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## **A REVIEW OF WATER QUALITY RESPONSES TO AIR TEMPERATURE AND PRECIPITATION CHANGES 2: NUTRIENTS, ALGAL BLOOMS, SEDIMENT, PATHOGENS**

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### **Abstract**

In this paper we review the published, scientific literature addressing the response of nutrients, sediment, pathogens and cyanobacterial blooms to historical and potential future changes in air temperature and precipitation. The goal is to document how different attributes of water quality are sensitive to these drivers, to characterize future risk, to inform management responses and to identify research needs to fill gaps in our understanding. Results suggest that anticipated future changes present a risk of water quality and ecosystem degradation in many U.S. locations. Understanding responses is, however, complicated by inherent high spatial and temporal variability, interactions with land use and water management, and dependence on uncertain changes in hydrology in response to future climate. Effects on pollutant loading in different watershed settings generally correlate with projected changes in precipitation and runoff. In all regions, increased heavy precipitation events are likely to drive more episodic pollutant loading to water bodies. The risk of algal blooms could increase due to an expanded seasonal window of warm water temperatures and the potential for episodic increases in nutrient loading. Increased air and water temperatures are also likely to affect the survival of waterborne pathogens. Responding to these challenges requires understanding of vulnerabilities, and management strategies to reduce risk.

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#### **SUPPORTING INFORMATION**

Additional supporting information may be found online under the Supporting Information tab for this article: Tables describing the models and range of projected water quality changes for specific watersheds.

## Keywords

climate variability; future climate; water quality; nonpoint source pollutants; point source pollutants; watersheds

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## INTRODUCTION

Nutrients (nitrogen and phosphorus), sediment, and microbial pathogens are some of the most common pollutants impacting U.S. water bodies (Bricker *et al.*, 2008; Sprague *et al.*, 2009; USEPA, 2009, 2014, 2016). Nutrient pollution has direct effects on aquatic life and contributes to cyanobacterial blooms (potentially producing toxins), hypoxia (low levels of oxygen), and other adverse ecosystem impacts. Changes in sediment loads can alter the physical habitat necessary to support fish and other aquatic life (USEPA, 2006). Sediment is also a vector for contaminants transported to water bodies (e.g., pathogens, nutrients, and metals). Waterborne pathogens (including bacteria, viruses, and protozoa sourced to fecal waste) are a direct threat to human health (an estimated 90 million illnesses occur annually), with health-care costs attributed to some of the leading causes of waterborne diseases estimated at \$2.2 to \$3.7 billion annually in the U.S. (DeFlorio-Barker *et al.*, 2018; Collier *et al.*, 2012). The transport of these pollutants to water bodies is driven by watershed hydrologic and biogeochemical processes interacting with land use, and is strongly mediated by air temperature and precipitation (Langland *et al.*, 2003; Belmont, 2010; Jordan *et al.*, 2014; Kaushal *et al.*, 2014; Howarth *et al.*, 2006; Howarth, 2008; Dubrovsky *et al.*, 2010).

Anticipated future increases in air temperature, changing precipitation patterns and altered watershed hydrological processes associated with climate change will affect pollutant transport in specific watershed settings (e.g., physiographic and hydro climatic setting, water management and other human activities) (Georgakakos *et al.*, 2014; USGCRP, 2017; Wear and Greis, 2012). In colder/mountainous areas influenced by snow (e.g., the northern and western U.S.), warmer air temperatures affect the timing and seasonality of snow and snowmelt, including increased winter precipitation in the form of rain as opposed to snow, and snowmelt earlier in the winter and spring.

These changes are likely to alter the magnitude and timing of seasonal peak flows, including runoff peaks earlier in the spring, and decrease summer low flows (Paul *et al.*, 2019; Birsan *et al.*, 2005; Jefferson *et al.*, 2008; Ficklin *et al.*, 2013c, USEPA, 2016). Increases in air temperature also alter nutrient biogeochemical cycling, primary productivity, solubility, reaction rates, microorganism survival, evapotranspiration, soil moisture and other factors related to water quality (e.g. stratification) (Kundzewicz *et al.*, 2008; USGCRP, 2017; Coffey *et al.*, 2014; Havens and Paerl, 2015; Wear and Greis, 2012; Chang *et al.*, 2010).

Regionally variable changes in the amount and seasonal timing of precipitation and runoff are also anticipated (Melillo *et al.*, 2014). In the U.S., historically wetter regions (e.g., northern and eastern regions) are likely to receive increased annual precipitation with corresponding increases in runoff; whereas historically drier regions (the arid southwest, southern Great Plains, and parts of the southeast) will likely receive less precipitation resulting in decreased runoff (Arnell, 2003; Milly *et al.*, 2005; USEPA, 2013). In addition,

an increasing frequency of heavy precipitation events - even in areas that receive less annual precipitation - and longer dry periods between precipitation events are projected (Prein *et al.*, 2016). These changes will alter the mobilization and transport of pollutants from upland sources to water bodies (Jackson *et al.*, 2005; Salant *et al.*, 2008; Hancock, 2012; Ffolliott *et al.*, 2013; di Silvio and Basson, 2008; Gellis *et al.*, 2003; Ford *et al.*, 2011; Giorgi *et al.*, 2011; IPCC, 2012; Vose *et al.*, 2011; West, 2002). Rates of sediment, pathogen (from fecal sources) and nutrient transport typically correlate to precipitation and runoff (Jackson *et al.*, 2005; Salant *et al.*, 2008).

Reducing the risk of climate change impacts on water quality requires an understanding of vulnerabilities in different regional and watershed settings, which can provide a starting point for effective, sustainable adaptation strategies. The goals of this review are to document potential future water quality responses to climate change, to characterize future risk, to inform management responses and to identify research needs to fill gaps in our understanding. In this paper, we focus on potential future responses of nutrients, sediment, pathogens and cyanobacterial blooms. A companion paper (Paul *et al.*, in review) discusses streamflow, water temperature and salt water intrusion. We note that water quality changes are complex and affected by multiple, interacting climatic, watershed and human factors. In line with study objectives, we focus only on water quality sensitivity to air temperature and precipitation. Discussion of changes in land use, water management and other factors affecting water quality is beyond the scope of this effort.

## APPROACH

Relevant literature was identified through a search of recently published peer-reviewed and gray literature (post 2000) examining water quality responses to historic or future changes in air temperature and precipitation. Literature searches were conducted in common scientific databases (e.g., PubMed) using appropriate search terms. Our search identified studies that evaluated either the observed sensitivity of nutrients, cyanobacterial blooms, sediment and waterborne pathogens to climate drivers, or simulations of potential future changes using water quality models driven by air temperature and precipitation scenarios. Studies of water quality responses to observed weather and natural, internal climatic variability document how different attributes of water quality are sensitive to air temperature and precipitation. Modeling studies suggest potential changes in response to alternative future conditions; however, study comparisons can be complicated due to differences in the methods and models used. To the extent possible, we synthesize information from each type of study to make inferences about the risk presented by future changes. Where possible, regional differences are identified. Note that modeling studies typically provide an “ensemble” range of outcomes in response to different future scenarios and time periods etc. In such cases, we make the simplifying assumption that the ensemble mean or median suggests a “more likely” direction of change for the purposes of risk management.

## RESULTS

### Nutrients

**Observed Sensitivity.**—Changes in air temperature and precipitation are key drivers of nutrient mobilization and transport into and within water bodies. Warmer temperatures generally increase nutrient availability, solubility, and cycling in aquatic and terrestrial ecosystems (Murdoch *et al.*, 2000; Kundzewicz *et al.*, 2008; Duan and Kaushal, 2013). Rates of nitrification (Murdoch *et al.*, 2000; Kundzewicz *et al.*, 2008; Pourmokhtarian *et al.*, 2012), mineralization (Brooks and Williams, 1999; Andersen *et al.*, 2006; Hartman *et al.*, 2014), and denitrification (N loss to the atmosphere) (Murdoch *et al.*, 2000; Andersen *et al.*, 2006; Battarbee *et al.*, 2012; Boyacioglu *et al.*, 2012; Alam and Dutta, 2013) have been shown to increase with temperature. Decomposition and mineralization rates of organic nutrients in soils increase at higher temperatures (Shrestha *et al.*, 2012). The net effects of these changes determine nutrient availability, affecting transport and fate of constituents (Ryberg *et al.*, 2017; Andersen *et al.*, 2006; Boyacioglu *et al.*, 2012; Shrestha *et al.*, 2012; Alam and Dutta, 2013; Hartman *et al.*, 2014; Worrall *et al.*, 2003).

Precipitation and runoff mobilize and transport nutrients (dissolved and sediment-adsorbed) from upland sources to water bodies (Figure 1) (Murdoch *et al.*, 2000; Howarth *et al.*, 2006; Kaushal *et al.*, 2008; Kundzewicz *et al.*, 2008; Ficklin *et al.*, 2010; Wilson and Weng, 2011; Joyner and Rohli, 2013; Jiang *et al.*, 2014). When runoff encounters nutrient-rich sources (e.g., recently fertilized fields, shallow enriched groundwater, organic waste), nutrient mobilization and transport generally increases (Creed and Band, 1998; USEPA, 2013; Murdoch *et al.*, 2000; Whitehead *et al.*, 2009b). Numerous studies show increases in nutrient loading that correlate with precipitation and runoff volume (Najjar *et al.*, 2000; Noges *et al.*, 2011; Michalak *et al.*, 2013; Jiang *et al.*, 2014; Orem *et al.*, 2015; Morse and Wollheim, 2014). For example, in the Mississippi river, precipitation patterns have been suggested to be the controlling factor in nitrogen loading from land and transport through the river system as land use and cover has remained relatively constant in recent times (Donner and Scavia, 2007). During drier periods, elevated nutrient concentrations can occur downstream of point source discharges (e.g., groundwater contributions, wastewater treatment plant discharges, industrial effluent etc.) due to reduced flow volume and decreased instream dilution (Murphy *et al.*, 2014).

Several national scale studies have examined historical nutrient trends in U.S. waterbodies (Sprague *et al.*, 2009, Howarth *et al.*, 2006; Dubrovsky *et al.*, 2010; Prasad *et al.*, 2010; Hale *et al.*, 2013; Zhang *et al.*, 2013; Howarth, 2008; Sinha and Michalak, 2016; Sprague *et al.*, 2011). Stets *et al.*, (2015) report that flow-weighted nitrate concentrations (a metric which reflects loads) increased at 22 monitoring stations throughout the U.S. (8 in the East, 5 in the Midwest, 5 in the Great Plains and mainstem Mississippi River and 4 in the West) from 1945 to 1980, but stopped increasing between 1981 and 2008. However, attribution of trends to specific drivers like air temperature and precipitation is often not possible and many studies link observations to changes in land use, fertilizer application, atmospheric deposition, point source discharges and other factors. One recent study suggests that spatial variability in loading is typically dominated by nutrient inputs but that interannual variability and the

occurrence of extreme loading are driven by precipitation across much of the U.S. (Sinha and Michalak, 2016). Loecke *et al.*, (2017) showed that that changing precipitation patterns (drought-to-flood transitions), interact with agricultural land use to deteriorate water quality in the Midwestern U.S. In the Chesapeake Bay, N and P loads have been linked to changes in temperature and precipitation patterns associated with the El Niño Southern Oscillation (ENSO) and Atlantic Multi Decadal Oscillation (AMO); however, such correlations are not universally found in studies (Prasad *et al.*, 2010).

**Potential Future Changes.**—Nutrient load responses to future climate scenarios have been assessed in several watershed scale modeling studies, although geographic coverage is limited (see Figure 2 for nitrogen and Figure 3 for phosphorus). Understanding future changes is particularly challenging due to the uncertainty surrounding precipitation projections at the local scale, and interaction with future changes in watershed management and human activities influencing nutrients. Future increases in air temperature are more certain and are expected to affect nutrient mobility by extending the growing season, changing nutrient uptake by plants, influencing ET and altering biogeochemical cycling in soils. Some studies suggest that increasing denitrification, associated with warmer temperatures, will often outweigh the effects of increased precipitation and lead to lower nitrogen yields in some U.S. locations (Alam *et al.*, 2017; Panagopoulos *et al.*, 2014).

Modeling studies applying similar methods to watersheds across the nation generally project increases in total nutrient loads (median) for much of the eastern and northern U.S. (Fant *et al.*, 2017; Sinha *et al.*, 2017; Johnson *et al.*, 2015). Other watershed scale modeling studies in northern and eastern regions also suggest increased annual riverine nutrient loads (Figure 1 and 2) in response to projected increases in rainfall and runoff (Howarth *et al.*, 2006; Tu, 2009; Pourmokhtarian *et al.*, 2012; Tong *et al.*, 2012; Chang *et al.*, 2010; Wang and Kalin, 2018). However, simulations include scenarios that differ in the direction of change (negative to positive) for many watersheds, correlating with variability in precipitation projections.

Other seasonal and episodic changes in nutrient loading are also expected due to changing precipitation patterns. Increases are more likely in winter-spring when a larger proportion of annual rainfall is projected to occur. The frequency and magnitude of heavy precipitation and runoff events is expected to increase in all parts of the U.S. (Prein *et al.*, 2016). and could increase the risk of episodic high nutrient loading from nonpoint sources (e.g., manured land etc.) (Fant *et al.*, 2017; Lee *et al.*, 2016). Heavy precipitation and runoff are often associated with the flushing of nutrients that have accumulated on the land surface, especially nutrients adsorbed to sediment particles (Loecke *et al.*, 2017).

Drier and warmer conditions in parts of the Southwest, particularly in summer-fall, are expected to contribute to decreases in annual runoff and nutrient loading to water bodies (Johnson *et al.*, 2015, Ye and Grimm, 2013; Chang *et al.*, 2010). However, it is anticipated that water bodies in other regions of the U.S. could also see a reduction in flow volume during occasional summer-fall dry periods. This is likely to reduce dilution and could cause higher nutrient concentrations downstream of point source discharges. In addition, lower

summer flows are expected to result in longer water residence times and could increase the risk of eutrophication.

Anthropogenic activities, such as urbanization, deforestation, forest fragmentation and intensification of agricultural production, will interact with climate change and could increase the risk of excess nutrient loading (Tong *et al.*, 2012; Alam *et al.*, 2017; Wear, 2013). However, few studies have modeled future changes in land use and climate simultaneously to assess potential effects on nutrient loads.

**Ecosystem Level Impacts.:** Temperature and changes in precipitation associated with climate change could increase the risk of eutrophication (associated with excess nutrient loading) in parts of the U.S. and cause a cascade of adverse ecosystem impacts in water bodies. Eutrophication is associated with increased phototroph growth rates which are reflected in dense algal blooms/plant growth and linked to hypoxia and anoxia (Neilan *et al.*, 2013; Paerl and Huisman, 2008; Paerl *et al.*, 2011). For example, the consequences of eutrophication are particularly evident where the Mississippi River enters the Gulf of Mexico – the hypoxic, or “dead zone” here is the largest in the United States and was measured to be 13,080 km<sup>2</sup> in 2014 (Porter *et al.*, 2015). Increased nuisance algae and plant growth can also impact aquatic life by altering food quantity/quality, reducing physical habitat quality (e.g., disrupted visual predation and impeded movement) and ultimately influencing species composition (Feminella and Hawkins, 1995; Slavik *et al.*, 2004 Paerl *et al.*, 2006; Paerl *et al.*, 2011; Fu *et al.*, 2012; Paerl and Paul, 2012; Wang *et al.*, 2007; Evans-White *et al.*, 2009). A comprehensive description of algal blooms and associated impacts is discussed in the next section.

Nutrient enrichment of rivers and streams can also contribute to acidification in estuaries and coastal systems. Excessive coastal eutrophication causes greater respiration of fixed carbon. This increases CO<sub>2</sub>, especially in bottom waters, and causes acidification (Borges and Gypens, 2010; Cai *et al.*, 2011; Duarte *et al.*, 2013; Wallace *et al.*, 2014). Nutrient enrichment together with increasing atmospheric carbon dioxide are considered the primary contributors to increasing acidity (higher partial pressure of CO<sub>2</sub>/lower pH) in aquatic systems (Fu *et al.*, 2012; Duarte *et al.*, 2013; Wallace *et al.*, 2014). These changes influence a wide range of biochemical reactions, including many critical to the maintenance of water quality and fisheries. For example, coastal acidification is expected to influence algal species composition and primary production, causing disruptions to the food webs relying on algae and algal productivity, as well as harmful algal blooms (Fu *et al.*, 2012).

### Cyanobacterial Blooms

**Observed Sensitivity.**—The number of toxic cyanobacterial blooms, water bodies affected, and toxic species reported have all increased during the past few decades in the U.S. (Loftin *et al.*, 2016; Anderson *et al.*, 2012). In August 2016, at least 19 States had public health advisories related to cyanobacterial blooms (see Figure 4). Bloom occurrences are typically not predictable because interactions among multiple natural and anthropogenic factors determine the likelihood and severity in specific water bodies (Brooks *et al.*, 2016). Eutrophication, from human activities such as agriculture and urbanization, has been

suggested as a leading cause for the expansion and persistence of cyanobacteria (Taranu *et al.*, 2015; Heisler *et al.*, 2008). However, cyanobacterial blooms can also be affected by precipitation driven nutrient enrichment, solar radiation, water temperature, conditions characteristic of lentic systems (calm and still water), and other drivers (e.g., carbon dioxide, sea level rise) (Figure 5) (NSTEPS, 2015; USEPA, 2013).

Precipitation driven nutrient loading (soluble and adsorbed to sediment) has been associated with cyanobacterial blooms in large freshwater ecosystems (e.g., Lake Taihu and other large lakes in China, the U.S. Great Lakes, Lake Victoria and Lake George, Africa) (Qin *et al.*, 2010; Paerl *et al.*, 2011) as well as in estuaries like the Neuse Estuary in North Carolina, and the Potomac Estuary in Maryland (Sellner *et al.*, 1988; Robson and Hamilton, 2003; Paerl, 2008). The amount and timing of precipitation and runoff also affects key factors controlling cyanobacterial growth and dominance, including water residence time, vertical stratification, salinity in estuarine systems, and phytoplankton growth rates. Additionally, bed sediment often acts as a reservoir for nutrients and cyanobacteria for extended periods (Reichwaldt and Ghadouani, 2012; Potts, 1994). High flows following heavy precipitation can stimulate blooms by disturbing and resuspending cyanobacteria and nutrients stored in bed sediment.

Higher water temperatures and associated changes in chemical and biological processes promote cyanobacterial bloom formation and dominance in a range of aquatic ecosystems (Paerl and Paul, 2012; USEPA, 2013; Funari *et al.*, 2012; Havens and Paerl, 2015). As a group, cyanobacteria have a competitive advantage at higher temperatures - often above 25 degrees Celsius (°C) - than other algae (USEPA, 2013; Paerl and Huisman, 2009; Paerl *et al.*, 2011). Warmer surface waters strengthen vertical water column stratification which promotes the formations of cyanobacterial blooms. Without vertical mixing of the water column, cyanobacterial species can regulate intracellular gas vesicles to control their buoyancy (e.g., *Microcystis*, *Anabaena*), floating upward to optimize photosynthetic production (light absorption and use of atmospheric CO<sub>2</sub>) and sinking downward to optimize nutrient acquisition. This provides a competitive advantage over heavier, algae with faster sinking rates such as diatoms (Reynolds, 1987). Increases in atmospheric CO<sub>2</sub> can change water chemistry, lowering pH and favoring cyanobacteria that can grow faster in such conditions (USEPA, 2013; Paerl and Huisman, 2009).

Toxin-producing cyanobacteria strains are generally poor competitors for light compared to nontoxic strains (Kardinaal *et al.*, 2007), but can thrive in stable, calm waters common during dry periods. Long water residence times and reduced flow volumes which commonly occur in the late summer-fall generally favor the occurrence of cyanobacterial blooms (Elliott, 2012; Li *et al.* 2015; Paerl and Scott, 2010; Michalak *et al.*, 2013). Under these conditions, cyanobacteria enhance their competitive advantage by forming dense surface blooms, optimizing light absorption (Kahru *et al.*, 1993; Ibelings *et al.*, 2003) while shading deeper nonbuoyant algae (Huisman *et al.*, 2004). Cyanobacteria can also cope with the cell-damaging effects of ultraviolet radiation more effectively than nontoxic strains (Paerl and Otten, 2013).

Previous observations indicate that the ideal scenario for blooms is heavy rainfall that enhances runoff and nutrient delivery, followed by a protracted drought during the summer-

fall period, when water temperatures and stratification are maximal (Paerl and Scott, 2010; Reichwaldt and Ghadouani, 2012). This pattern has been seen in multiple systems (Paerl and Huisman, 2009). For example, heavy precipitation preceded the formation of a large cyanobacterial bloom outbreak in Lake Erie in 2015 that affected up to 300 square miles of the western basin (NOAA, 2015). Heavy rainfall in June (7 inches at Toledo, Ohio) resulted in high runoff and delivery of large amounts of bioavailable and reactive phosphorus into the western portion of the lake (NOAA, 2015). Subsequent calm conditions and warm waters caused an extended period of weak lake circulation, which resulted in abnormally long residence times. Those conditions provided perfect incubation, seeding, and growth conditions for bloom development. Similar drivers have been indicated for bloom events in the Ohio River, the Neuse River Estuary in North Carolina, the Potomac River and other Chesapeake Bay tributaries (Youngstrom and Emery, 2015; Sellner *et al.*, 1988; Robson and Hamilton, 2003; Paerl, 2008).

**Projected Future Changes.**—Few modeling studies assessing the response of cyanobacterial blooms to future conditions were identified in the current scientific literature. One recent national-scale modeling assessment reported that the extent of cyanobacterial blooms is likely to increase primarily due to higher water temperatures and increased nutrient levels associated with changing land use and precipitation patterns (see Figure 6) (USEPA, 2017; Chapra *et al.*, 2017). Simulations indicated that the mean number of days of bloom occurrence would change from about 7 days per year per water body under current conditions, to 16–23 days in 2050 and 18–39 days in 2090 (USEPA, 2017). The largest increases in occurrence were suggested for the northeast U.S., while the greatest impacts to recreation, in terms of costs, were projected in the Southeast (USEPA, 2017; Chapra *et al.*, 2017).

Other studies also point to an increased risk of cyanobacterial blooms across the U.S. in response to future climate change (Trtanj *et al.*, 2016; Najjar *et al.*, 2010). Increased air and water temperatures are expected to expand the seasonal window of occurrence, particularly at higher latitudes (Trtanj *et al.*, 2016; Wells *et al.*, 2015; Moore *et al.*, 2008). Warmer temperatures are known to promote vertical density stratification and provide conditions more favorable to cyanobacteria (Paerl and Scott, 2010; Havens and Paerl, 2015; Mantzouki *et al.*, 2018). More frequent heavy precipitation events and associated excessive nutrient loading will increase the risk of eutrophic water conditions which enhance cyanobacterial growth (Prein *et al.*, 2016; Li *et al.*, 2015). Longer dry periods are expected between precipitation events in many parts of the U.S. during the summer-fall seasonal window of bloom occurrence (USGCRP, 2017). These conditions increase water residence times, cause weak water circulation and contribute to warmer water temperatures - all of which favor bloom development. In the marine environment, increases in salinity, associated with sea level rise, could also promote stratification and increase the risk of cyanobacterial blooms (Paerl and Paul, 2012; Fu *et al.*, 2012). Any of these future conditions, alone or in combination, could trigger the development of cyanobacterial blooms and the release of harmful toxins.



**Ecosystem Level Impacts.:** An increase in the occurrence of cyanobacterial bloom events would have negative ecosystem and economic consequences including impacts on human health, recreation and aquatic life (Lopez *et al.*, 2008; Anderson *et al.*, 2012; Hudnell, 2010; Lopez *et al.*, 2008; USEPA, 2015a; NSTEPS, 2015). Toxins, associated with bloom events, represent a risk to human health through drinking water and recreational contact. Recent bloom events, such as in 2015 on the Ohio River, demonstrated the challenge that treatment facilities face in providing safe drinking water when encountering extreme blooms (IWG-HABHRCA, 2016). Standard drinking water treatment methods can remove cyanobacterial cells and low levels of cyanotoxins, but additional treatment technologies can be needed, especially when source waters are in the midst of a bloom or confronted with high levels of cyanotoxins (USEPA, 2015b). The effects on recreation, tourism (e.g., closed water bodies, habitat value) and aquaculture (e.g. fishing, shellfish beds) can also be significant but have not been well quantified and documented in the U.S.

Bloom events often kill aquatic organisms directly through the production of algal toxins or indirectly if dissolved oxygen supplies become depleted (NSTEPS, 2015). Hypoxia and cyanobacterial blooms are typically closely associated (Elliott, 2012; Lopez *et al.*, 2008; USEPA, 2015a). Following a bloom, bacterial decomposition of cyanobacterial detritus consumes dissolved oxygen, which may quickly decrease to levels insufficient to sustain most aquatic life, producing hypoxic or anoxic conditions (CENR, 2010). Cyanobacterial blooms also reduce water clarity and prevent sunlight from reaching submerged aquatic vegetation and benthic microalgae, reducing the release of oxygen from photosynthetic plants (IWG-HABHRCA, 2016). In lakes and reservoirs, vertical stratification reduces the rate at which oxygen can replenish deeper waters. As a result, settling and decomposition of organic matter in bottom waters and sediments can rapidly deplete available oxygen in stratified bodies of water (IWG-HABHRCA, 2016). Changes in chlorophyll-a (a measure of cyanobacterial biomass), water clarity, and dissolved oxygen are primary reasons for aquatic life use impairments in U.S. water bodies (USEPA, 2014b).

## Sediment

**Observed Sensitivity.**—Sediment in rivers and streams is derived from upland sources transported by runoff, and from erosion of instream bed and banks. The balance between instream (channel) bed and bank erosion and sediment storage is highly dynamic. If transport (erosion and bank incision) exceeds delivery, degradation occurs. Conversely, where sediment delivery exceeds instream transport capacity, channels can aggrade and store excess sediment (Brakebill *et al.*, 2010; Zhang *et al.*, 2013; James, 2013). Few studies were identified assessing long-term precipitation related trends in sediment loads, but sediment responses to weather events have been widely reported (Jackson *et al.*, 2005; Salant *et al.*, 2008; Jordan *et al.*, 2014; Nichols *et al.*, 2013). Precipitation, together with land use, management and vegetative cover, are considered key drivers of sediment delivery to streams (Belmont, 2010; Jordan *et al.*, 2014; Swank *et al.*, 2014).

In addition to precipitation, changes in air temperature and humidity indirectly affect sediment transport through changes in evapotranspiration, soil moisture and ground cover (vegetation) that influence runoff. Vegetative cover is a key resistive force to erosion from

upland areas (Sherrif *et al.*, 2018; SWCS, 2003; O' Neal *et al.*, 2005). In the Southwest, high rates of erosion due to lack of vegetation and steep topography are evident in some locations (Nichols *et al.*, 2013). Lower soil moisture reduces the production of plant biomass, leaving soils more susceptible to erosion (Baruti, 2004; Flanagan and Johnson, 2005). In contrast, high soil moisture can reduce the stability of steep hillslopes, bluffs, and stream banks, increasing sediment input from mass movement. Changes in temperature can also influence erosion rates through effects on soil microbial activity (Frey *et al.*, 2013), rates of organic matter decay (Pietikainen *et al.*, 2005) and the formation of surface crusts on soils (Pruski and Nearing, 2002).

Sediment erosion and delivery to water bodies, and instream bed and bank erosion are driven by precipitation and runoff (Lawler *et al.*, 1997). Many studies have reported that suspended sediment loads strongly correlate with rainfall and runoff (Jackson *et al.*, 2005; Salant *et al.*, 2008; Jordan *et al.*, 2014). Jiang *et al.*, (2014) found that turbidity in different U.S. regions was positively correlated to latitude and generally followed a geographic pattern of greater to lesser turbidity similar to that of average annual precipitation.

The intensity and annual timing (seasonal distribution) of precipitation are also important (Hancock, 2012). Sediment transport is highly episodic, occurring mainly during larger events which mobilize upland sediments (Ffolliott *et al.*, 2013; di Silvio and Basson, 2008; Gellis *et al.*, 2003). Heavy rainfall at times when soils have reduced vegetative cover (e.g., in spring before vegetation ground cover is established) often contribute a big proportion of the annual sediment loads (Poff, 1992; Goode *et al.*, 2012). The timing of rainfall relative to agricultural tillage can therefore be particularly important. In the Chesapeake Bay watershed, sediment loads were found to be highly sensitive to transport during heavy rainfall and high runoff, the dynamics of which are influenced by long-term weather trends as well as decadal-scale oscillations (Langland *et al.*, 2012). Intense rainfall can drive upland sheet erosion, erosion in rills and gullies, mass movements (e.g. landslides), and debris flows (di Silvio and Basson, 2008). During such events, streamflow driven bank incisions and failures (erosion and mass wasting of sediments) can be a major mechanism of erosion in some systems (Lawler *et al.*, 1997; Fraley *et al.*, 2009).

Wildfires in the western U.S. have strongly influenced short term sediment yields in some watersheds (e.g., Snake River basin, Idaho) (Goode *et al.*, 2012; Lancaster *et al.*, 2001). Wildfires typically reduce vegetative cover, expose soils, alter soil physical properties and make large areas of land more susceptible to erosion (Goode *et al.*, 2012). Drought, pests and disease outbreaks also reduce vegetative cover (Sturrock *et al.*, 2011; Wolfe *et al.*, 2008).

**Projected Future Changes.**—Understanding sediment load responses to potential future changes in air temperature and rainfall is challenging given the importance of episodic, heavy precipitation on erosion and transport. The long-term response of vegetation to increases in air temperature and CO<sub>2</sub>, and changes in soil moisture is also central as vegetative cover typically reduces erosion (SWCS, 2003). Annual loads can therefore be dominated by a few heavy precipitation and high runoff events, particularly at times of the year when vegetative cover is minimal. Interactions with the local landscape and watershed conditions, including human activities such as land use change, will thus be important in a

changing future. The effects of changing precipitation and runoff patterns on dynamics in different watershed settings will initially depend on whether sediment loads are source limited (i.e. loading varies little with changes in rainfall) *versus* transport limited (i.e. loading increases with increased rainfall) and subsequently whether future conditions cause a watershed to switch states (Goode *et al.*, 2012).

Figure 7 displays the locations of watershed scale studies that assess potential future sediment load responses. Many studies in the northern and eastern U.S. suggest a risk of increased annual loads. These changes broadly correlate with projected increases in annual precipitation and runoff. In parts of the interior West and Southwest, decreases in summer-fall runoff could contribute to lower annual sediment loads. Increased heavy precipitation and runoff events throughout the country (Prein *et al.*, 2016) are expected to increase the risk of episodic high sediment loading to water bodies. Studies broadly concluded that higher soil erosion rates are likely, particularly if increases in heavy precipitation (frequency and intensity) are realized (Pruski and Nearing, 2002; Sun, 2013; Marion *et al.*, 2014; Chang *et al.*, 2010; Wang and Kalin, 2018).

Stored legacy sediments (in floodplains, channel bars, streambeds, and reservoir) have the potential for remobilization as a source of secondary pollution (Langland *et al.*, 2003; Fraley *et al.*, 2009; Carter *et al.*, 2003; James, 2013; Buchty-Lemke *et al.*, 2016; Niemitz *et al.*, 2013). In areas of the mid-Atlantic and Southeast, stored legacy sediment from past agricultural practices and other human activities are currently a major source of in-stream sediment loads (James, 2013). Increased frequency and magnitude of heavy precipitation and runoff could amplify the significance of stored sediments as sources contributing to stream loads in the future.

In drier and warmer areas prone to wildfire, like California, more frequent fire-related impacts on stream sediment loads could occur (Goode *et al.*, 2012). However, studies about future effects on sediment have yet to consider changes due to this type of disturbance.

**Ecosystem Level Impacts.:** Sediment transport is a natural function of rivers and streams, but is also one of the most common pollutants affecting U.S. water bodies. Natural sediment storage increases the net residence time in a watershed and can provide an ecosystem service (Herman *et al.*, 2003). However, large changes in sediment deposition and transport can alter channel morphology (e.g., slope, channel cross-section) and have substantial effects on the physical habitat necessary to support fish and other aquatic life (USEPA, 2006; Poff, 1992; Brakebill *et al.*, 2010). For aquatic life, excess sediment in the water column can decrease interstitial habitat space (e.g., changes in channel form, infiltration and infilling of gravel and cobble beds), cause loss of suitable habitat, reduce light penetration and impact spawning and hatching (e.g., limiting oxygen amounts in the spawning beds, and trap newly hatched fish) (Waters, 1995; Wood and Armitage, 1997; USEPA, 2006). In addition, changes in the streambed sediment volume can affect flow depth, conduction, and hyporheic exchange, which influence water temperature and pollutant processing (Caissie, 2006). Suspended sediment and turbidity concentrations also influence water temperature through effects on the absorption or reflection of solar radiation - high turbidity focuses solar energy on the upper part of the water column. It should be noted, however, that it can be difficult to

separate the direct effects of sediments from other changes that are interrelated (e.g., streamflow) and affect biota (Bond and Downes, 2003).

Excessive sediment loading often introduces high levels of sediment-adsorbed pollutants (e.g., pathogens, nutrients, metals, and toxic substances) to water bodies that can present a risk to human and ecosystem health (Stout *et al.*, 2014; Ahmadi *et al.* 2014; Ffolliott *et al.*, 2013; Ficklin *et al.*, 2010; Ahmadi *et al.*, 2014). Concentrations of particle adsorbed pathogenic microorganisms are often higher than those dissolved in the water column (Kim *et al.*, 2010; deBrauwere *et al.*, 2014), and several studies have shown elevated turbidity to be correlated with increased outbreaks of waterborne illness (Morris *et al.*, 1996; Schwartz *et al.*, 1997). Elevated levels of sediment particles in source waters can therefore be an issue for drinking water treatment by causing physical disruption to filtration processes and other treatments challenges related to adsorbed pollutants (e.g., disinfection of pathogens and disinfection by-product formation).

## Pathogens

**Observed Sensitivity.**—The occurrence of microbial pathogens in water bodies is influenced by temperature which affects survival, and precipitation that drives the transport of fecal waste (which may contain pathogenic organisms) from upland sources (Figure 8). Sunlight, moisture conditions, salinity, and other factors also influence survival in the natural environment (USEPA, 2001). Most waterborne pathogens sourced to fecal waste (e.g., pathogenic *E. coli*) can survive for long periods in different environmental matrices (e.g., soil, manure, and water) when temperatures are low, when they are protected from external factors such as ultraviolet radiation, and when appropriate moisture and nutrients are available (Tyrrel and Quinton, 2003; USEPA, 2013; Rogers and Haines, 2005; Pommepuy *et al.*, 1992).

Pathogenic *E. coli* can survive longer in water bodies at lower temperatures and up to two or three times longer in river and lake sediments (bedded sediment can provide sufficient moisture, availability of nutrients and protection from sunlight) (USEPA, 2009a; Davies *et al.*, 1995; Decamp and Warren, 2000; Jamieson *et al.*, 2004; Koirala *et al.*, 2008; Kim *et al.*, 2010; Characklis *et al.*, 2005; Sherer *et al.*, 1992). Protozoans, such as *Giardia* and *Cryptosporidium*, can survive from months to more than 1 year in cool water (approximately 5 °C or less) (Ziemer *et al.*, 2010).

At temperatures exceeding 30 degrees (°C), survival rates for pathogens sourced to fecal waste have been shown to decrease (Rogers and Haines, 2005; USEPA, 2001; Hofstra, 2011; Vermeulen and Hofstra, 2014; Herrador *et al.*, 2015).

However, warm summer temperatures can also contribute to human exposure by extending the period of warm-weather recreational uses. In addition, higher water temperatures provide more favorable conditions for some naturally occurring organisms, like *Naegleria fowleri*, and *Vibrio* spp. in coastal systems. Two documented cases of *Naegleria fowleri* infection in Arkansas were associated with exceptionally warm water temperatures during periods in which air temperatures were above 37 °C degrees. Health officials reported elevated water temperatures (and other factors) as being conducive to *Naegleria fowleri* at the time of

exposure (Matthews *et al.*, 2014). Jiang *et al.*, (2015) reported increases in waterborne and food-borne salmonellosis risk in Maryland during extreme temperatures based on a 30-year baseline (i.e., 1960 – 2012).

Fecal bacteria levels in water bodies usually correlate positively with precipitation and runoff (Curriero *et al.*, 2001; Kistemann *et al.*, 2002; Schijven and Husman 2005; Nichols *et al.*, 2009; Kratt *et al.*, 2010; Hofstra 2011; Funari *et al.*, 2012; Cann *et al.*, 2013; Herrador *et al.*, 2015; Tryland *et al.*, 2011; Vermeulen and Hofstra, 2014). Precipitation driven sediment transport can also be important in governing the mobility and loading of sediment-adsorbed microorganisms (Kim *et al.*, 2010; de Brauwere et al 2014; Soupier and Pandey, 2016). Many waterborne disease outbreaks have been shown to be associated with heavy rainfall (Figure 8) (WHO, 2003; Rizak and Hruday, 2008; Curriero *et al.*, 2001; Kistemann *et al.*, 2002; Cann *et al.*, 2013) and elevated turbidity (Abia *et al.*, 2016; Morris *et al.*, 1996; Schwartz *et al.*, 1997). For example, Curriero *et al.*, (2001) analyzed 548 waterborne disease outbreaks that occurred in the U.S. between 1948 and 1994 and found that over half of them were preceded by heavy rainfall. Severe winter storms and snowmelt were linked in part with the largest reported cryptosporidiosis outbreak in the U.S., which occurred in Milwaukee in 1993 (MacKenzie *et al.*, 1994). In Maryland's portion of the Chesapeake Bay, annual and seasonal precipitation totals from 1979 to 2013 had a strong positive relationship with average fecal bacteria levels in shellfish harvest waters (Leight *et al.*, 2016).

Frequent high fecal bacteria levels at times of no precipitation are typically connected to land management factors such as point sources discharges or failing septic/sewer systems (Cahoon *et al.*, 2016). In several locations, periods of dry weather followed by heavy rainfall have also preceded outbreaks of waterborne disease (TDOH, 1999; Patz *et al.*, 2000; Funari *et al.*, 2012). Extended dry periods can allow fecal waste to accumulate on land (e.g., beaches, pastures, forestry) and then be flushed by precipitation and runoff into water bodies (Stewart *et al.*, 2013; Funari *et al.*, 2012).

Many studies indicate that heavy precipitation and other weather events have been contributing factors to outbreaks of waterborne disease in the U.S. (see Figure 9). However, it is worth noting, that the interactions of human activities and management (e.g., water treatment failures) with extreme weather events are responsible in many outbreak instances.

**Projected Future Changes.**—Projected increases in temperature together with changes in precipitation and runoff are expected to alter pathogen survival and transport to water bodies (Trtanj *et al.*, 2016; Hofstra, 2011; Coffey *et al.*, 2014). Relatively few studies, however, use numerical modeling to assess potential future changes in pathogen fate and transport in U.S. water bodies (see Figure 9). Modeling work in Virginia, Mississippi, and Illinois suggests increased fecal indicator bacteria loads (FIB - indicate the potential presence of pathogenic organisms) in a changing future (Coffey *et al.*, 2015a, b; Liu *et al.*, 2010; Jayakody *et al.*, 2015; Patz *et al.*, 2008). Other theoretical studies suggest an increased risk of human exposure and waterborne illnesses (Lo Iacono *et al.*, 2017; Trtanj *et al.*, 2016; Levy *et al.*, 2016; Vavrus and Behnke, 2014; Uejio *et al.*, 2014; Coffey *et al.*, 2014; Jiang *et al.*, 2015).

Higher air and water temperatures, projected throughout the U.S. (Melillo *et al.*, 2014; Hill *et al.*, 2014), are expected to reduce survival rates for some common organisms sourced to fecal waste, such as pathogenic strains of *E. coli* (Schijven and Husman, 2005). Other species like naturally occurring *Vibrio* spp. and *Legionella* grow faster in warmer water and may become more prevalent geographically and seasonally (Lipp *et al.*, 2002; Jacobs *et al.*, 2015; Trtanj *et al.*, 2016; Najjar *et al.*, 2010). Warmer temperatures could also lead to the spatial expansion of new microorganisms (e.g., such as amoeboid pathogens *Naegleria* and *Acanthamoeba*), vectors, and intermediary hosts (Harrus and Baneth, 2005; Hoskisson and Trevors, 2010).

Future changes in precipitation and runoff are likely to present an increased risk of non-point source fecal waste loading (e.g., NPSs as well as urban sanitary sewer overflows and combined sewer overflows) to water bodies. The delivery of fecal waste (which can contain pathogenic organisms) to water bodies is largely episodic, driven by rainfall. While uncertainty remains regarding regional changes in rainfall (USGRCP, 2017), future increases in the proportion occurring in large-magnitude events increase the likelihood of loading (dissolved and adsorbed to sediment) from upland fecal sources (Hofstra, 2011; Coffey *et al.*, 2014; Cann *et al.*, 2013; Trtanj *et al.*, 2016; Strauch *et al.*, 2014). Associated high streamflow volumes could also re-suspend and mobilize microorganisms stored in river and lake bed sediments (Soupir and Pandey, 2016; Wu *et al.*, 2009; Garzio-Hadzick *et al.*, 2010). In addition, potential increases in dissolved organic matter and browning of water bodies during heavy precipitation events has the potential to reduce solar UV inactivation of pathogens (Williamson *et al.*, 2017).

Dry periods are expected to become common in many regions, particularly in the mountain west and interior southwest (Melillo *et al.*, 2014). Lower flow volumes and reduced dilution during these periods could result in episodic increases in fecal waste levels downstream of point-source discharges (e.g., wastewater treatment plants) (Senhorst and Zwolsman, 2005; Johnson *et al.*, 2009; Hofstra, 2011; Funari *et al.*, 2012; Cann *et al.*, 2013; Coffey *et al.*, 2014; Strauch *et al.*, 2014).

In coastal and estuarine waters, future changes in salinity due to sea level rise could affect pathogen survival (Burge *et al.*, 2014). Most waterborne pathogens have significantly lower survival rates in high-salinity environments than in less saline environments (Canteras *et al.*, 1995; Bordalo *et al.*, 2002). However, *Vibrio vulnificus* favors more moderate salinities, while *Vibrio parahaemolyticus* and *Vibrio alginolyticus* favor higher salinities (Trtanj *et al.*, 2016; Urquhart *et al.*, 2014).

**Ecosystem Level Impacts.:** An increased risk of human exposure to pathogens through drinking water and recreational contact is anticipated across the U.S. due to warming air temperatures and changing precipitation patterns (pathways are illustrated in Figure 8) (Coffey *et al.*, 2014; Sterk *et al.*, 2013). Communities relying on untreated drinking water sources (e.g., private groundwater wells and other sources not served by a treated supply) are likely to be at greater risk of exposure and waterborne illness (Pons *et al.*, 2015). Many previous waterborne disease outbreaks have been linked to drinking water supplies sourced from groundwater - shallow groundwater wells influenced by surface runoff are more

vulnerable to precipitation driven fecal contamination (Levin *et al.*, 2002). In source waters used for treated drinking supply (where disinfection and filtration are effective at removing pathogens), utilities may face challenges from increased microbial contamination. For example, a higher level of disinfection (e.g., chlorination and ozone) could be required to inactivate pathogens, which would increase the potential for generation of disinfection by-products, many of which have negative human health consequences (USEPA, 2009c).

Maintaining water bodies within existing recreational water quality standards could also be an issue if projected increases in microbial loads are realized. Many waterborne illnesses occur from recreational water use during warmer weather (DeFlorio-Barker *et al.*, 2018; Hlavsa *et al.*, 2014; Curriero *et al.*, 2001; Freeman *et al.*, 2009; McBride *et al.*, 2014). Higher seasonal air temperatures are expected to expand the window of recreational water use and increase the risk of human exposure to waterborne pathogens (Casman *et al.*, 2001; Schijven and Husman, 2005). Additionally, more frequent violations of U.S. recreational water quality standards have been projected in some studies (Coffey *et al.*, 2015a, b; Liu *et al.*, 2010; Jayakody *et al.*, 2015).

## FUTURE RESEARCH

This review suggests that anticipated future increases in air temperature and changes in precipitation present an increased risk of water quality and ecosystem degradation in many U.S. locations. Responding to this challenge requires a more detailed understanding of key vulnerabilities in different regional and watershed settings, and the development of effective adaptation strategies to reduce risks. Research needs emerging from this review to improve understanding and better inform management responses include the following:

- i. Knowledge of potential future water quality responses is limited by the small number of studies that currently exist. Water quality is highly variable spatially among U.S. regions and in specific watershed settings, and temporally during the year in response to runoff. These spatial gaps can pose challenges for decision makers, who need to make specific decisions in the best interest of their programs (Watts *et al.*, 2015). More watershed scale assessments applying consistent approaches are needed to extend our geographic understanding, allow spatial comparisons and identify areas at most risk. This is essential to broaden our understanding of potential future responses and inform the development of local-scale response strategies for managing future water quality risks.
- ii. Observed trends in water quality typically reflect the interaction between climatic variability, changes in land use, changes in management and other factors. This makes attribution of changes to specific drivers difficult. Efforts to link specific water quality changes over time to climate and weather events would add to our knowledge about responses in different watershed settings. Many current studies of observed responses to climate effects are limited to a relatively short temporal window of data availability (e.g., 1990s to 2010s) or sparse spatial distribution. Relationships between trends and climatic drivers are also often obscured by changes in other factors such as land use. Expansion and continuation of existing, long-term water quality monitoring networks could thus

reveal new insights into the drivers of trends. For example, many Long Term Ecological Research sites (see <https://lternet.edu/site/>) (and others) continuously monitor changes in water quality, providing unique opportunities to examine connections between water quality trends and climate (Morse and Wollheim, 2014; Ford *et al.*, 2011; Worrall *et al.*, 2003; Sun *et al.*, 2008). New technologies should also be explored and/or expanded such as use of continuous in-situ monitoring, regular remotely sensed monitoring, or other approaches that can provide efficient, accurate, and reliable detection and tracking of observed water quality responses (Schaeffer *et al.*, 2015; Urquhart *et al.*, 2017; Paerl and Huisman, 2009; Brooks *et al.*, 2016).

- iii. Improving the way uncertainties are characterized and communicated, particularly those associated with long-term changes in precipitation, is needed to best inform management decision making (Helgeson, 2018; Watts *et al.*, 2015; Johnson *et al.*, 2015). Precipitation and runoff are well known drivers of pollutant loading and most responses correlate with changes in precipitation. Also, effects of warming air temperatures on water quality are understood with a relatively high level of confidence. Effectively communicating relative confidence of potential impacts in a way that's meaningful to decision makers would better inform risk management.
- iv. Advancing models and methods used to simulate water quality responses to future scenarios can provide additional confidence in projections. Simulations of pollutant fate and transport can be subject to error (Novotny and Stefan, 2007), and inadequacies have been reported in many studies (Tu, 2009; Crossman *et al.*, 2013; Jha *et al.*, 2015; Johnson *et al.*, 2015). Areas of concern include sediment resuspension processes (Jamieson *et al.*, 2004; Droppo *et al.*, 2009; Coffey *et al.*, 2010a, 2010b), subsurface transport, and capabilities to simulate pollutant responses during extreme flow events (Benham *et al.*, 2006; Beckers *et al.*, 2009). In addition, modeling cyanobacterial blooms is currently difficult due to the many causal factors that can initiate events (Brooks *et al.*, 2016; Lopez *et al.*, 2008; Ho and Michalak, 2015).
- v. Given the risk of potentially large water quality changes, perhaps most importantly, information is needed about the type, extent and performance of Best Management Practices (BMPs) necessary to meet management objectives under changing future conditions (Alam *et al.*, 2017; Sun *et al.*, 2008). Increases in air temperature and altered precipitation, are expected to affect the performance of BMPs designed to remove pollutants (Wagena and Easton, 2018). For example, more intense precipitation increases erosion and transport from agricultural fields, increases leaching through sub-surface pathways, and reduces contact time in practices that rely on filtration. BMP performance can also be affected by changes in plant growth (e.g., for filter strips or cover crops), by increased rates of decay of surface residue, and more rapid cycling of nutrients under warmer conditions. Impacts on water quality will ultimately depend on the effectiveness of management responses (e.g., BMPs) which may not have been designed to cope with anticipated pressures (Wear and Greis,



2012). Successful adaptation strategies should therefore emphasize flexibility and robustness, and will need to select BMPs that reduce vulnerabilities across a wide range of possible future conditions (Heisler *et al.*, 2008).

## SUMMARY AND CONCLUSIONS

In this paper we review the historical and potential future effects of changes in air temperature and precipitation on nutrients, cyanobacterial blooms, sediment and waterborne pathogens. Water quality changes are complex and affected by multiple, interacting climatic, watershed and human factors. This review focuses only on water quality responses to changes in climatic drivers.

Future changes in nutrients, sediment, waterborne pathogens and cyanobacterial blooms will increase the risk of water quality and ecosystem degradation in many U.S. locations. There is, however, a high degree of spatial and temporal variability among watersheds. Increasing variability of precipitation is anticipated throughout the U.S., including a greater proportion of annual precipitation occurring in heavy events, and longer dry periods between events. Projected water quality responses in different locations reflect these changes, with many studies suggesting future changes in nutrient, sediment and fecally sourced pathogen loads that correlate with projected precipitation.

Increased air and water temperatures could have wide ranging impacts on water quality, aquatic life and human use. An expanded seasonal window of warm water temperatures and the potential for more frequent episodic nutrient loading (associated with increased heavy precipitation) will increase the risk of cyanobacterial blooms in large rivers, lakes and estuaries. Warmer air and water temperatures are expected to reduce survival for some waterborne pathogens sourced to fecal waste (e.g., pathogenic *E. coli*) but increase survival and northward expansion of others (e.g., naturally occurring *Naegleria fowleri*, and *Vibrio* species in coastal systems).

Managing the risk of harmful impacts will require adaptation strategies that reduce vulnerabilities in different watershed settings, across a range of plausible future conditions. An improved understanding of the ability to manage anticipated impacts can provide a more complete assessment of where and how watersheds are vulnerable.

## Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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**Research Impact Statement:** Climate change will have direct and cascading effects on water quality that vary in different regional and watershed settings, and could present a risk to human health and the environment.

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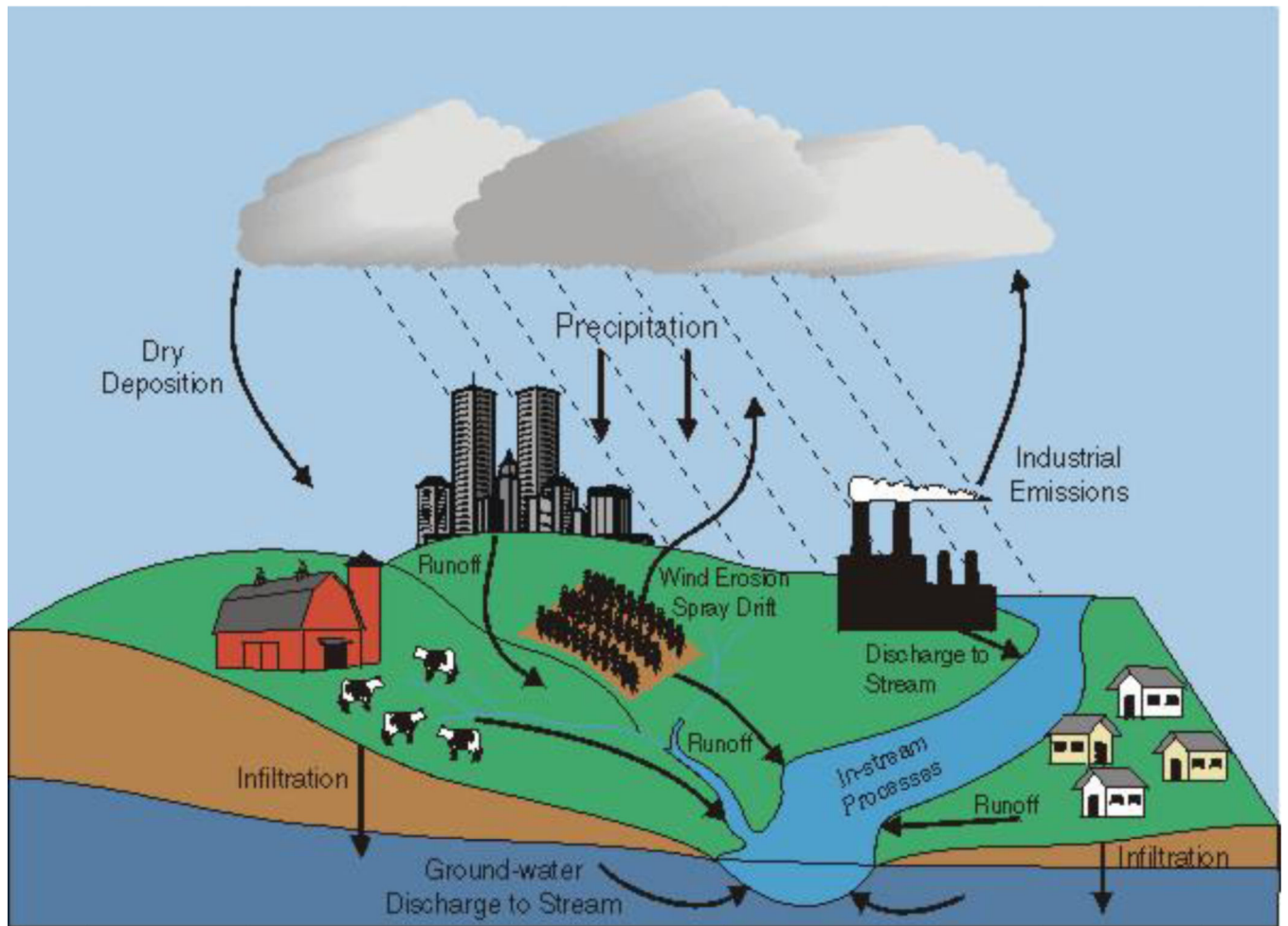
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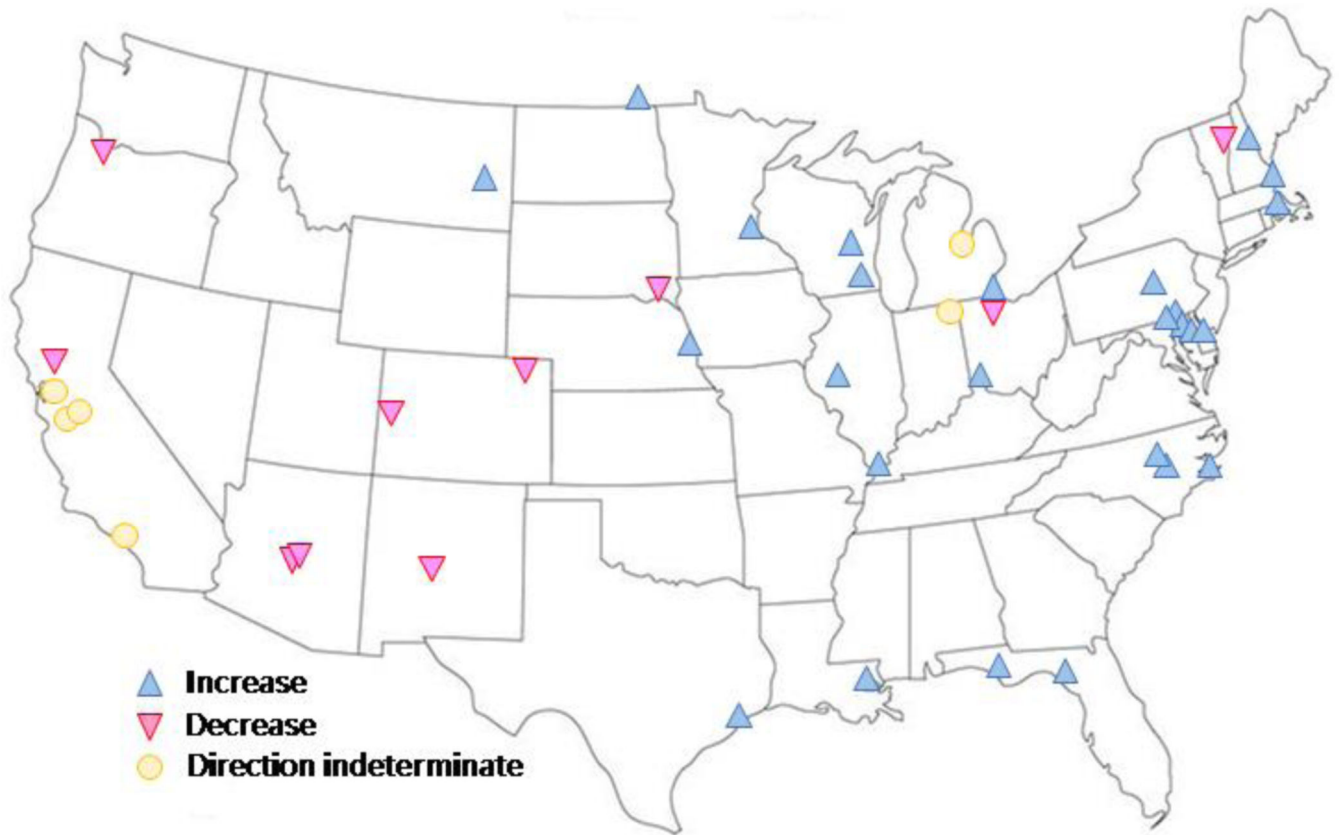
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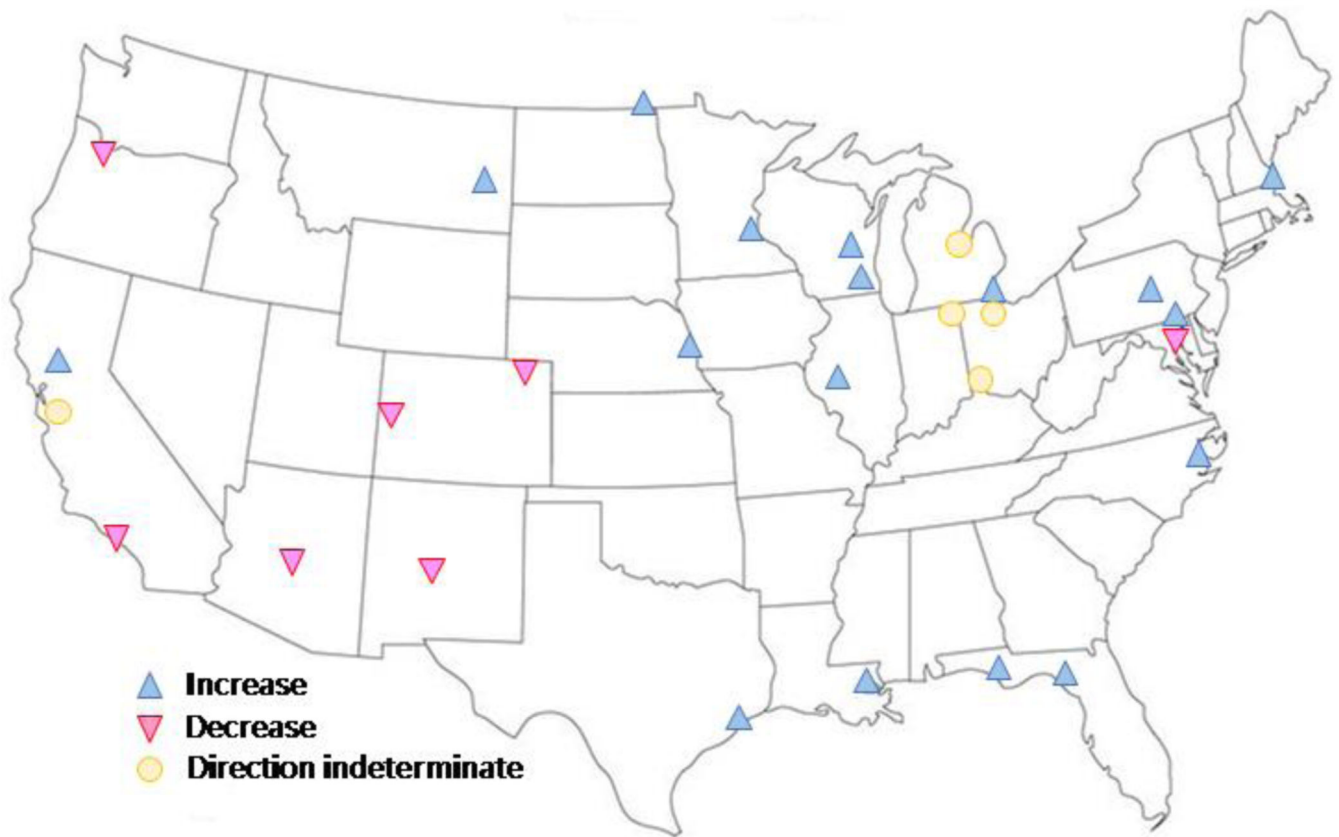


**FIGURE 1.** Nutrient movement in the hydrological cycle. Source: Belval and Sprague (1999).



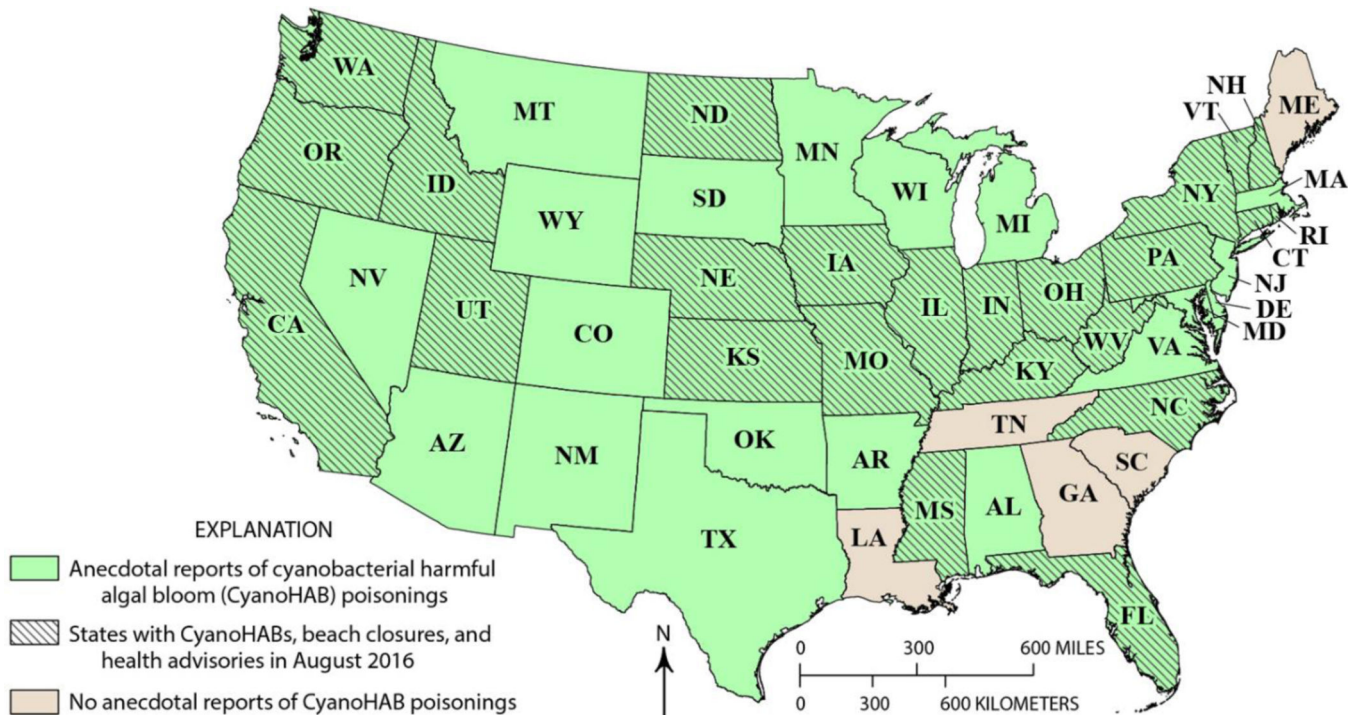
**FIGURE 2.**

Location of watershed modeling studies assessing nitrogen load responses to future climate change scenarios. Symbols indicate the suggested direction of change (based on ensemble median, annual loads). Studies not reporting an ensemble median (e.g., a range, or sensitivity) are shown as “Direction indeterminate.” Detailed information about each study is provided in the Supporting Information.

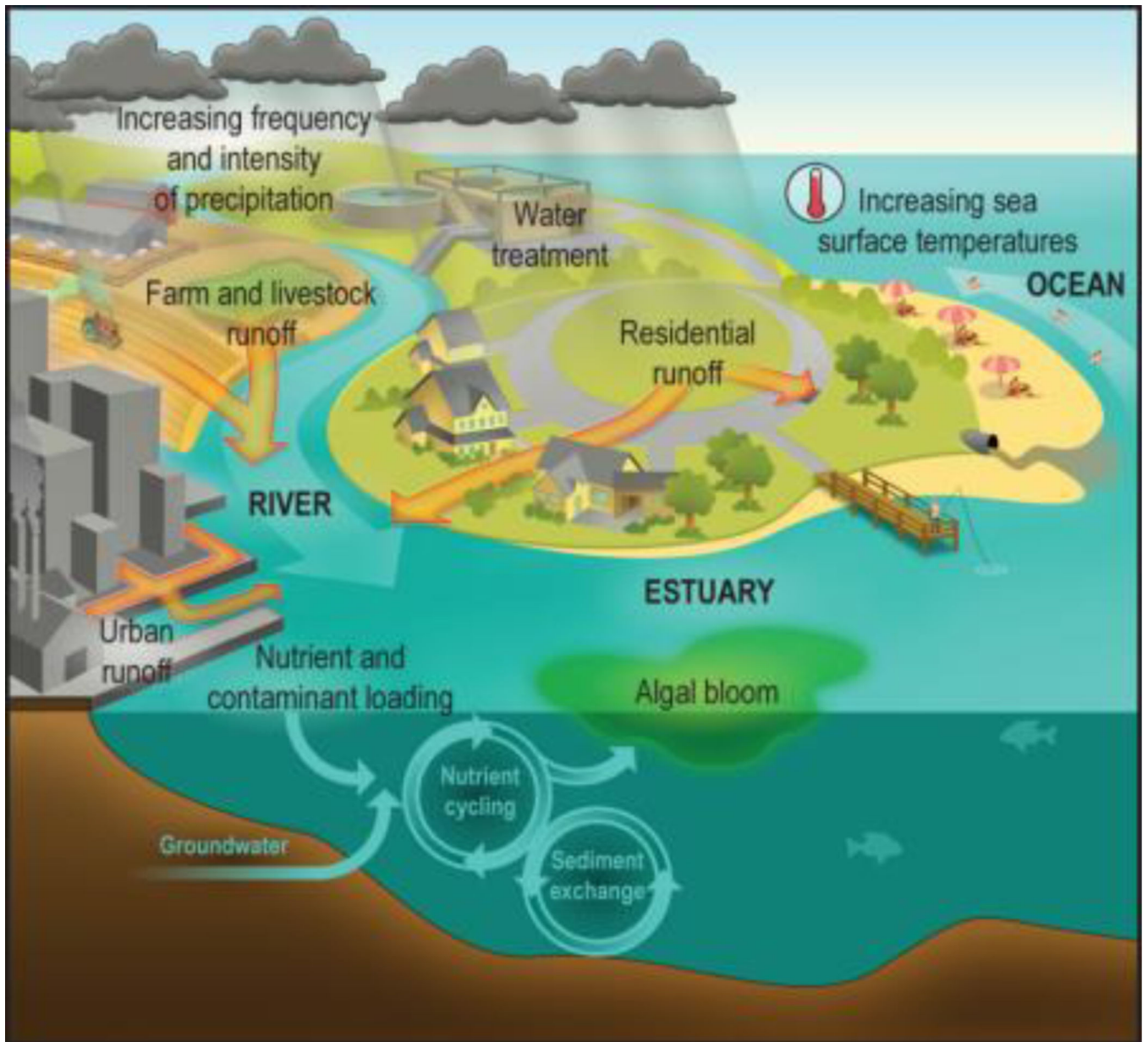


**FIGURE 3.**

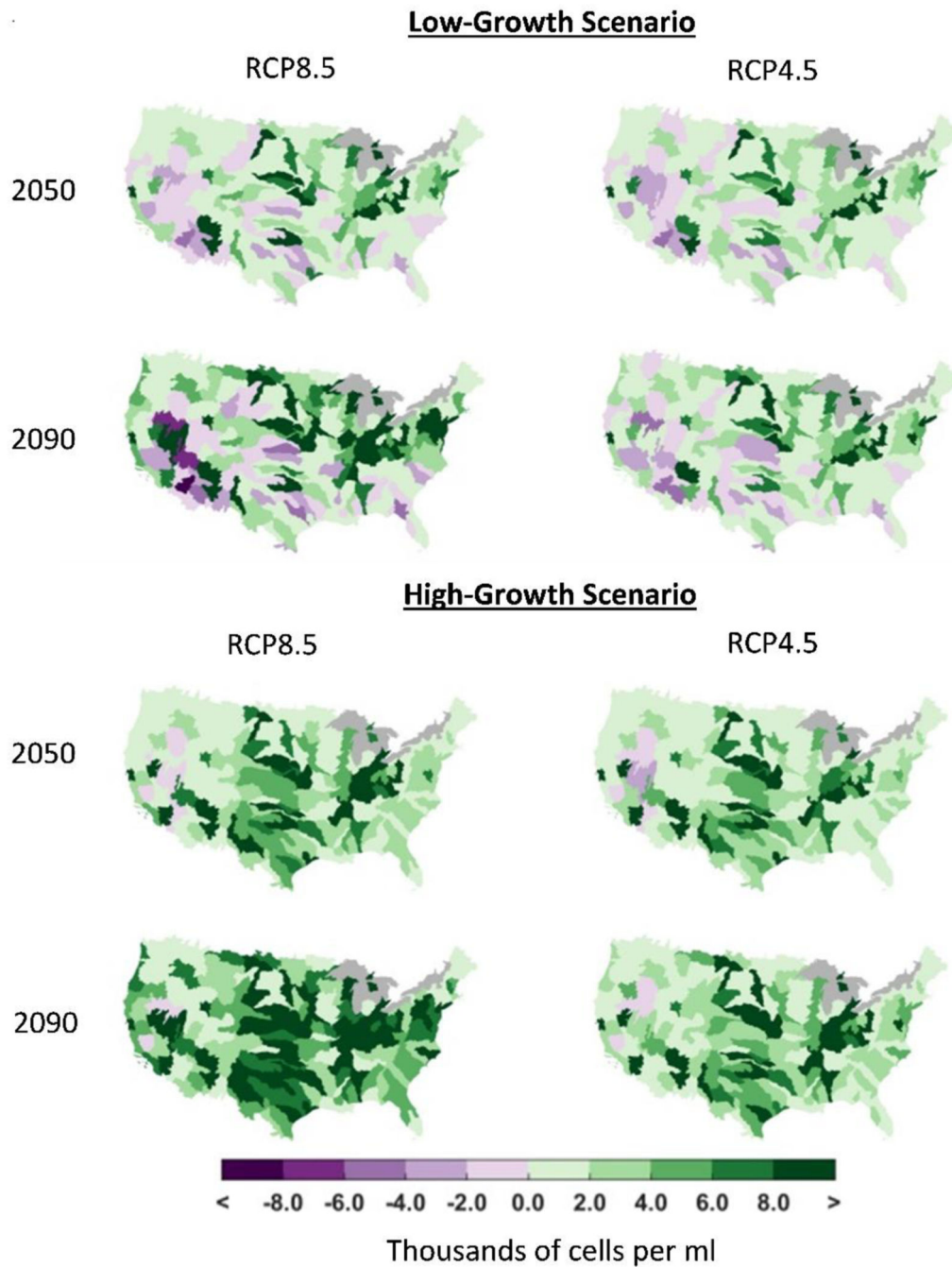
Location of watershed modeling studies assessing phosphorus load responses to future climate change scenarios. Symbols indicate the suggested direction of change (based on ensemble median, annual loads). Studies not reporting an ensemble median (e.g., a range, or sensitivity) are shown as “Direction indeterminate.” Detailed information about each study is provided in the Supporting Information.



**FIGURE 4.** National status for cyanobacterial blooms in August 2016. Source: USGS (2016).

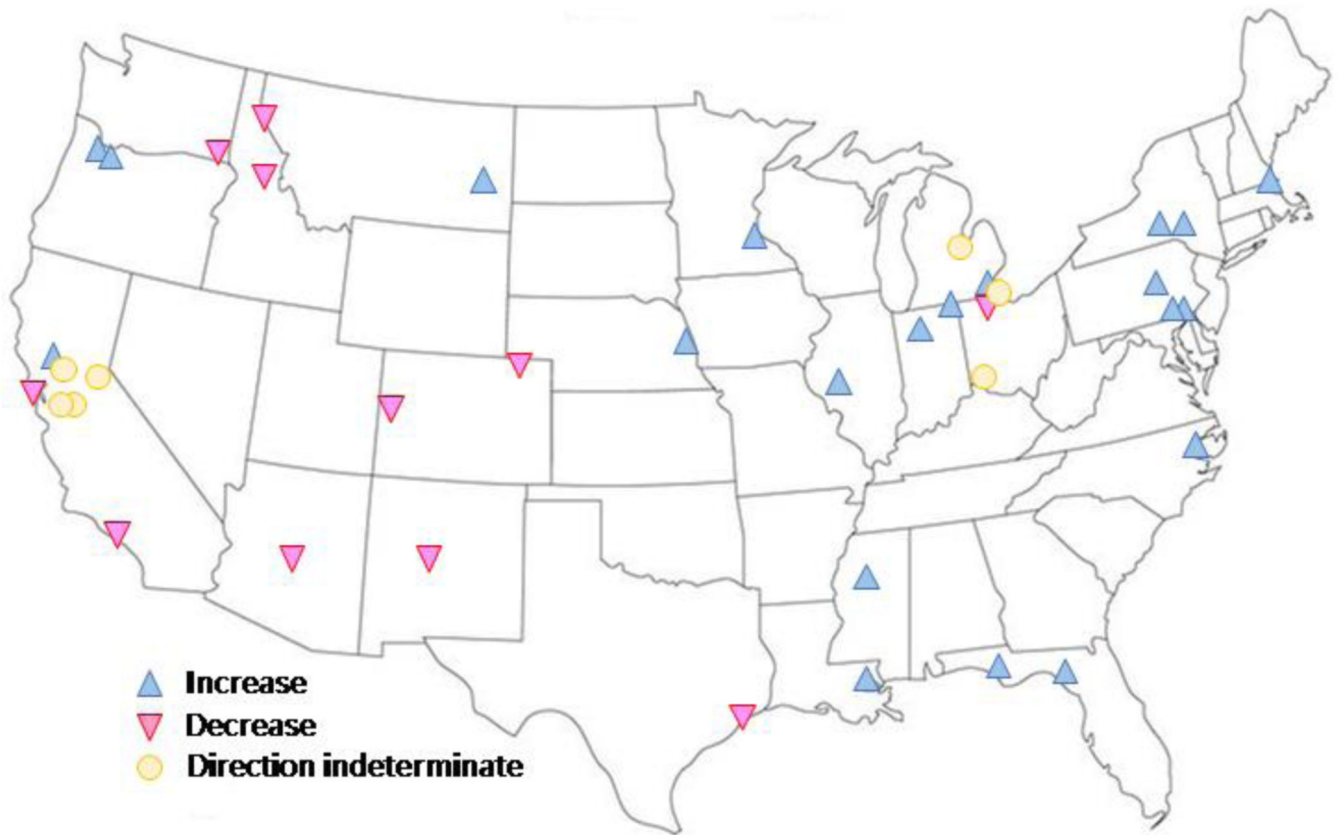


**FIGURE 5.** Factors influencing the formation of cyanobacterial blooms. Source: Trtanj et al. (2016).



**FIGURE 6.**

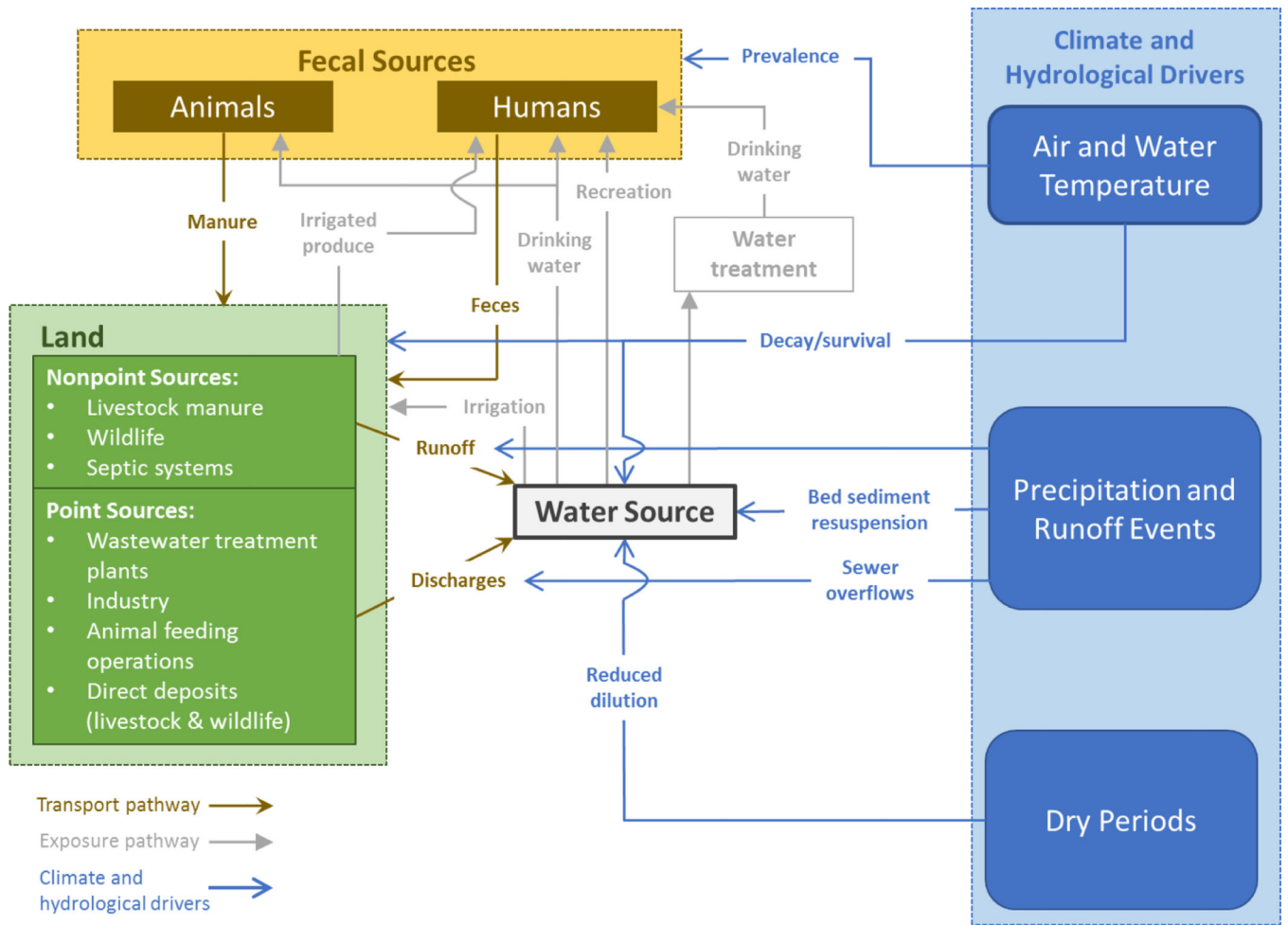
Changes in average annual water body surface cyanobacteria (thousands of cells per mL) for a low growth (plateaus as temperatures reach 26°C) and a high growth scenario (assumes a linear growth rate with changes in temperature) in 2050 (2040–2059) and 2090 (2080–2099) relative to a control scenario. Values for each representative concentration pathway (RCP) are associated with the average results for five future scenarios and are aggregated to the four-digit hydrologic unit code level, weighted by water body surface area. See USEPA (2017) and Chapra et al. (2017) for information about the modeling approach.



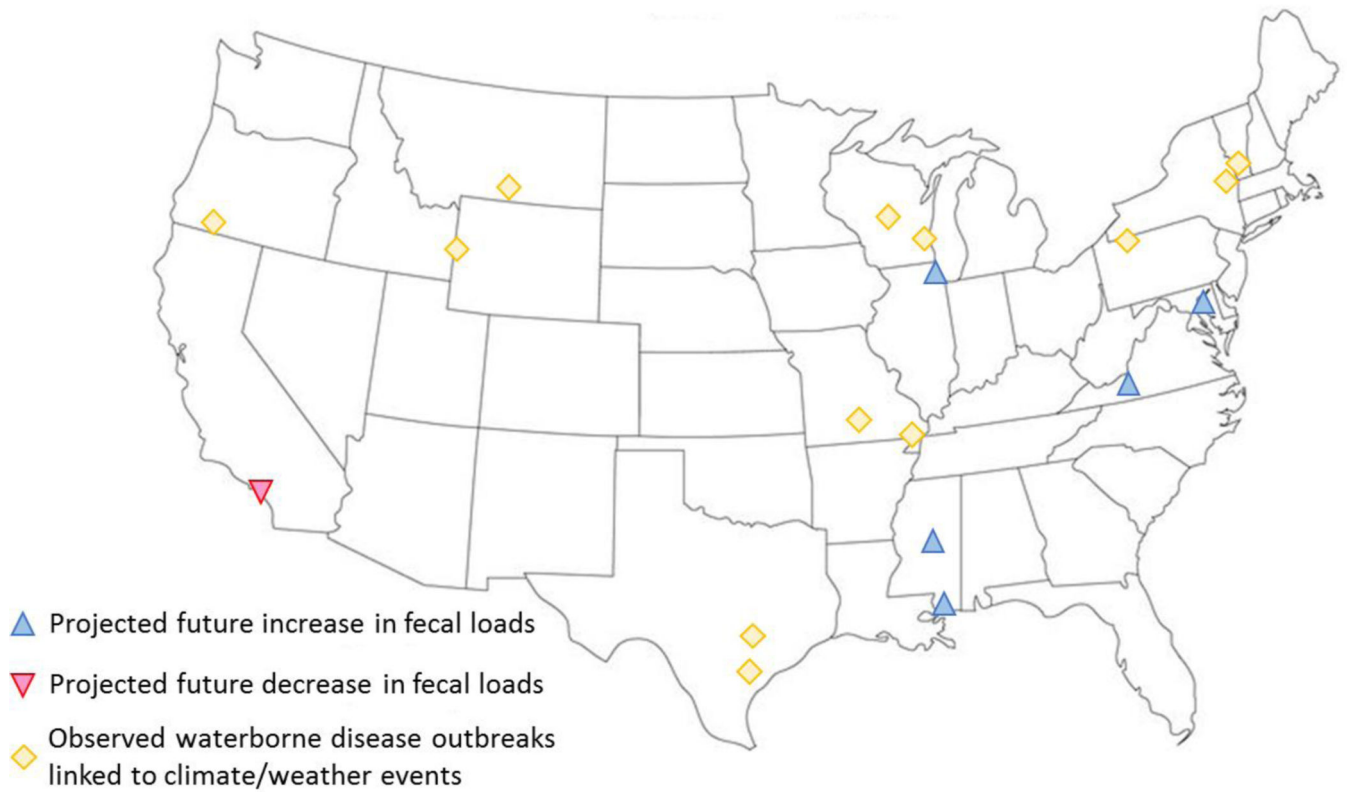
**FIGURE 7.**

Location of watershed modeling studies assessing sediment load responses to future climate change scenarios. Symbols indicate the suggested direction of change (based on ensemble median, annual loads). Studies not reporting an ensemble median (e.g., a range, or sensitivity) are shown as “Direction indeterminate.” Detailed information about each study is provided in the Supporting Information.





**FIGURE 8.** Conceptual model of waterborne pathogens responses to climate and hydrological drivers. Adapted from Coffey et al. (2014) and Hofstra (2011).

**FIGURE 9.**

Location of watershed modeling studies assessing fecal load responses to future climate change scenarios. Projected future symbols indicate the suggested direction of change (based on ensemble median, annual loads). The locations of some observed waterborne illness outbreaks that have been linked to weather events are also given. Detailed information about each study is provided in the Supporting Information.