fecal loading (deposited directly to streams) include wastewater treatment plants,

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direct discharges from residential housing, and cattle and wildlife defecating in streams. The type and timing of agricultural operations that affect FC loading (e.g., manure application, livestock grazing, winter housing, livestock hours spent in stream etc.) were also considered in the BSLC set up for each watershed. Livestock defecation directly to streams is calculated based on available data about the fraction of time spent in stream per day in each watershed (varies monthly and by watershed) and the fraction of unrestricted stream access in each watershed (as accurate data on the amount of stream access is generally unavailable, this often a calibration parameter). We used available information to quantify fecal loading from wildlife (Benham et al., 2013; MPCA, 2006); however, determining wildlife contributions is difficult due to the lack of accurate population data that exists for many watersheds (Coffey et al., 2015; Oliver et al., 2016). Additional information on source characterization can be found in the online supporting information and associated reports (Benham et al., 2013; MPCA, 2006; Tetra Tech, 2012; Zeckoski et al., 2005).

FC die-off on the land surface is affected by many interacting factors that can be difficult to fully replicate in watershed models. In this study, die-off on land is represented in the BSLC as the limit on surface accumulation of transportable FC load and is considered to follow an exponential decay (Chick's law) (Crane & Moore, 1986). With a constant accumulation rate, the asymptotic limit on accumulation as time goes to infinity is equal to the accumulation rate divided by the die-off rate. The limit value was specified monthly and independently for each land segment (Benham et al., 2006). A comprehensive explanation of the BSLC set up, inputs and calculations can be found in Zeckoski et al. 2005.

HSPF model set up and calibration—We used HSPF (Bicknell et al., 2005; Duda et al., 2001) to model hydrology and in-stream FC concentrations under historical and projected future climatic conditions. The model simulates NPS runoff and pollutant loadings, performs flow routing through streams, and simulates in-stream water quality processes. It estimates runoff from both pervious and impervious parts of the watershed and streamflow in the channel network. The fate of FC on land segments (impervious and pervious) and in stream (as dissolved pollutants) is accounted for in the model. In HSPF and other watershed modeling applications (e.g., SWAT), in-stream die-off is modeled using a temperature-corrected first-order decay function, and temperature is the only environmental variable that is used to modify die-off.

Model set up and calibration methods for the Chippewa and Tye watersheds have been described in detail in previous reports (Benham et al., 2013; Tetra Tech, 2012). The Chippewa River watershed was divided into 62 sub-watersheds (5 weather zones) and the Tye River watershed was divided into 50 sub-watersheds (1 weather zone) based on homogeneity of land use, soil type stream network connectivity and monitoring locations (flow, water quality and weather stations). Both models were calibrated and validated using observed historic weather, streamflow, and FC data within each watershed. A credible fit was obtained for various metrics of flow for the calibration and validation periods at gauging stations in both watersheds. The water quality calibration was performed at an hourly time step to calculate the simulated minimum-maximum values over a period of 5 days – the aim of the approach is for the observed FC data to fall roughly within the range of values simulated near the date of observed data sample. Calibration of in-stream FC concentrations

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was also within recommended ranges (Kim et al., 2007) (see online supporting information). Post calibration, validation results were considered reasonable representations to assess potential FC responses under different future climate and management conditions. Additional information on hydrological and water quality calibrations is provided in online supporting information to this paper.

Future Climate Scenarios

Future climate scenarios were initially screened using EPA's LASSO tool (https://lasso.epa.gov/). We considered 20 Global Climate Models (GCMs) and 2 Representative Concentration Pathways (RCPs) and chose 8 projections that capture a range of drier-warmer to wetter-warmer future conditions. The selected 8 GCM projections (from the Coupled Model Intercomparison Project Phase 5 - CMIP5) are for mid-century (30 years: 2035 – 2065), assume a "business-as-usual" greenhouse gas emissions trajectory (RCP 8.5), and are statistically downscaled in space and time using the Multivariate Adaptive Constructed Analogs (MACA) method. The MACA approach uses a collection of historical observations to scale from monthly to daily time step coupled with spatial bias correction, ensuring a reasonable representation of the temporal structure of local rainfall (Abatzoglou & Brown, 2012). MACA provides simultaneous downscaling of precipitation, temperature maximum and minimum, humidity, wind, and radiation. These outputs were acquired from the MACA website (https://climate.northwestknowledge.net/MACA/) and used to create internally consistent hourly time series of precipitation, air temperature, solar radiation, wind, and potential evapotranspiration and daily time series of dewpoint temperature and cloud cover.

GCM projections suggest that mid-century climate is likely to be wetter and warmer in the Chippewa and Tye watersheds (Figure 2). Annual precipitation changes range from small decreases (−1% to −5%) to increases >15%. In the Chippewa watershed, increases in winter, spring and fall precipitation are generally expected, with small decreases in summer. Increases in mean precipitation are projected for all seasons in the Tye watershed; however, scenarios range from wetter to drier futures depending on the GCM. Mean annual temperature is expected to increase (+1.7 ^{o}C to +4.4 ^{o}C) in both watersheds with largest increases projected in the summer. Figure 2 summarizes projected mid-century changes in annual temperature and precipitation for the Chippewa and Tye watersheds.

The developed HSPF models were forced by the 8 GCM outputs chosen for each watershed (full ensemble in Figure 2) to assess changes in streamflow and in-stream FC. FC responses to the effects of different management scenarios were evaluated for a reduced ensemble of 4 climate scenarios, capturing an appropriate sample of wetter to drier futures (along the hydrological gradient in Figure 2). This reduced the number of HSPF simulation runs required but maintained a representative selection of future climatic and hydrological conditions. Simulated changes were calculated by comparing output for two 30-year periods: a 1975 – 2005 baseline period and a mid-century period from 2035 – 2065 (centered at 2050). Results are presented as the relative simulated difference from baseline. This focuses the comparison on projected differences in climate and helps minimize the effects of any residual biases inherent in the GCM output and HSPF simulations.

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Management Scenarios

Restoration plans have been developed for the Chippewa and Tye watersheds that provide recommendations about the type and extent of BMPs necessary to reduce FIB loading and improve water quality under current conditions (Benham et al., 2013; CRWP, 2016; MPCA, 2006, 2017; VaDEQ, 2014). We used these plans as a guide to select a set of 5 BMP scenarios that focus on reductions from FC sources that have the largest impact on water quality: 1. Individual Sewage Treatment Systems (ISTS) upgrades and repairs, 2. Manure management (Manr_Man); 2. Pasture management (Pas_Man); 3. Riparian Buffers (Rip_Buf); 4. Restricted stream access (Res_Acc). A brief description of each BMP and effects on FC loading is given in the online supporting information.

BMP scenarios are represented in HSPF models using a percent removal of FC (a reduction factor) from the area of land or fecal source targeted. Implementation of individual BMPs is assumed to be the maximum extent possible, watershed-wide given the existing land use and FC sources in each study watershed. This allows comparison of the relative sensitivity of individual BMPs to hydroclimatic change. For source control BMPs, like ISTS upgrades/ repairs and restricted access to streams, it's assumed that watershed-wide implementation eliminates FC loading from the target source. For treatment control BMPs, the reduction factor used is based on average efficiencies reported in the literature (Agouridis et al., 2005; Bicudo & Goyal, 2003; Lenhart et al., 2017; Peterson et al., 2019; Richkus et al., 2016; Zeckoski et al., 2007). A summary of FC load reduction efficiencies for the five individual BMP scenarios is provided in Table 3.

The effectiveness of a hypothetical mixed BMP scenario representing a combination of different BMP (*BMP_Mix*) types is also assessed. This scenario reflects implementation plans that have been developed in each watershed to improve microbial water quality under current conditions (Benham et al., 2013; MPCA, 2017). The plans propose feasible FC loading reductions, mainly through ISTS upgrades/repairs, manure management, pasture management, and restricted stream access. Riparian buffers were not suggested as a priority BMP to reduce FC loading in either watershed plan. Table 4 outlines the net FC load reduction factors used for each *BMP_Mix* scenario. In practice, water quality management plans consider readily implementable opportunities and the location of key fecal sources when siting BMPs.

Results

Streamflow Responses to Future Climate

Figure 3 displays simulated annual and seasonal streamflow changes relative to the 30-year baseline (1975 – 2005) in the study watersheds. In the Chippewa, small increases in average annual flow (multi-model median) together with changes in seasonality occur in response to 8 mid-century climate scenarios. Increases in winter and spring streamflow are projected for most climate scenarios, which likely represents a shift to more winter rainfall and less snow. Small decreases in summer and fall streamflow are suggested based on median values (multi-model). In the Tye watershed, increases in annual streamflow generally occur in response to simulations driven by selected climate scenarios. Seasonally, winter and spring

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streamflow are projected to increase, while small decreases in summer streamflow are suggested (based on multi-model medians). There is a wide range of streamflow responses for fall; however, the multi-model median indicated little change (+2%) relative to the 30-year baseline. Simulated annual and seasonal streamflow changes in both watersheds largely correspond with climate projections which broadly point to wetter winter-spring conditions and drier-warmer summer conditions (see seasonal climate projections in the online supporting information). However, a range of future streamflow changes are evident (from negative to positive) with the response generally dependent on the individual GCM scenario used to force HSPF (e.g. GCMs projecting drier-warmer futures, such as MIROC-ESM, generally correlate with decreases in annual streamflow).

The 90th percentile flow (Q10) is the discharge rate which was equaled or exceeded for 10% of the simulated time period and is often used as a high flow metric. High flow events typically occur in response to precipitation events of high intensity or longer duration, and are correlated with large increases in pollutant loading from upland sources (Coffey et al., 2018). Figure 4 shows simulated future changes in Q10 flow conditions for the Chippewa and Tye watersheds. A wide range of responses are evident with the direction of change dependent on the GCM scenario simulated. In both watersheds, Q10 flows increase under wetter-warmer scenarios (e.g., GFDL-ESM2G) but decrease for drier-warmer scenarios (e.g., MIROC-ESM). Multi-model medians suggest that Q10 flow rates are more likely to increase in the future.

Fecal Coliform Responses to Future Climate

Fecal Coliform Loads—Figure 5 displays simulated FC load responses to the full ensemble of climate scenarios (also see the online supporting information for tabular summaries). In-stream FC loads generally increase under simulated future conditions; however, there is a wide range of responses. In the Tye watershed, the multi-model median increases annually and seasonally. Similarly, in the Chippewa watershed, increases in average FC load are projected annually, and for winter, spring and fall. Decreases in FC load occur for summer (median: −10%), which is generally projected to be warmer and drier in this location. Drier-warmer futures (e.g., MIROC-ESM and CCSM4) correlate with decreased in-stream FC loads in the Chippewa watershed (see online supporting information Figures S7 and S8). Under these conditions, decreased precipitation reduces the transport of FC from land-based sources, while warmer temperatures can reduce FC survival (Coffey et al., 2014; Coffey et al., 2018). Wetter-warmer futures (e.g., GFDL-ESM2G and CNRM-CM5) drive more FC transport from NPS (e.g., manured land and livestock grazing on pasture) and are broadly associated with increases in FC load in both watersheds. Changes in FC load in both locations generally follow changes in precipitation and temperature projected by individual GCMs – see the online supporting information for more details (Figures S7, S8, S9 and S10).

High flow events are commonly associated with the transport of a large proportion of annual FC and sediment loading from upland sources to streams – this is also the situation in both study watersheds. Bed sediment agitation and stream bank erosion caused by high flow events can also resuspend FC stored in the stream channel. Figure 4 display the percent change in FC loads (relative to the 30-year baseline) for Q10 flows. FC loads (multi-model median) generally increase in the Chippewa (+30%) and the Tye (+43%) for Q10 flow conditions. Large increases in FC loads (>100%) are associated with Q10 flows under wetter-warmer scenarios (e.g., GFDL-ESM2G and CCSM4). The proportion of FC loading occurring during high flow events also increases for wetter scenarios. This suggests that more FC loading will transpire due to precipitation events of higher intensity or greater duration, which are projected to be more frequent in the future. For drier-warmer scenarios (e.g., MICROC-ESM), FC loads and the proportion of FC loads driven by Q10 flow conditions decreases when compared to the 30-year baseline.

Concentration-based Recreational Water Quality Criteria (RWQC)—From a human health risk perspective, changes in likelihood of peak contamination events, and magnitude of fecal concentrations during peak contamination events, are more relevant than changes in loads. We also examined exceedances of concentration-based RWQC for FC under different future climate scenarios. In the U.S., RWQC are used to identify waterbodies that exceed state water quality standards [section 303(d) of the U.S. Clean Water Act]. Two RWQC are generally applied: (i) a calendar month geometric mean concentration (GMC); and (ii) a single sample maximum concentration (SSMC). Table 5 outlines the RWQC for FC that have been applied in Minnesota (pertinent to Chippewa watershed) and Virginia (pertinent to Tye watershed).

Figure 6 displays the exceedance rate of applicable FC RWQC for the 30-year baseline and future periods in the Chippewa and Tye. In both watersheds, the multi-model median suggests that exceedances of the geometric mean standard and single sample maximum are likely to increase. Drier-warmer futures (e.g., MIROC-ESM, inmcm4, HadGEM2-ES265, CCSM4) are associated with increased mean FC concentrations and more RWQC exceedances. Under these conditions, lower streamflow volumes can reduce the dilution of direct loadings (e.g., direct defecation from livestock and wastewater discharges), concentrating in-stream FC levels. For wetter-warmer futures (e.g., bcc-csm1–1-m, GFDL-ESM2G, CNRM-CM5 CSIRO-Mk3–6-0 etc.) decreases in exceedance rates are evident. Although wetter conditions are generally associated with greater FC loading to streams, increased streamflow volumes often dilute FC concentrations (Benham et al., 2006; Coffey et al., 2014; Coffey et al., 2016; Fonseca et al., 2015; Senhorst & Zwolsman, 2005).

Management Responses Under Future Climate

Figure 7 displays simulated changes in FC load under historical and future hydroclimatic conditions (reduced ensemble consisting of 4 climate scenarios) for 6 management scenarios (5 individual BMP scenarios and 1 mixed BMP scenario). For the 5 individual BMP scenarios, implementation is assumed to be to the maximum extent possible, watershed-wide given the existing land use and FC sources. FC reduction factors for individual BMP scenarios (see Table 3) and the mixed BMP scenarios (see Table 4) are held constant in the HSPF models under historical and future conditions.

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Individual BMP Scenarios

Simulations generally show that individual BMP scenarios are not as effective under future hydroclimatic changes in the Chippewa and Tye watersheds. BMP scenarios that focus on treatment control (*Pas_Man*, *Rip_Buf* and *Man_Man*) are strongly influenced by the effects of increasing precipitation which mobilizes more FC from upland sources. The ability of treatment control BMP scenarios to offset precipitation-driven increases in FC loading therefore deteriorates for wetter-warmer futures (e.g., GFDL-ESM2G and MRI-CGCM3). Source control BMPs scenarios which eliminate an FC loading source (e.g., *Res_Acc*) are not as effective under future conditions, mainly because stream loads are also impacted by associated changes in flow and FC transport from other sources (e.g., non-point). However, in each watershed, BMPs that target the key fecal contributors (see contributions in Table 2), such as runoff from pasture (improved pasture management) and direct fecal discharges from livestock (restricted stream access through stream fencing), can effectively decrease FC loading and reduce the impacts of future hydroclimatic changes. Improved manure management (affects cropland loading) and ISTS upgrades (affects residential loading) had little impact in reducing FC loads, as residential and cropland sources only contribute a small proportion of the total FC load in both watersheds (see Table 2).

Figures 8 and 9 display seasonal FC load changes in response to individual BMPs and climate scenarios in the Chippewa and Tye watersheds, respectively. The growing season is particularly important as many agricultural operations occur during this period that have a major influence on FC fate and transport in agricultural watersheds. In spring, manure application to cropland and first access to pasture for grazing livestock (spring - fall) is common and the time of year often corresponds with precipitation-driven FC loading events. Simulations suggest that pasture management, riparian buffers and improved manure management can mitigate potential increases in FC loading associated with wetter-warmer future springs. During the warmer seasons, when livestock tend to spend more time in streams consuming water and cooling (riparian shading and lower water temperatures can reduce heat stress) (Coffey et al., 2015; Kay et al., 2018; Nardone et al., 2010), restricting stream access through stream bank fencing decreased in-stream FC loads for all future climate scenarios. Like annual responses, BMPs targeting critical fecal contributors each season can reduce FC loads in each watershed under future conditions; however, the individual BMP scenarios are generally not as effective compared to historical conditions. In summary, simulated responses indicate that improved BMP efficiencies (higher FC reductions) or implementation of additional BMPs (combinations) may be required to offset increases in FC load associated with future hydroclimatic changes.

Mixed BMP Scenario

In the Chippewa watershed, decreases in annual FC load for the mixed BMP scenario under future conditions range from −13% to −53% (see “*BMP_Mix*” in Figure 6). In the Tye watershed, simulations also indicated that annual FC loads decreased for the mixed BMP scenario under future conditions (−2% to −23%). The BMP combination scenarios are most effective, expressed as a percent reduction, during the growing season (spring, summer and fall); however, a wide range of responses exist for spring and fall, where increases in load correlate with wetter-warmer future projections. In both watersheds, the mixed BMP

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scenarios were least effective when HSPF was forced by wetter-warmer futures (e.g., MRI-CGCM3 in the Tye and GFDL-ESM2G in the Chippewa).

Figure 10 shows simulated RWQC exceedance rates for the BMP combination scenario under historic and future conditions. In both watersheds, the *BMP_Mix* scenario reduces GMC exceedances relative to historic and future conditions (no management). More exceedances of the SSMC RWQC occur in the Chippewa watershed (2000 cfu mL^{-1}) for simulations of the *BMP_Mix* scenario under drier-warmer futures (e.g., MIROC-ESM). This suggests that intermittently high FC concentrations may be unavoidable during periods of lower flow volume, common in the late summer and fall, which reduce the assimilative capacity of waterbodies receiving direct FC loadings (e.g., wastewater treatment discharges and direct deposits from livestock/wildlife). In the Tye watershed, the *BMP_Mix* scenario is effective at reducing the SSMC RWQC exceedance rate (decrease of 3% to 5% for the 1000 cfu mL^{-1} Virginia standard) relative to baseline and future rates.

Discussion

Anticipated future hydroclimatic changes present a risk of degraded recreational water quality due to increased fecal loading to waterbodies (Coffey et al., 2018; Hernroth & Baden, 2018; Hofstra, 2011; Iqbal et al., 2019; Jeon et al., 2019; Patz et al., 2000; Vermeulen & Hofstra, 2014). If realized, these changes could jeopardize efforts to restore and maintain waterbodies within RWQC and increase the risk of human exposure to pathogenic organisms through recreational contact (Coffey et al., 2014; Coffey et al., 2018; Hofstra, 2011; Patz et al., 2000). The impacts of hydroclimatic changes on fecal loading will ultimately depend on the effectiveness of management responses implemented to reduce impacts. This study, while specific to two small, agricultural watersheds, provides general insights regarding the range of potential impacts, and effectiveness of common management practices for reducing risk of microbial water quality impairment. Specifically:

1. What are the net impacts of future hydroclimatic changes on in-stream FC?

   Median changes across the ensemble of climate scenarios show increases in FC loads. GCM projections suggests that wetter-warmer futures are more likely in the study watersheds and model simulations indicate that these conditions will drive increases in FC loading from upland sources to waterbodies. However, RWQC exceedance rates decreased for wet futures, potentially due to more dilution associated with increased streamflow volumes.

   For drier-warmer futures in the Chippewa watershed, decreases in FC load are evident. Lower in-stream FC survival rates and decreased loading from upland sources are typically linked with these conditions (decreased runoff and low soil moisture). Simulations do, however, indicate that more exceedances of RWQC occur for drier-warmer futures in both watersheds. Drier conditions drive periods of lower streamflow volume and often concentrate FC from direct stream loadings such as direct sewer pipes, wastewater discharges and livestock direct defecation. The results broadly concur with others that have used modeling to

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assess future changes in microbial water quality (Coffey et al., 2016; Coffey et al., 2015; Fonseca et al., 2015; Jayakody et al., 2015).

2. How will future hydroclimatic changes affect current BMP based strategies for meeting water quality goals?

Individual BMP scenarios: Model simulations in these watersheds suggest that individual BMP scenarios can compensate for climate-driven increases in FC loads; however, the BMP scenarios are more effective under historic climate conditions. When evaluating whether a BMP can or cannot reduce the effects of hydroclimatic change, watershed conditions (e.g., contributions from individual FC sources) and extent of BMP implementation are important considerations. The success of BMPs in reducing FC loading therefore often varies depending on factors like the siting, land use and fecal sources. We used a simple approach and simulated what could be considered typical efficiencies (reduction factors) with the assumption that each BMP scenario is applied watershed-wide.

Implementing BMPs that address critical FC sources directs practices to factors contributing disproportionally to degradation and is a key part of meeting water quality goals. In the Chippewa and Tye watersheds, BMP scenarios targeting key FC sources reduced loading under historic and future hydroclimatic conditions. Improved pasture management and restricting livestock access to streams with fencing are easily implementable measures that were successful in reducing loading for a range of futures (wetter to drier). Limiting access to waterbodies is considered a critical measure as fresh fecal matter is not subjected to environmental conditions that can reduce FC survival (e.g., sunlight, desiccation) (Crowther et al., 2003; Tilman et al., 2011; Zeckoski et al., 2007).

The modeling results also suggest that BMPs targeting key FC sources at critical times of year can reduce vulnerabilities for a range of potential future hydroclimatic changes. For example, the growing season (spring through fall) represents a period of increased risk as many agricultural activities influencing FC loading occur (e.g., livestock grazing, manure application) and it is also the most active time of year for recreational water use in many locations. In spring and fall, improved pasture management and riparian buffers reduced FC loading from upland sources under historic and future hydroclimatic conditions (relative to the 30-year baseline). Similarly, restricting access to streams was most effective during the warm weather season from spring through fall (historic and future) when livestock have access to pasture and tend to spend more time in streams and riparian zones.

Mixed BMP Scenario: Results for the mixed BMP scenario (*BMP_Mix)* indicate the relative ability/feasibility of managing potential future impacts using a typical, current management response. Results show wetter-warmer futures mobilize and transport more FC from upland sources, reducing the mitigation effects of *BMP_Mix* scenario in both watersheds. These responses suggest that more extensive implementation of BMPs may be required to offset increases in FC load associated with future hydroclimatic changes.

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Concentration-based RWQC exceedances (relative to the 30-year historic baseline) decreased in response to the *BMP_Mix* scenarios for most future climate scenarios; however, the *BMP_Mix* scenarios were less effective in reducing RWQC exceedances for simulations of drier-warmer futures. During drier conditions, constant direct loadings from sources not targeted in the *BMP_Mix* scenario, like wastewater treatment plants and wildlife direct defecation, may cause more RWQC exceedances due to reduced flow during low flow periods. The simulated *BMP_Mix* scenarios only focus on direct loading reductions from residential (no direct sewage discharges from rural dwellings) and livestock (no direct defecation due to stream access) – efforts to manage these fecal sources are often more effective and easier to accomplish in agricultural watersheds.

In this study we applied a simple BMP approach based on reduction factors; however warming air temperatures and changing precipitation patters will affect pollutant loading to BMPs and their function/performance. Instream microbial water quality will also be influenced by climate driven changes in flow (e.g. dilution or concentration of pollutant inputs). The success of restoration actions will ultimately depend on the type and magnitude of future changes that occur in different regions, and the type and number/extent of management responses implemented. Managers should focus on strategies that provide resilience to a range of potential future conditions when considering the type and extent of BMPs necessary to attain recreational water quality goals.

Future Research

This work builds upon current understanding about future climate-driven microbial water quality responses and assesses the effectiveness of management responses to mitigate potential impacts. While the developed models appear to provide a reasonable representation of potential future responses, modeling FC fate and transport is particularly challenging as many of the factors (e.g., temperature, ultraviolet light, moisture and nutrient availability) are difficult to accurately represent in HSPF and other models. Areas where advances would improve FC modeling have been described in other studies (Baffaut, 2010; Benham et al., 2006; Cho et al., 2016; Hofstra et al., 2019; Oliver et al., 2016) and we do not provide a comprehensive discussion of all potential limitations in this article. With more extreme weather events (heavy precipitation, periods of drought and heat waves) projected, the inclusion of the following are important in the context of future conditions:

a. Sediment related processes: Sediment interacts directly with microorganisms through adsorption/desorption – these processes are extremely important in governing the mobility of microorganisms (Cho et al., 2010b; Kim et al., 2010; Pachepsky et al., 2006; Pachepsky & Shelton, 2011; Pandey et al., 2018). Pathogenic bacteria can survive for up to several months in the sediment reservoir (which provide favorable conditions), presenting a risk of resuspension in the water column (Pachepsky & Shelton, 2011; USEPA, 2001). Projected future increases in heavy precipitation events and associated high streamflow are expected to increase the frequency of pathogen resuspension in bed sediment (Coffey et al., 2014; Coffey et al., 2018; Hofstra, 2011). Given the importance of

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streambed sediment as a FC source and the potential for more resuspension events, it is important to be able to incorporate these processes into existing models (Cho et al., 2010b; Kim et al., 2010; Pandey et al., 2018; Pandey & Soupir, 2013).

b. Low flow conditions: Longer and more frequent summer dry periods are expected to drive extended periods of low-flow conditions in many locations. When developing models in under drought conditions, simulations often lead to suspiciously high in-stream bacteria concentrations due to inputs from direct stream loadings (Benham et al., 2006; Hyun Seong et al., 2013). In such instances, model outputs are often filtered out or modified to provide a more realistic representation of in-stream concentrations. Improving methods and capabilities to model processes surrounding low-flow conditions would reduce the likelihood of erroneous simulations during drought conditions. Recent studies have also suggested that fecal organisms can be released into the stream water column through hyporheic exchange during low flow conditions (Pachepsky et al., 2017; Park et al., 2017; Stocker et al., 2016).

c. Quantification of FC survival: Anticipated future hydroclimatic changes are likely to affect many of the factors influencing survival. Most waterborne pathogens sourced to fecal waste can survive for long periods, or even re-grow, in different environmental matrices (e.g., soil, manure, and water) when conditions are favorable (e.g., low temperatures, no ultraviolet radiation, appropriate moisture and nutrients) (Cho et al., 2016; Manyi-Loh et al., 2016; Pachepsky et al., 2006; Pommepuy et al., 1992; Tyrrel & Quinton, 2003; USEPA, 2013). However, in most watershed models, survival is represented by a single dependence on temperature (Chick’s law), and re-growth is often not accounted for (Cho et al., 2016). More accurate survival relationships that account for other factors in addition to temperature are needed to provide better representation of fate and transport processes (Benham et al., 2006; Brouwer et al., 2017; Cho et al., 2016).

d. Subsurface FC contributions: Hydroclimatic changes are likely to affect FC contributions via subsurface transport pathways. Available watershed-scale models typically include only very basic representations of both subsurface hydrologic processes and associated FC contributions (Benham et al., 2006; Cho et al., 2016; Oliver et al., 2016) . This remains a relatively unexplored area of research and advances in subsurface simulation capabilities could address existing limitations.

e. Land-use change: microbial water quality is highly sensitive to interactions between climate, land use, and management (e.g., agricultural production systems) (Coffey et al., 2014). However, the combined effects of future changes in these factors are seldom considered in modeling studies (Paul et al., 2018). Integration of potential future changes in land use and management can provide further insight for decision makers, such as quantifying importance of key drivers

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(e.g., climate or land use). This could enable the development of more robust management responses based on relative vulnerability.

f. Management responses: additional research is needed to better understand and model physical/functional changes in BMPs performance. Most modeling studies typically use simple approaches to simulate BMP practices (e.g., reduction factors); however, BMP function can be affected by numerous interrelated factors (e.g., precipitation intensity, plant growth etc.). Advances which examine how BMP performance changes over time and in response to different weather events (e.g., more extreme precipitation) are needed to better inform decision makers. Improving model representation of BMP function and sensitivity to changes in climate can provide a more complete understanding of vulnerabilities.

Conclusions

This study contributes to a growing understanding of potential future hydroclimatic changes on microbial water quality and the effectiveness of management responses for reducing the risk of water quality impairment. Simulation results in two small, agricultural watersheds, the Chippewa (MN) and Tye (VA), suggest increased FC loads in response to anticipated, mid-century hydroclimatic changes. Wetter-warmer futures typically lead to increases in FC loads due to greater transport from upland sources to waterbodies. Drier-warmer futures generally lead to decreases in loading but can result in more exceedances of concentration-based RWQC due to decreased flow volumes during low flow conditions. Simulated management scenarios suggest the sensitivity of different BMPs to anticipated changes in hydroclimatic conditions (expressed as changes in percent pollutant reduction) and indicate the general ability/feasibility of managing future climate risk with commonly implemented BMPs. All BMP scenarios evaluated showed performance sensitivity to future hydroclimatic change. BMPs targeting the key sources of fecal pollution, such as runoff from pasture (improved management) and direct discharges from livestock (restricted stream access through stream fencing), were least sensitive to changes, reducing in-stream loads under a range of conditions (wetter to drier). The success of efforts to attain recreational water quality goals will depend on the future conditions that emerge, and the resilience of management actions implemented in watersheds. Management actions to reduce risks should focus on resilient BMPs that function as intended under a range of plausible futures, be flexible, and be easily extended over time as needed.

Online supporting information

The following additional information supporting this paper may be found online: (1) Summary of FC source characterization for the Chippewa watershed (doi: 10.6084/m9.figshare.11868372) (2) Summary of FC source characterization for the Tye watershed (doi: 10.6084/m9.figshare.11868375) (3) Additional information (doi: 10.6084/m9.figshare.11868477) about (a) Model calibration and validation results; (b) Projected changes in air temperature and precipitation for each watershed; (c) Simulated streamflow responses to future climate scenarios in each watershed; (c) Simulated FC responses to future climate scenarios in each watershed; (d) Simulated responses to individual BMP

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scenarios in each watershed; (e) Simulated responses to a mixed BMP scenario in each watershed.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgements

The study could not have been completed without help from many sources. The authors thank numerous colleagues at USEPA Office of Research and Development, and Office of Water whose thoughtful comments and feedback were invaluable to completing this study. The dataset MACAv2-METDATA was produced with funding from the Regional Approaches to Climate Change (REACCH) project and the Southeast Climate Science Center (SECSC). We acknowledge the World Climate Research Programme's Working Group on Coupled Modeling, which is responsible for CMIP, and we thank the climate modeling groups for producing and making available their model output. For CMIP, the U.S. Department of Energy's Program for Climate Model Diagnosis and Intercomparison provides coordinating support and led development of software infrastructure in partnership with the Global Organization for Earth System Science Portals. This research was supported in part by an appointment to the U.S. Environmental Protection Agency (EPA) Research Participation Program administered by the Oak Ridge Institute for Science and Education (ORISE) through an interagency agreement between the U.S. Department of Energy (DOE) and the EPA. ORISE is managed by ORAU under DOE contract number DE-SC0014664. All opinions expressed in this paper are the author's and do not necessarily reflect the policies and views of EPA, DOE, or ORAU/ORISE.

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Highlights

- Increased FC loading from nonpoint sources is associated with wetter-warmer futures.
- Drier-warmer futures reduced FC loads but caused more recreational water quality criteria exceedances.
- More extensive BMP implementation may be needed to meet water quality goals.

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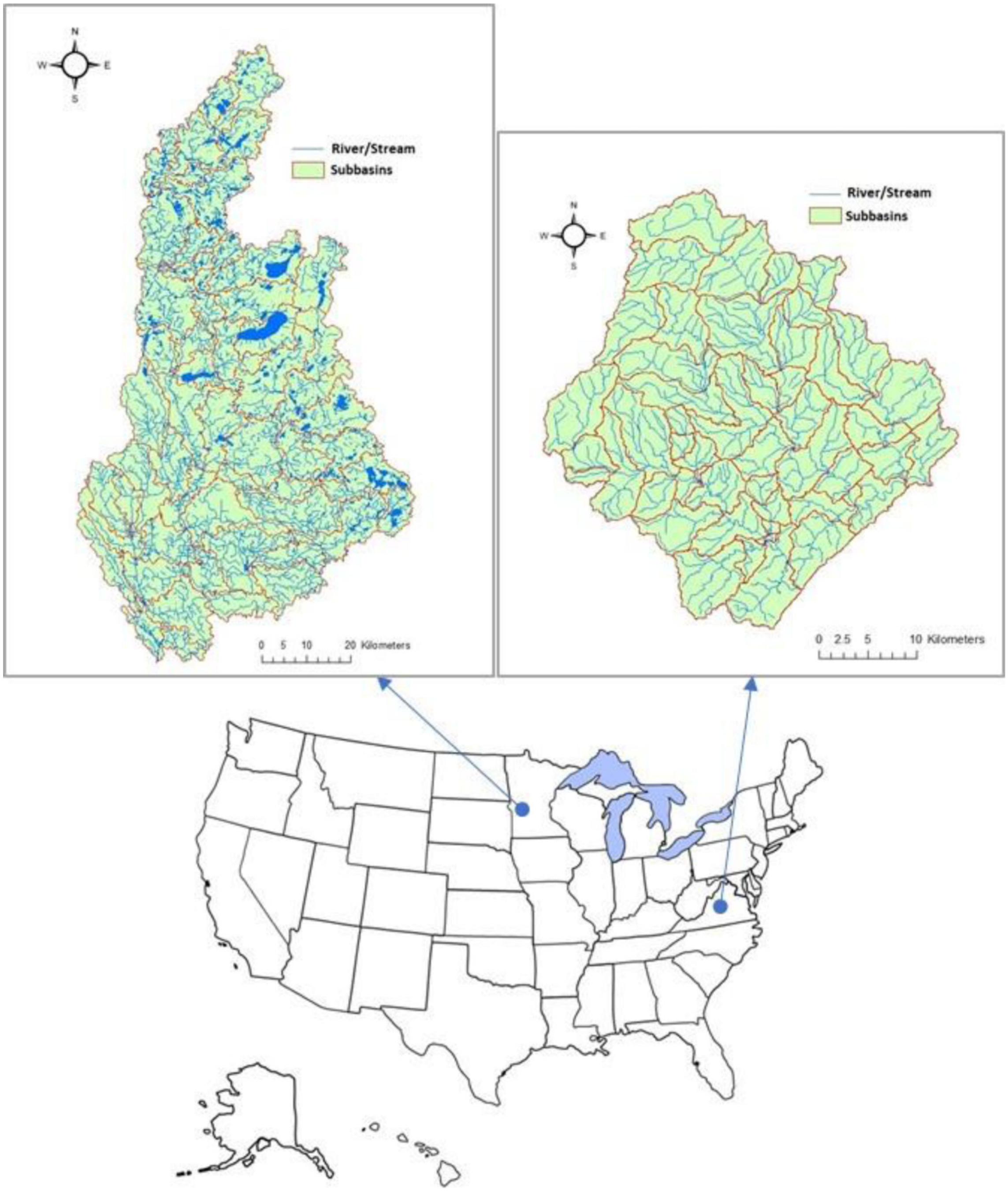

Figure 1.
The Chippewa watershed, MN and the Tye watershed, VA.

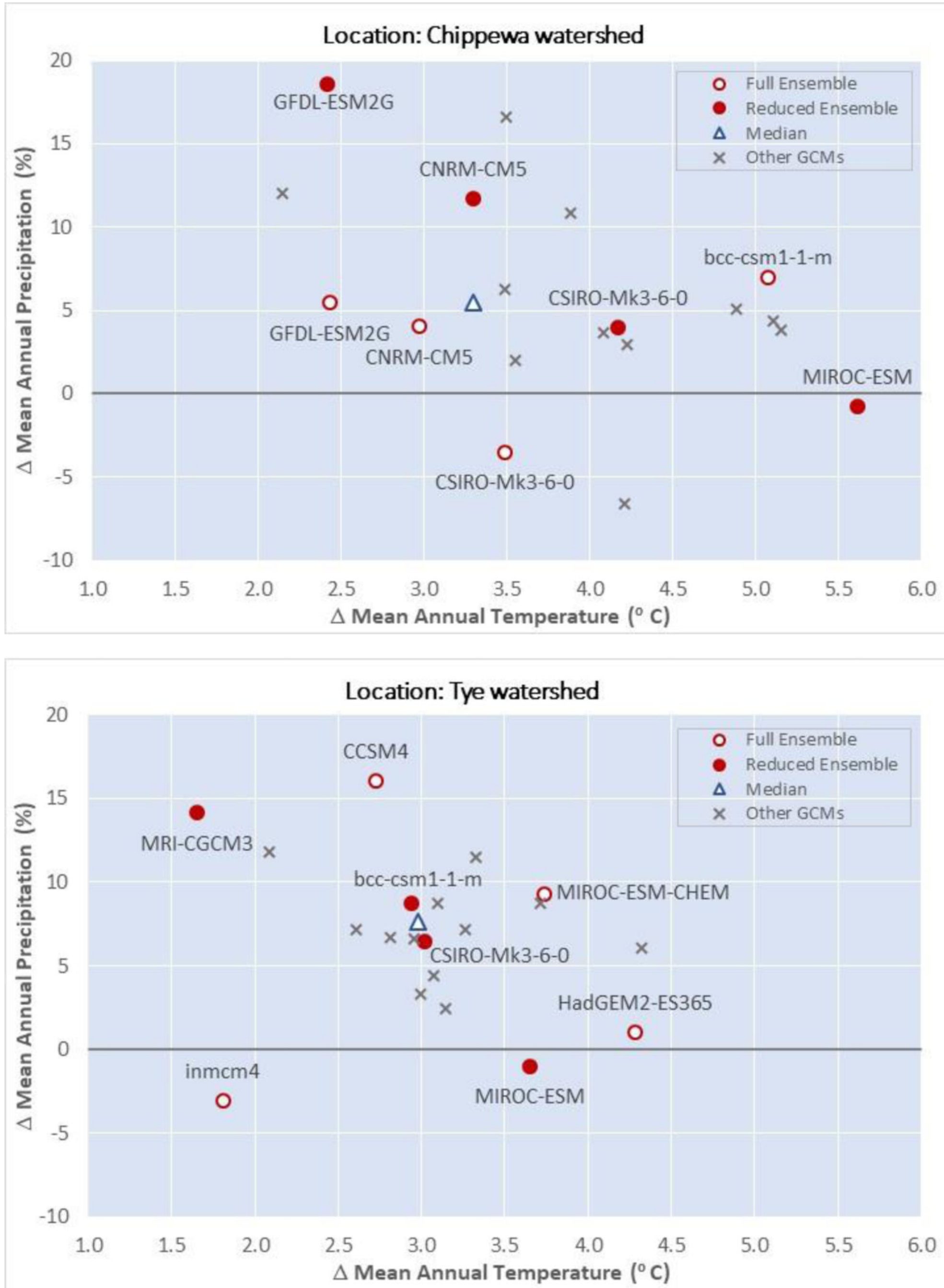

Figure 2.

A summary of projected future mid-century air temperature and precipitation changes in the Chippewa watershed and Tye watershed for 8 GCM projections (represented by circles). The “Reduced Ensemble” is used for management scenario simulations. “Median” is calculated based on the “Full Ensemble”. Changes displayed are based on data from MACAv2-METADATA and further information (e.g., full GCM names) can be found at: https://climate.northwestknowledge.net/MACA/.

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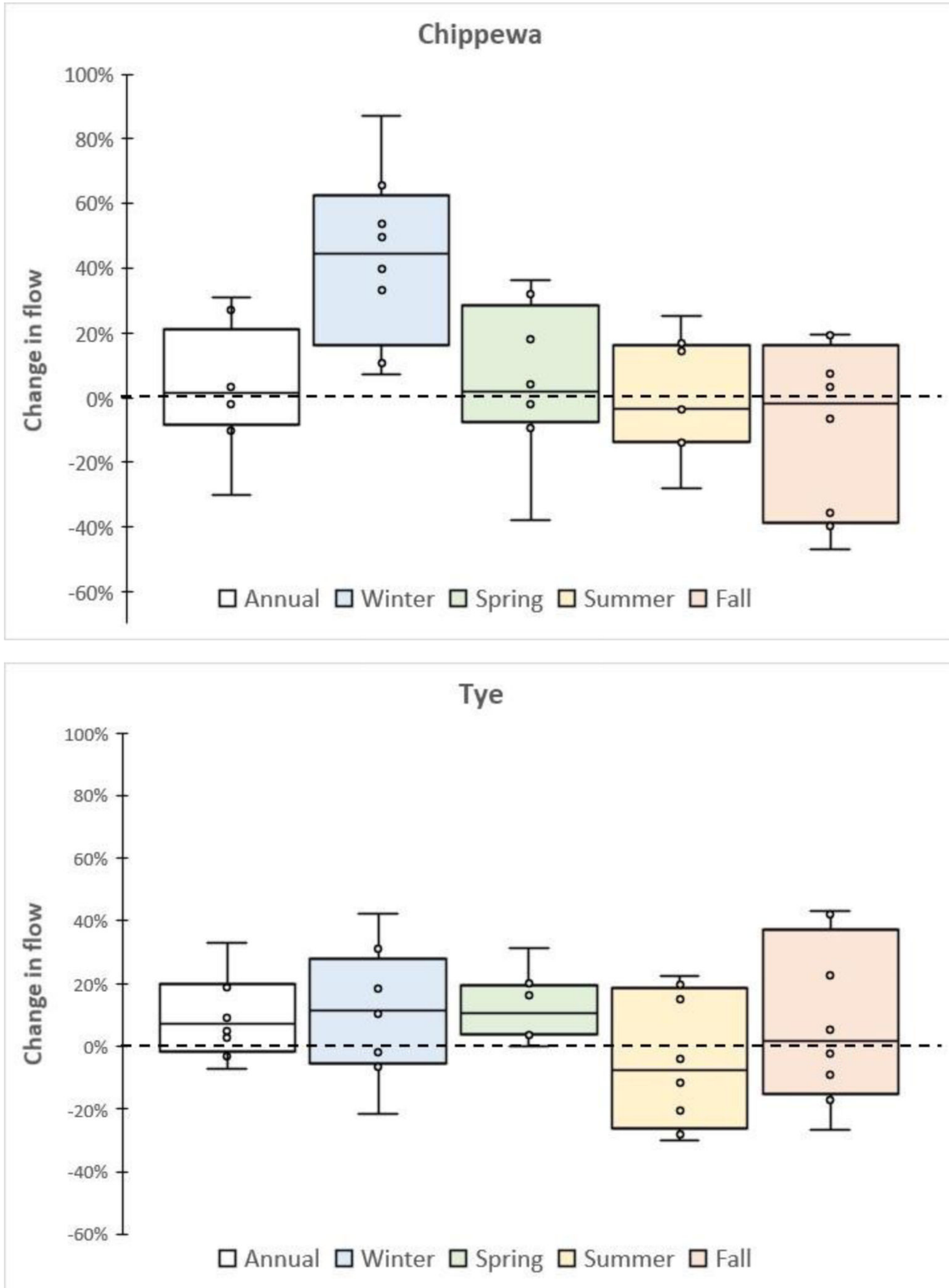

Figure 3.
Projected future changes in streamflow in the Chippewa and Tye watersheds. Percent change values are relative to the simulated 30-year baseline (1975–2005) for each model. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Additional information about projected streamflow changes is provided in the online supporting information. Winter: Dec, Jan, Feb; Spring: Mar, Apr, May; Summer: Jun, Jul, Aug; Fall: Sep, Oct, Nov.

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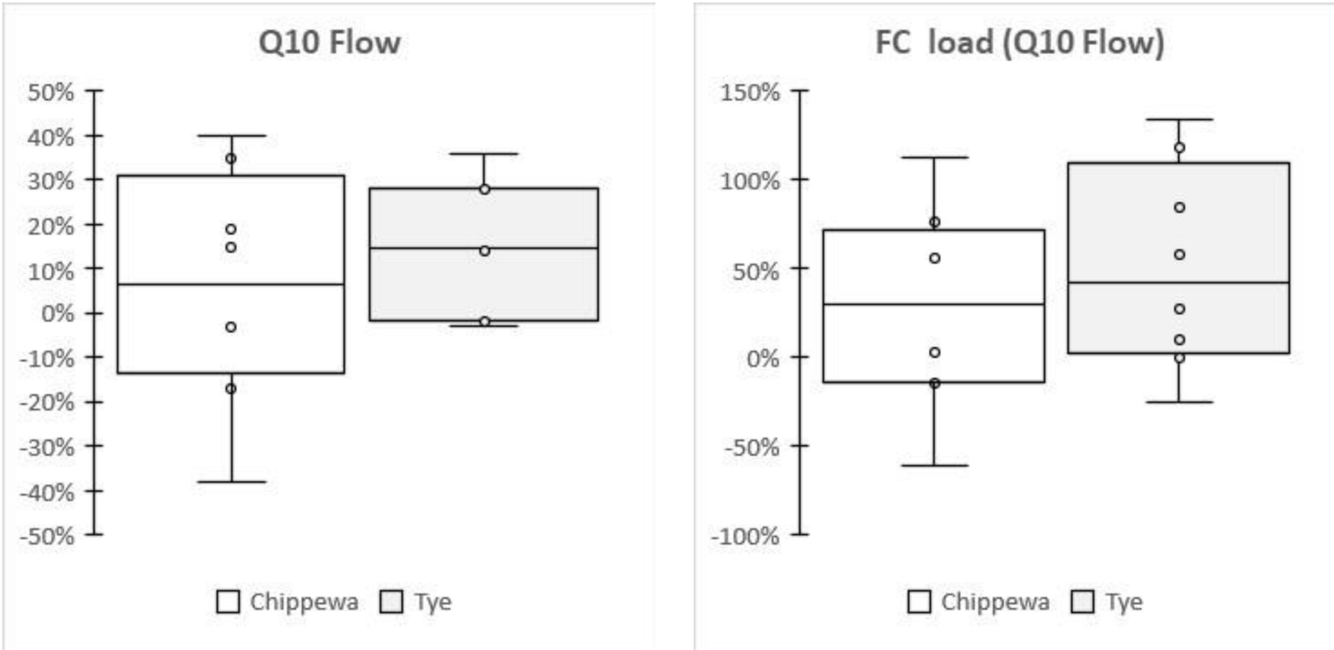

Figure 4.
Projected future changes in Q10 streamflow and FC load (under Q10 conditions) in the Chippewa and Tye watersheds. Percent change values are relative to the simulated 30-year baseline (1975–2005) for each model. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Details about changes associated with individual GCMs are provided in the online supporting information.

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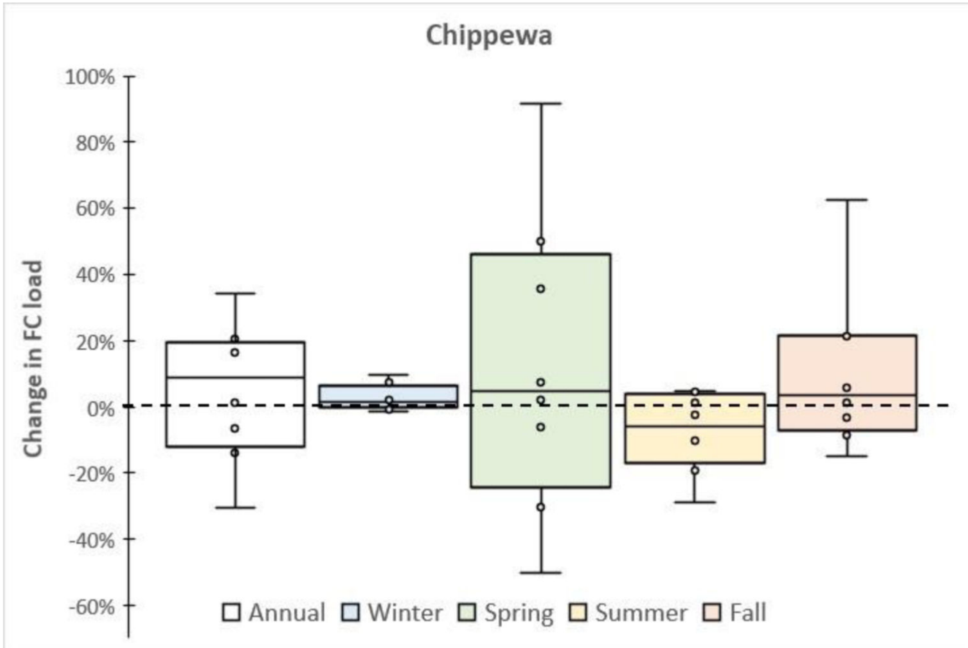

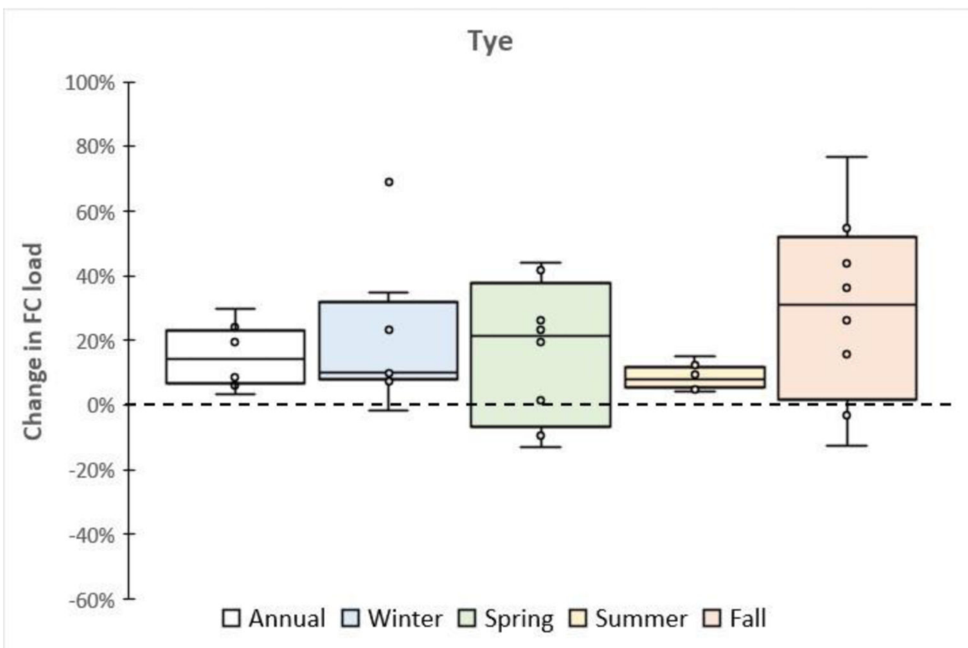

Figure 5.

Projected future changes in average FC load in the Chippewa and Tye watersheds. Percent change values are relative to the simulated 30-year baseline (1975–2005) for each model. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Details about changes associated with individual GCMs are provided in the online supporting information. Winter: Dec, Jan, Feb; Spring: Mar, Apr, May; Summer: Jun, Jul, Aug; Fall: Sep, Oct, Nov.

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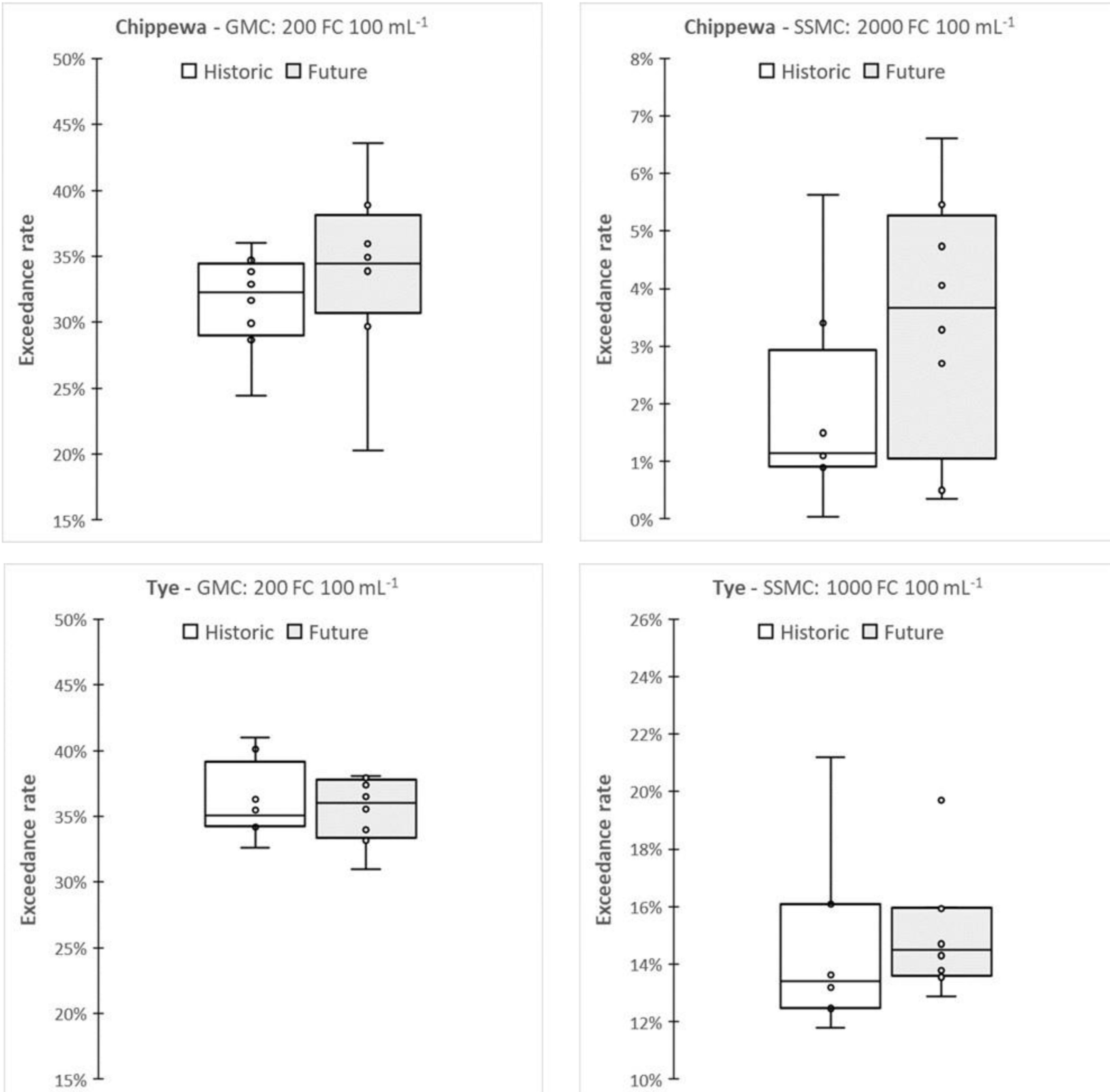

Figure 6.

Recreational water quality criteria exceedances rates for historic (baseline) and future conditions in the Chippewa and Tye watersheds. Values for historical are simulated baseline for each model, and not observed. GMC = Geometric mean concentration; SSMC = Single sample maximum concentration. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Note: Chippewa exceedance rates are for the April to October period.

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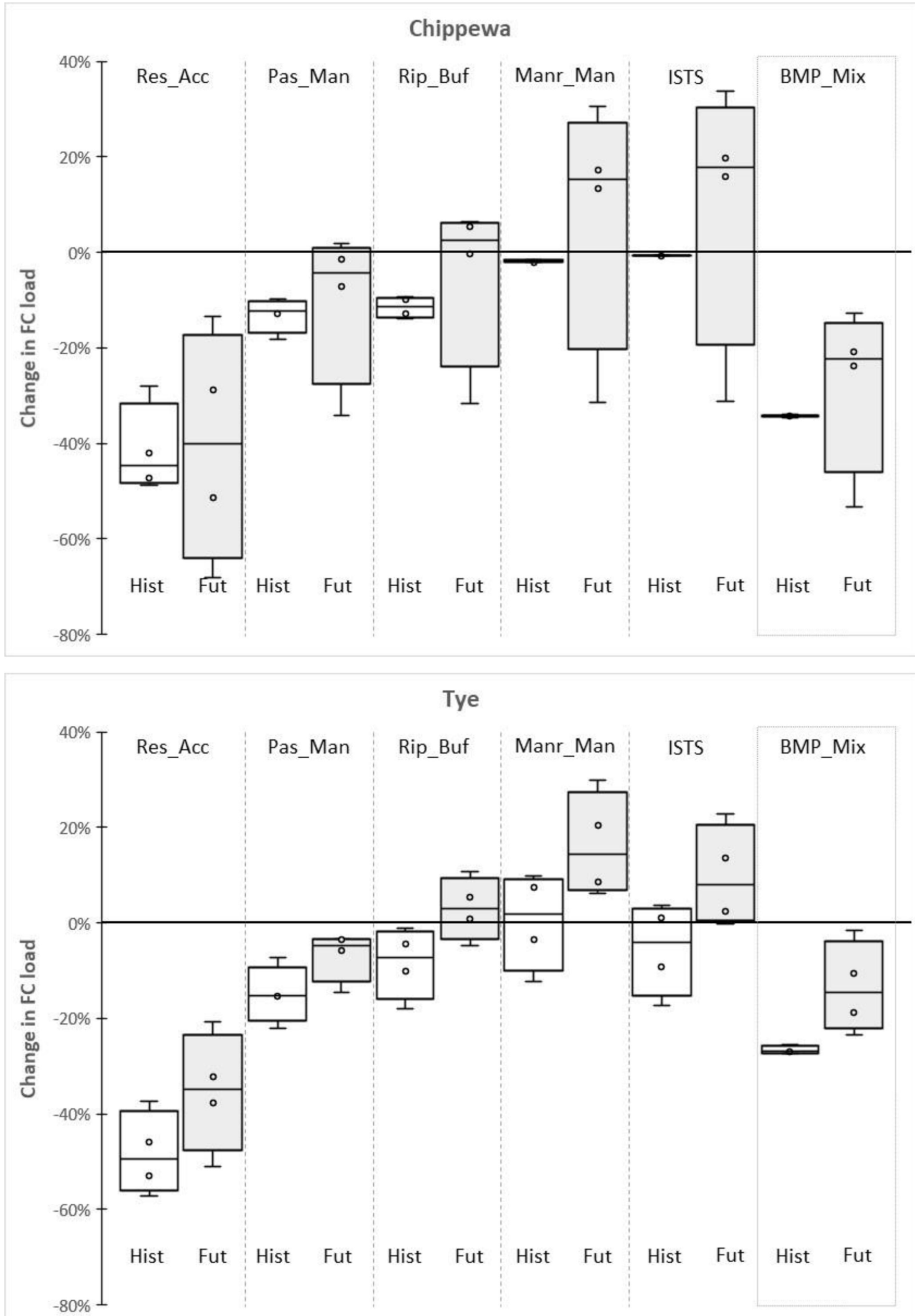

Figure 7.

Changes in annual fecal coliform load for 6 management scenarios relative to the historic baseline (1975–2005) in the Chippewa and Tye watersheds. Box and whisker plots show max and min (whiskers), 25^{th}, 50^{th} and 75^{th} percentiles (box) and individual values for each GCM (circles). Details about changes associated with specific GCMs are provided in the online supporting information. Values for historical are simulated 30-year baseline (1975–2005) for each model, and not observed.

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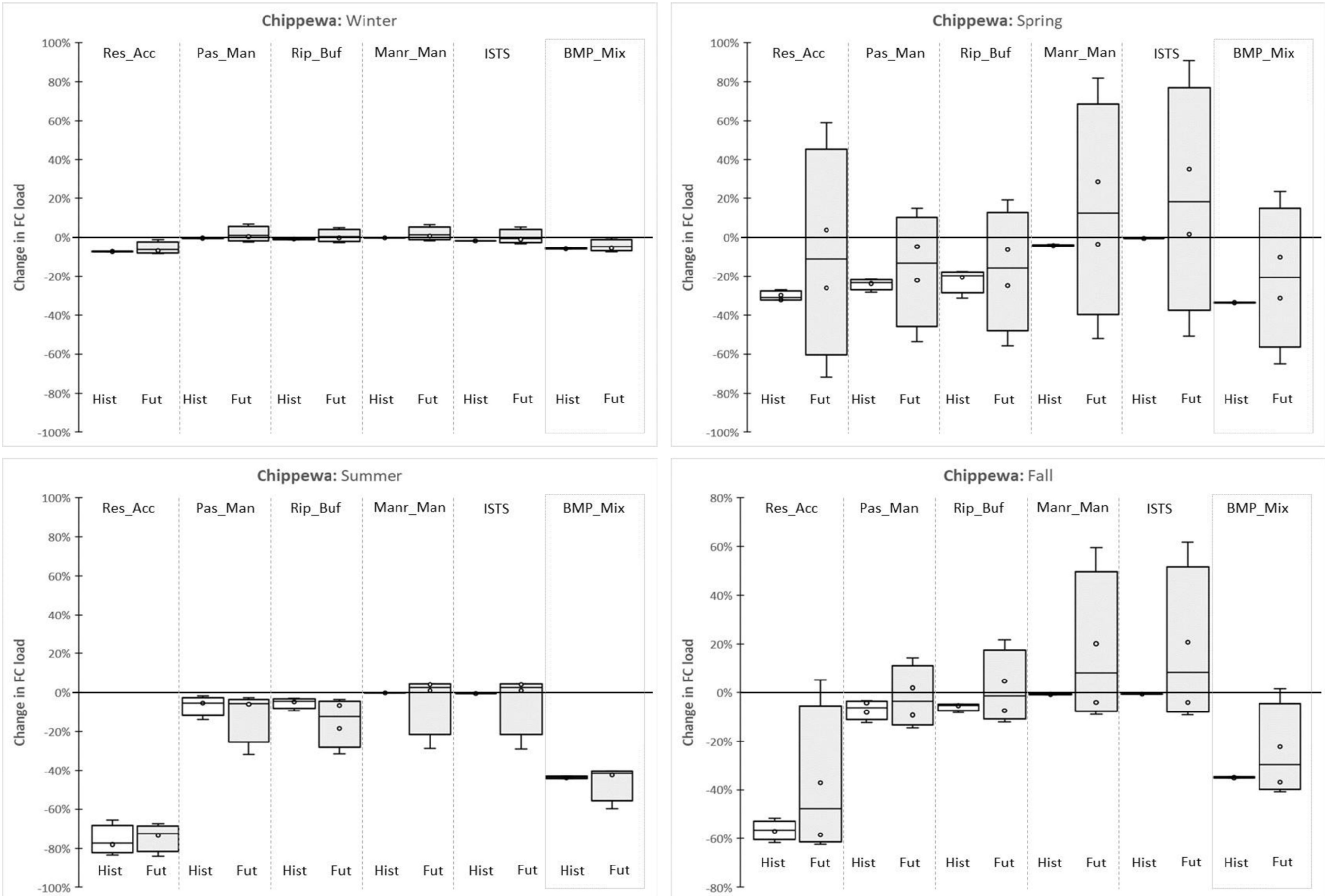

Figure 8.

Changes in seasonal (meteorological) fecal coliform load for 6 management scenarios relative to the historic baseline (1975–2005) in the Chippewa watershed. Details about changes associated with specific GCMs are provided in the online supporting information. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Values for historical are simulated 30-year baseline (1975–2005) for each model, and not observed. Winter: Dec, Jan, Feb; Spring: Mar, Apr, May; Summer: Jun, Jul, Aug; Fall: Sep, Oct, Nov.

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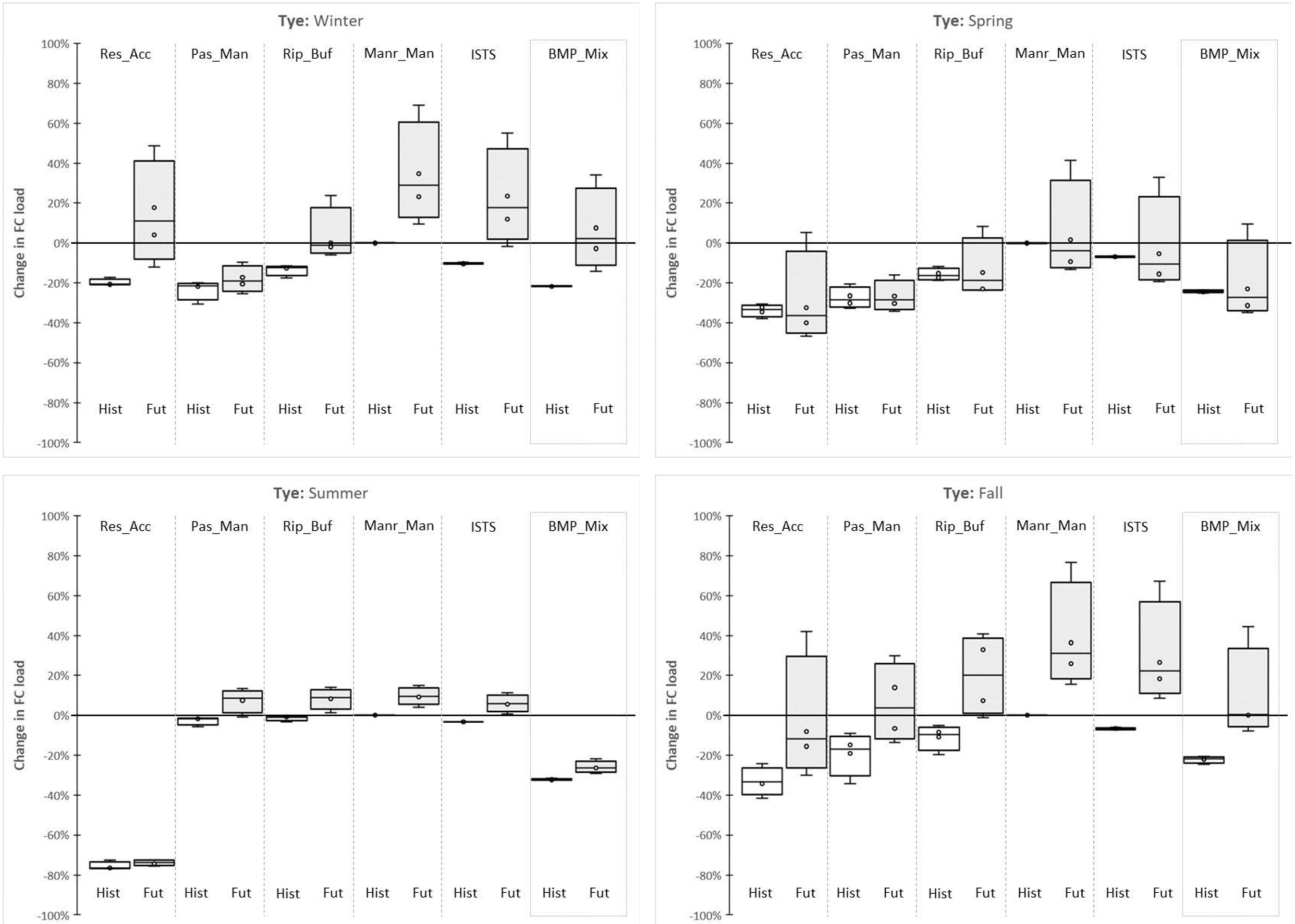

Figure 9.

Seasonal changes in FC load for 6 management scenarios relative to the historic baseline (1975–2005) in the Tye watershed. Details about changes associated with specific GCMs are provided in the online supporting information. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Values for historical are simulated 30-year baseline (1975–2005) for each model, and not observed. Winter: Dec, Jan, Feb; Spring: Mar, Apr, May; Summer: Jun, Jul, Aug; Fall: Sep, Oct, Nov.

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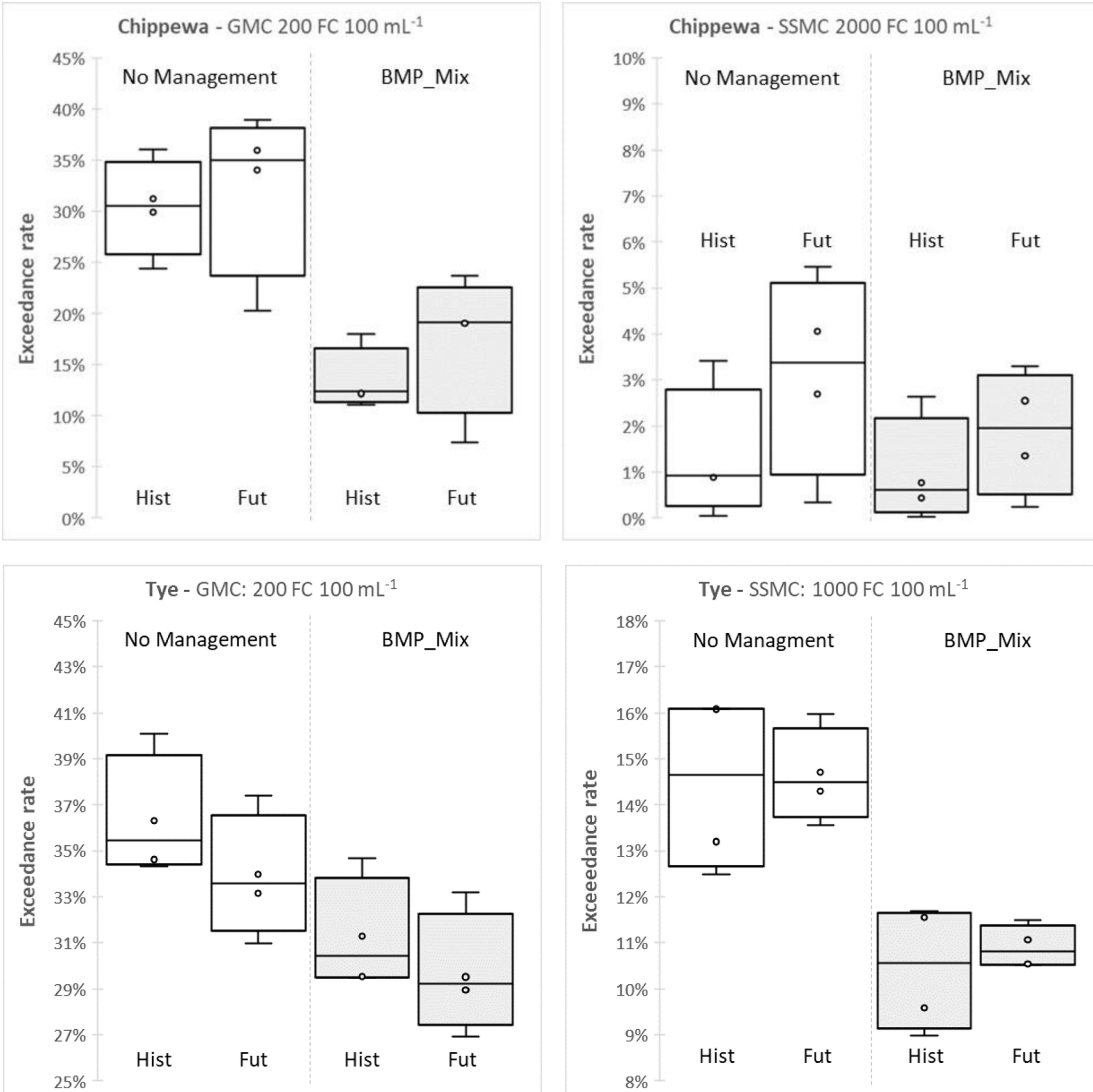

Figure 10.

Recreational water quality criteria exceedances rates for the 30-year historic baseline and future periods, and a mixed BMP scenario (historic and future). GMC = Geometric mean concentration; SSMC = Single sample maximum concentration. Values for historical are simulated 30-year baseline (1975–2005) for each model, and not observed. Box and whisker plots show max and min (whiskers), 25th, 50th and 75th percentiles (box) and individual values for each GCM (circles). Note: Chippewa exceedance rates are for the April to October period.

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Table 1.

Factors considered in determining fecal loads to agricultural land and streams (Source: Zeckoski et al., 2005).

Management Area	Determining Factors
Land receiving manure	• Number of livestock[(a)] • Percent of time livestock are confined[(a)] • Manure application rates to different land uses (default rates are those recommended for nutrient management planning) • Availability of land for manure application • Fraction of manure incorporated
Streams	• Number of livestock on pasture[(a)] • Stream access of each pasture • Time spent in and around streams[(a)] • Percent of livestock defecating in the stream
Pasture	• Number of livestock on pasture[(a)] • Fraction of time remaining livestock have been allocated to confinement or streams

[(a)] Varies monthly and by watershed (depending on local conditions)

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Table 2.

Estimated annual fecal coliform loadings to streams and from different land use categories.

FC Sources	*Chippewa Watershed*		*Tye Watershed*	
	FC loading (10^{12} cfu yr^{-1})	**% Contribution**	**FC loading (10^{12} cfu yr^{-1})**	**% Contribution**
Direct Loading to streams[1]	507	~1%	559	~1%
Diffuse Loading to land (NPS)				
Cropland	4123	5%	367	<1%
Pasture	77,799	87%	63,404	92%
Residential[2]	6353	7%	3161	5%
Forest/Other	136	<1%	1613	2%
Total	**88,918**		**68,704**	

[1]Includes discharges from points sources (including permitted WWTPs) and direct stream discharges from livestock, wildlife and residential houses without ISTS or sewer connections.

[2]Residential includes ISTS and pet waste

Note: Wildlife contributions are uniformly distributed across the watersheds. Forested loading is from wildlife only.

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Table 3.

A summary of fecal coliform load reduction factors for pasture, crop and residential BMPs.

Best Management Practice	Abbreviation	[(a)] FC reduction Factor used (%)	FC reduction factor range (%)	Reference
Source Control:				
ISTS upgrades/repairs	*ISTS*	99	5 – 99	(Richkus et al., 2016)
Restricted stream access	*Res_Acc*	99	30 – 99	(Lenhart et al., 2017; Peterson et al., 2019; Richkus et al., 2016; Zeckoski et al., 2007)
Treatment Control:				
Pasture management	*Pas_Man*	90	60 – 96	(Richkus et al., 2016)
Riparian buffers	*Rip_Buf*	51	28 – 100	(Bicudo & Goyal, 2003; Peterson et al., 2019; Richkus et al., 2016)
Manure management	*Manr_Man*	75	44 – 99	(Richkus et al., 2016)

[(a)] Used for model simulations and assumed to be implemented watershed-wide to assess BMP performance

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Table 4.

Simulated mixed BMP scenarios targeting FC reductions (net) from four key fecal sources in the study watersheds.

Watershed:	Mixed BMP Scenario (*BMP_Mix*): FC Load Reduction Targets			
	ISTS upgrades/repairs	**Manure management**	**Pasture management**	**Restricted stream access**
Chippewa	99%	10%	50%	50%
Tye	99%	5%	5–30%	70–99%

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Table 5.

Applicable fecal coliform recreational water quality criteria in Minnesota and Virginia.

State	*(a)* Geometric mean concentration (GMC)	*(b)* Single Sample maximum concentration (SSMC)	Reference
Minnesota *	200 FC 100 mL^{-1}	2000 FC 100 mL^{-1}	(MPCA, 2006)
Virginia	200 FC 100 mL^{-1}	1000 FC 100 mL^{-1}	(SWCB, 2011; VaDEQ, 1994)

(a) no exceedances

(b) no more than 10% of sample should exceed

* Apply April through October only