

Potential Impacts of Climate Change on Groundwater Quality

Zahid Aziz, Ph.D.
Division of Science and Research; NJDEP

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State of New Jersey
Phil Murphy, Governor

**Department of Environmental
Protection**
Shawn M. LaTourette, Commissioner



Division of Science & Research
Nicholas A. Procopio, Ph.D., Director

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Summary

This report synthesizes available scientific literature on the potential responses of different hydrogeological and biogeochemical processes to climate change and discusses how these processes could impact groundwater quality. This review suggests that the effects of climate change are likely to cause ephemeral and long-term impacts on groundwater quality driven by modifications of hydrogeological processes, including precipitation, groundwater recharge, discharge, capacity, and seawater intrusion. These modifications would influence biogeochemical reactions and the ultimate chemical fate and transport of contaminants, and are likely to drive the variability of both anthropogenic and geogenic contaminants.

Below are some specific examples of hydrogeological and geochemical changes and associated groundwater quality changes that are likely to occur as a result of climate change:

1. Climate change is expected to raise groundwater temperature in urban and rural areas. Changes in groundwater temperature would increase mineral solubility and microbiological activities, which has the potential to elevate concentrations of certain major elements such as silica (Si); potassium (K); fluorine (F); heavy metals including copper (Cu), cadmium (Cd), nickel (Ni), lead (Pb), lithium (Li), and zinc (Zn); redox-sensitive elements including arsenic (As), iron (Fe), manganese (Mn), phosphorus (P); and dissolved organic carbon (DOC) in the shallow groundwater.
2. Groundwater recharge delivers electron acceptors (such as oxygen (O_2^{2-}), nitrate (NO_3^-), ferric iron (Fe^{3+}), sulfate (SO_4^{2-}), organic carbon, and pollutants into the aquifer, which impact existing subsurface geochemical conditions and affect the dissolved concentrations of a wide variety of contaminants, including redox-sensitive trace elements. Rising groundwater levels due to increased recharge would saturate the normally unsaturated soil layer, reduce oxygen levels, and mobilize natural trace elements and anthropogenic contaminants stored in the unsaturated layer. Therefore, recharge under changed frequency and intensity of precipitation related to climate change poses increasing threats to groundwater by mobilizing geogenic contaminants (such as As and Mn) and anthropogenic contaminants such as NO_3^- , hydrocarbons, other organic contaminants, emerging contaminants (such as per- and polyfluoroalkyl substances and microplastics), and pathogenic microorganisms into aquifers.
3. Climate change is expected to cause large changes in precipitation frequency and intensity, leading to extreme weather events like more frequent flooding and drought. Flooding can make the surface soil anaerobic and mobilize geogenic contaminants like As, Fe, Mn, P, and other metals such as chromium (Cr), molybdenum (Mo), and vanadium (V) in soils which eventually are transported to shallow aquifers via infiltration and recharge. During droughts, decreased precipitation causes a decline in groundwater recharge and level, which may lead to the mobilization of contaminants such as NO_3^- , SO_4 , and As due to the altered redox condition in the aquifer.

4. In coastal areas, shallow groundwater is expected to respond dynamically to sea-level rise (SLR) leading to seawater intrusion (SWI), rising groundwater levels, and coastal flooding. The increase in salinity by SWI can mobilize metals (As, Cd, Co, Cu, Hg, Pb, Se, V, Ra) from soils/sediments to groundwater through cation exchange and desorption by chloride (Cl) and organic complexes. Similarly, rising groundwater and coastal flooding are likely to affect contaminant concentrations in shallow coastal aquifers.

Climate change will have a broad impact on groundwater quality. However, quantification of the impacts is difficult due to the complex responses of the hydrogeological system to climate variability, range of scenarios associated with climate models, and global socio-economic developments that will determine the scale of changes to the climate system. Denser networks of groundwater observation points for long-term monitoring are essential to provide necessary baseline data and supplement existing data sources. Such monitoring networks would improve the quantification of hydrogeological systems that are likely to be affected by climate change at local to regional scales.

1.0 Introduction

Approximately 33% of the New Jersey public supply water is from groundwater withdrawal and about 13% of New Jersey residents (approximately 1 million people) obtain their drinking water from private wells (Dieter et al. 2018; NJDOH, 2022). Furthermore, groundwater baseflow plays a significant role in maintaining biodiversity and habitat quality for sensitive ecosystems in lakes, rivers, and coastal lagoons (Humphreys 2006; Kløve et al., 2011; Eamus et al., 2015; Havril et al., 2018; Chiloane et al., 2022; KarisAllen et al., 2022; Boulton et al., 2023). As climate change continues to raise global temperatures and modify weather patterns, the effects of climate change can be expected to impact the water cycle (Figure 1) and groundwater resources in a variety of complex ways (Cuthbert et al., 2019a; Allan et al., 2020; Douville et al., 2021; Peters-Lidard et al., 2021; Gascuel-Oudou et al., 2023). Increasing temperature, precipitation, and sea-level are among the major phenomena likely to impact New Jersey's water resources as a direct result of climate change (NJDEP, 2020). These changes will affect the hydrological cycle by altering recharge, groundwater levels, flow processes, and storage (Green et al. 2011; Jyrkama and Sykes 2007; Stuart et al. 2011; Deshmukh et al., 2022; Swain et al., 2022). Climate models indicate a general increase in annual precipitation, flooding associated with extreme precipitation, and storm surges along New Jersey and other northeast states coastlines (Karl et al., 2009; Mallakpour and Villarini, 2015; Bevacqua et al., 2020; NJDEP, 2020; Swain et al. 2020; DeGaetano, 2021, Nazarian et al., 2022; Jong et al., 2023).

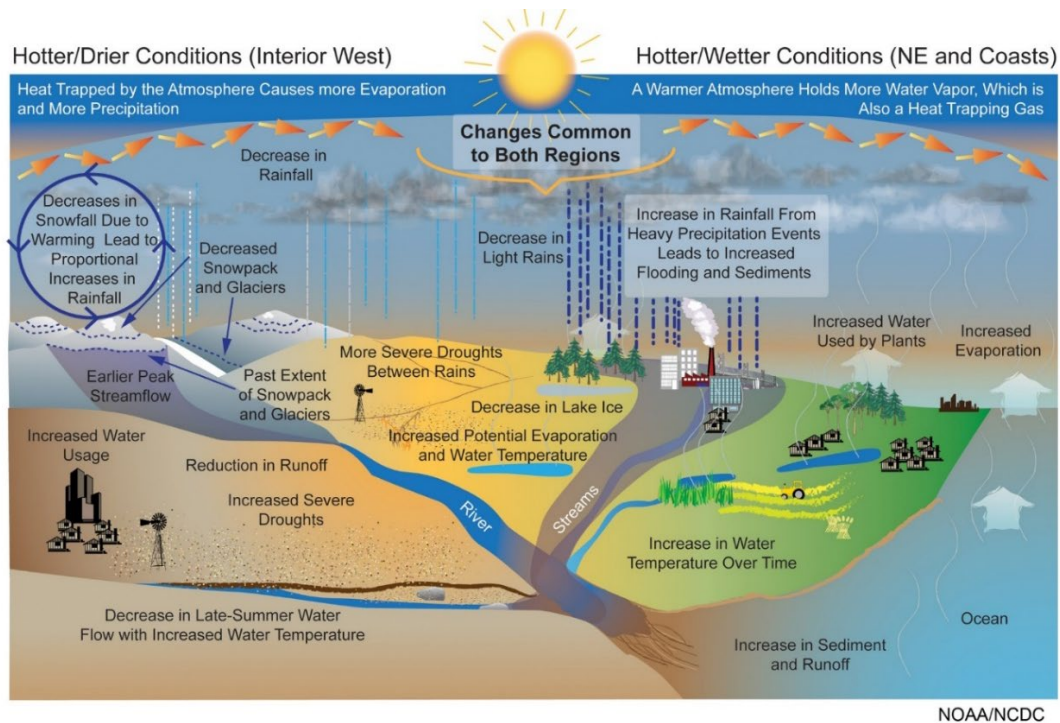


Figure 1. Projected changes in the water cycle. Increasing temperature, precipitation, and sea-level are primary effects of climate change on water resources (Source: Karl et al., 2009).

Notably, a greater proportion of the annual rainfall is expected to occur during fewer events, with longer dry periods between the events (Karl et al. 2008; Melillo et al. 2014; Xue 2022). Decreased precipitation and longer dry periods during summer months may lead to more frequent and extended droughts (Trenberth, 2011; Krakauer et al., 2019; NJDEP, 2020; Polasky et al., 2022). Several studies have explored the potential impacts of climate change on water resources, particularly concerning surface water (Murdoch et al., 2000; Piao et al 2010; Bates et al., 2008; MacDonald GM, 2012; Arnell and Hughes 2014; Haddeland et al., 2014; Mimikou et al., 2000, Duran-Encalada et al., 2017; Cooley et al., 2021; Golub et al., 2022; IPCC, 2022; Xin et al., 2022; Wang et al., 2023). However, knowledge of the potential effects of climate change on groundwater, which is a significant component of the public water supply and water use in New Jersey (NJDEP, 2017) and across the United States (AGI, 2017; Dieter et al., 2017), is still limited (e.g. Bates et al., 2008; Bovolo et al., 2009; Green et al. 2011; Holman et al., 2012; Taylor et al., 2013; Barbieri et al., 2021; Uhl et al., 2022).

The response of groundwater to climate change is an intricate process, as climate change will affect hydrogeological processes and groundwater resources directly and indirectly (Earman and Dettinger, 2011; Green et al. 2011; Taylor et al., 2013; Liesch and Wunsch, 2019; Amanambu et al., 2020; Barbieri et al., 2021; Uhl et al., 2022). Groundwater aquifers are recharged mainly by precipitation or through interaction with surface water bodies (Figure 2), therefore the direct influence of climate change on precipitation and surface water ultimately affects groundwater systems (Kumar, 2016). Recent studies investigating the effects of climate change on

groundwater focus primarily on processes that affect groundwater quantity, such as recharge, groundwater level, discharge, and storage (Eckhardt and Ulbrich, 2003; Dragoni and Sukhija, 2008; Kundzewicz et al., 2008; Döll, 2009; McCallum et al., 2010; Ng et al., 2010; Gunduz and Simsek, 2011; Hanson et al., 2012; Kumar CP, 2016; Condon et al., 2020; Haidu and Nistor, 2020; Riedel and Weber, 2020; Al Atawneh et al., 2021; Arias et al., 2021; Ascott et al., 2022; Boumaiza et al., 2022; Dehghani et al., 2022; Swain et al., 2022; Gordu et al., 2023). However, studies of the impacts of climate change on groundwater quality remain limited, even though water quality, and not quantity, may be a more limiting factor for future potable usage (Green et al. 2011; Gurdak et al. 2012; Ayotte et al., 2015; Zhou et al., 2021; Geris et al., 2022; Lapworth et al., 2022; McMahon et al., 2022; Warziniack et al., 2022).

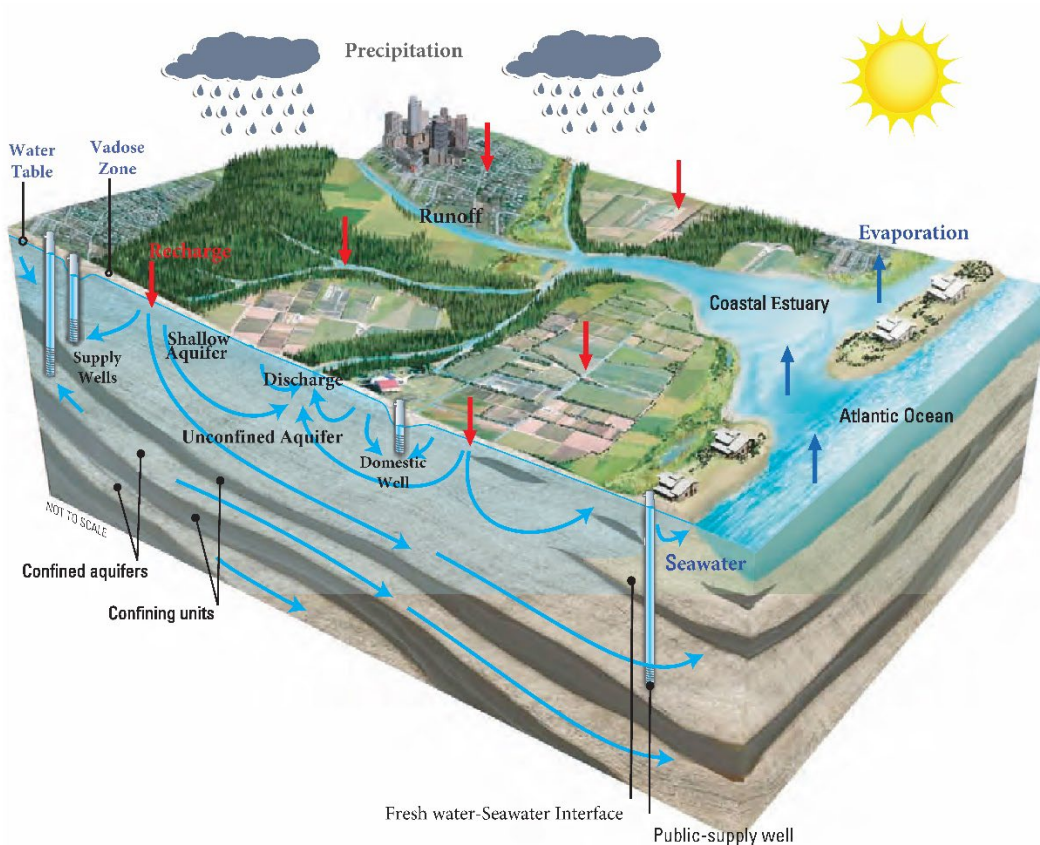


Figure 2. – Northern Atlantic Coastal Plain surficial aquifer system and its interactions with the climate. Groundwater recharge occurs from precipitation and surface water bodies. Groundwater flows through aquifers from recharge areas to discharge areas. Climate change affects groundwater resources by modifying processes such as recharge, groundwater level, discharge, and storage. The location of fresh water-seawater interface is driven by sea level rise and groundwater withdrawal (figure: modified from Denver et al., 2015).

Available studies show that climate change is likely to cause groundwater temperatures to increase (Menberg et al., 2014, Riedel T, 2019; Epting et al., 2020), seasonal declines in recharge and water levels from frequent and prolonged droughts (NJDEP, 2002; Schreiner-McGraw, et al. 2021), rising water level in response to enhanced recharge associated with increasing precipitation and flooding (Bjerklie, et al., 2012; Knott et al., 2019; Hemmerle and Bayer, 2020), and seawater intrusion related to sea-level rise (NJDEP, 2020; Jasechko et. al., 2020).

In general, groundwater quality is controlled by the amount of rainfall, the chemical nature of recharge water, surface water resources, and geochemical processes occurring within the aquifer (Etikala et al., 2019). Therefore, the sensitivity of groundwater to climate change depends, in part, on the depth of the groundwater table, where groundwater in shallow (typically less than 30 feet deep from the ground surface) unconfined aquifers (as depicted in Figure 2) is expected to be more responsive to climate change than groundwater at deeper depths or in a confined aquifer systems (Healy and Cook, 2002; Nolan and Hitt, 2006; Gunduz and Simsek, 2011; Klove et al., 2014; Menberg et al., 2014; Epting et al., 2021; Hare et al., 2021; Lam et al., 2021; Malakar et al., 2021; Benz et al., 2022; KarisAllen et al., 2022; Seidenfaden et al., 2022). Correspondingly, any modification to hydrogeological and geochemical processes, such as groundwater temperature, recharge, groundwater level (water table) fluctuation, sea-level change, and seawater intrusion would likely cause more pronounced changes to shallow groundwater quality. Shifts in these geochemical and hydrogeological conditions within an aquifer can cause the mobilization of both geogenic and anthropogenic contaminants from primary minerals or initiate their release from soils and sediments into groundwater, where they pose a much larger risk to human and ecological health (Fakhreddine et al., 2021; Zhao et al., 2023). For example:

- Changes in groundwater temperature would affect the solubility of minerals, impacting the groundwater chemistry of shallow aquifers.
- Organic matter (organic carbon) and/or electron acceptors such as O_2^{2-} , NO_3^- , and SO_4^{2-} introduced into the aquifer system via recharge would impact existing subsurface geochemical conditions through redox reactions that control the dissolved concentrations of a wide variety of contaminants, including redox-sensitive trace elements such as As, Fe, Mn and P.
- Rising groundwater levels would more frequently saturate the unsaturated soil layer, reduce oxygen levels, and mobilize and increase natural trace elements and anthropogenic contaminants stored in the unsaturated layer. Similarly, a decline in groundwater level would concentrate contaminants and/or change geochemical (redox) condition in the aquifer, which would affect the groundwater quality.
- Increased groundwater salinity (salinization) due to seawater intrusion would also increase the trace metal mobility significantly due to its influence on cation exchange equilibrium.

Despite the potential changes to groundwater quality from climate change-induced modifications of hydrogeological processes, the extent and rate of interactions between climate processes, hydrogeological and geochemical processes are not well understood (Arias et al., 2021). Deteriorating groundwater quality could pose long-term technical and economic challenges for groundwater and drinking water management and soil and groundwater remediation efforts.

This report is an assessment of hydrogeological and geochemical processes in shallow unconfined aquifers that are likely to be affected by climate change through a variety of mechanisms, including groundwater temperature change related to global air temperature change, changes in groundwater recharge rates and groundwater levels associated with changes in spatiotemporal precipitation patterns, as well as seawater intrusion (salinization). This report synthesizes existing literature and publications on potential responses of different hydrogeological and geochemical processes to climate change and discusses how these processes would impact groundwater quality based on current scientific knowledge of hydrogeology and geochemistry.

2.0 Literature Review and Methodology

This review synthesizes information from a variety of scientific material including books, peer-reviewed journals, and reports by government and state agencies. There are numerous studies published in the past several decades assessing the complexity of climate change and its impact on water resources. Some have discussed possible modifications of hydrogeological and geochemical processes in response to different climate scenarios. However, little is known about how such alterations of hydrogeological and geochemical processes may have already changed and/or will change groundwater quality in a changing climate. A comprehensive review of the literature summarizing and identifying hydrogeological and geochemical processes and their potential to change groundwater quality is an invaluable contribution to our current scientific understanding of the impacts of climate change.

3.0 Impacts on Groundwater Quality

There are many potential impacts of climate change on groundwater quality through modification of existing hydrogeological and geochemical processes, and it would be unrealistic to comprehensively include all the impacts in this review. Rather, the impacts that are relevant to the state of New Jersey and other northeast states are the primary focus of this review. Quantification of these impacts is difficult due to the complex responses of the hydrogeological system to climate variability, range of scenarios associated with climate models, and global socio-economic developments that will determine the scale of changes to the climate system. The extent and rate of changes will depend on the degree of adaptation and reduction of future carbon dioxide (CO₂)

emissions and hydrogeological system response. Quantitative analysis of climatic impacts on groundwater quality will not be explored in detail in this review, rather, this paper will focus on the broad implications of climate change on groundwater resources. Within the scope of this report, descriptions of different hydrogeological and geochemical processes, associated alterations due to climate change, and the potential effects of these altered processes on the groundwater quality are organized into the following categories: (1) groundwater temperature; (2) groundwater recharge; (3) extreme events (flood and drought); (4) sea-level rise (seawater intrusion, groundwater level rise, and coastal flooding); and (5) climate change impacts on anthropogenic contaminants, their transport, and remediation.

3.1 Groundwater Temperature and Quality

Shallow groundwater temperatures are mostly a function of the land surface temperature (Kurylyk et al., 2015; Benz et al., 2017), and are influenced primarily by diffusive or advective (with the infiltrating/recharge water) mechanisms (Kurylyk et al., 2015). However, fluctuations in groundwater temperature are attenuated with increasing depth or with increasing distance from recharge locations (Anderson, 2005; Bhaskar et al., 2012; Kurylyk et al., 2013; Kurylyk et al., 2015; KarisAllen et al., 2022). Although urbanization has a substantial influence on groundwater temperature (Taylor and Stefan, 2009; Mueller et al., 2018; Tissen et al., 2019; Böttcher and Zosseder, 2022), several natural factors, such as the mean temperature above the ground surface, thickness and thermal conductivity of the vadose zone (unsaturated soil layer), precipitation, infiltrating surface waters, and regional groundwater flow affect the groundwater temperature in undeveloped, rural, and urban areas (Calvache et al., 2011; Benz et al., 2017; Burns et al., 2017; Epting et al., 2020).

Various studies have reported annual groundwater temperature increases between 0.01°C and 0.13°C over multiple decades at shallow depths (< 30 ft from the surface) in less-disturbed rural areas (Maxwell and Kollet, 2008; Park et al., 2011; Bloomfield et al., 2013; Kurylyk et al., 2014; Kurylyk et al., 2015; Menberg et al., 2014; Colombani et al., 2016; Mastrocicco et al., 2018; Riedel T, 2019; Hemmerle and Bayer, 2020; Egidio et al. 2022). This subtle increase in groundwater temperature could be attributed to the magnitudes of the long-term increase in mean groundwater temperature being dampened by up to 70% compared to the change in surface temperature (Menberg et al., 2014). Furthermore, the mean subsurface temperature's response to the surface temperature usually lags for several years (Menberg et al., 2014; Bucci et al., 2020). A set of global climate models (GCM) and numerical simulations predict that the groundwater temperature at shallow depth (<30 ft) may increase within a range of 1.2–3.3°C by 2080 for a range of potential future climate changes (low to high CO₂ emissions) (Gunawardhana and Kazama, 2011; Arias et al., 2021; KarisAllen et al., 2022). These models demonstrate the depth dependency of warming, where temperature increases are more pronounced at shallower depths for a given timeframe and climate scenario (KarisAllen et al., 2022). Also, an analytical heat transfer model has projected that

for a very high CO₂ emission (2 x current emission) scenario, the mean annual temperature of shallow groundwater could rise by about 2.6–3.7°C by the end of 2100 in the Minneapolis/St. Paul area (Taylor and Stefan, 2009). The same model has also projected that groundwater temperature may increase up to 4°C when the effect of urban ground surface temperature on groundwater is considered. However, the very high CO₂ emission scenario, which depicts a worst-case scenario in the future, may not occur because of current global policies to reduce carbon emission and increase use of renewable energy (Hausfather and Peters, 2020).

A likely way that the groundwater would be affected by the surface temperature warming is through recharge especially from warmer surface water bodies such as lakes and streams/ivers (Epting et al., 2020). For example, a significant temperature increase (1°C) in groundwater over a couple of decades was observed in alluvial aquifers on the Swiss Plateau, which are recharged by river-bank infiltration (Figura et al., 2011). An increasing trend in surface water temperatures (both in lakes and streams/ivers) has been observed over recent decades in the USA and at a global scale (Michel et al., 2020; Woolway et al., 2020; Hare et al., 2021). Historical observations indicate that surface water temperatures have increased worldwide at a global average rate of 0.33°C per decade which is similar to or in excess of air temperature trends (Michel et al., 2020; Woolway et al., 2020). Several models predict an increase in median annual river temperatures ranging between 0.9°C for low-emission and 2-3.5°C for high-emission scenarios at the end of the century (2080–2090) (van Vliet et al., 2013; Piotrowski et al., 2021; Michel et al., 2022). Therefore, the source of groundwater recharge and the future climate conditions are likely to play an important role in determining groundwater temperature and subsequent groundwater quality changes in the future.

Elevated temperature can cause changes in biogeochemical processes, such as mineral weathering, chemical adsorption and desorption, gas solubility, and microbial redox processes in the subsurface environment, leading to groundwater quality changes (Brons et al., 1991; Appelo and Postma, 2005; Bates et al., 2008; Banks, 2012; Bonte et al., 2013a, 2013b, 2013c; Jesušek et al., 2013a; Jesušek et al., 2013b; Bonte et al., 2014; Kurylyk et al., 2014; Menberg et al., 2014; Possemiers et al., 2014; Saito et al., 2016; Lipczynska-Kochany, 2018; Riedel, 2019). In general, occurrences of elevated concentrations of Si, K, and F in groundwater are associated with higher temperatures and are due to the dissolution of minerals in aquifers (Welch and Ullman, 2000; Elango and Kannan, 2007; Bonte et al., 2013b; Saito et al., 2016; Riedel, 2019). Also, elevated temperatures decrease O₂ saturation through increased microbial respiration rates and reduced solubility and pH (Garcia and Gordon 1992; Yagasaki et al., 2006; Foulquier et al., 2009; Figura et al., 2013; Munz et al., 2019; Riedel, 2019). For example, an increase of 1°C in groundwater temperature can result in a 4% decline in O₂ saturation and a minimal decline in pH (Riedel, 2019). Consequently, increased temperature and lowered pH would increase the mobilization rate of heavy metal, such as Cu, Cd, Ni, Pb, Li, and Zn (McLean, 1992, Chuan et al., 1996; Li et al., 2013; Chen et al., 2016; Zhang et al., 2018; Lüders et al., 2020; Qasem et al., 2021; Iordache et al., 2022). Thermally-induced

desorption and/or mobilization of organic compounds, such as natural organic matter (OM), dissolved organic carbon (DOC), or organic contaminants from soils and sediments, has also been observed at higher temperatures (Karickhoff et al., 1979; Brons et al., 1991; Ten Hulscher and Cornelissen, 1996; Delle Site, 2001; Kaiser et al., 2001; Plazinski et al., 2009; Wang et al., 2010; Bonte et al., 2013b; Saito et al., 2016; Lipczynska-Kochany, 2018; Riedel T, 2019). Warmer temperatures within aquifers stimulate the microbial community (Ghiorse and Wilson, 1988; Price and Sowers, 2004; Jesubek et al., 2013a; Westphal et al., 2017; Retter et al., 2021) which promotes reducing geochemical conditions fueled by DOC (Selim, 2012). As an aquifer becomes geochemically reducing, redox-sensitive elements such as As, Fe, Mn, and P will be released in the groundwater by reductive dissolution of iron oxides and other minerals (Chapelle et al., 1955; McMahon and Chapelle, 2008; Barringer et al., 2010; Mladenov et al., 2010; Bonte et al., 2013b; Mailloux et al., 2013; McMahon et al., 2018). Moreover, temperature has significant impacts on the occurrence and rate of NO₃⁻, SO₄²⁻, and Fe reduction (Weber et al., 2010; Jesubek et al., 2013b; Possemiers et al., 2014; Shirokova et al., 2016; Schilling et al., 2019).

Changes in groundwater temperatures are likely to be subtle (1-2°C) at the end of the century (2080-2090) in response to surficial warming associated with climate change. Groundwater warming driven by climate change is likely to cause elevated concentrations of some major elements (Si, K, and F), heavy metals (Cu, Cd, Ni, Pb, Li, and Zn), redox-sensitive elements (As, Fe, Mn, and P) and DOC in shallow groundwater. However, groundwater quality change is a slow and diffuse process, which would be apparent only after a prolonged period of time if the surface temperature continues to rise due to climate change.

3.2 Groundwater Recharge and Quality

Natural groundwater recharge occurs through two major pathways depending on hydrogeologic setting (Alley et al. 2002; De Vries and Simmers, 2002; Healy, 2010; Amanambu et al., 2020; Li et al., 2021) - diffuse rain-fed recharge and focused recharge via leakage from surface waters (stream/river, lakes, etc.). Diffuse recharge occurs through direct infiltration from rainfall, snowmelt, and surface ponding that occurs uniformly over a large area (Stephens, 1994; Scanlon et al., 2002; Small, 2005; Ng et al., 2009; Barron et al., 2012; Wirmvem et al., 2017; Zhu et al., 2020; Schilling et al., 2021). Alternatively, focused or localized recharge occurs when fast and concentrated water infiltrates from discrete surface water bodies and drains through preferential flow paths (Lawrence and Upchurch, 1982; Winter, 1983; Scanlon et al., 2002; Wirmvem et al., 2017; Meixner et al., 2016; Ren et al., 2019; Hughes et al., 2021), particularly in areas with karst, fractured crystalline bedrock, and glaciated bedrock/outwash formations (Hartmann et al., 2017), similar to the geologic setting in the physiographic provinces of norther New Jersey. Precipitation (the distribution, amount, intensity, and timing) and evapotranspiration losses are the climate elements that primarily affect groundwater recharge (McCallum et al., 2010; Ng et al., 2010; Beigi and Tsai, 2015; Zomlot et al., 2015; Thomas et al., 2016; Jayakumar and Lee, 2017;

Al Atawneh et al., 2021), regardless of the recharge pathway. In response to warming temperatures, increased evapotranspiration can decrease the amount of precipitation that runs off as surface water or infiltrates to the subsurface as recharge leading to declined groundwater levels (Condon et al., 2020). Thus, heavy precipitation may not always increase recharge unless it exceeds evapotranspiration (Scanlon et al., 2005; Adane et al., 2019; Amanambu et al., 2020). Moreover, land use/land cover and underlying geology determine, in large part, whether a water surplus (precipitation minus evapotranspiration i.e., runoff) would infiltrate and be stored in the subsurface (Nolan et al., 2007; Reese and Risser, 2010; Taylor et al., 2013; Kløve et al., 2014; Mohan et al., 2018; Riedel and Weber, 2020). Some studies have predicted increased groundwater recharge due to projected increases in precipitation in the southwestern (McKenna and Sala, 2018; Borchardt, 2022), northern high plain (Meixner et al., 2016; Lauffenburger et al., 2018; Kishawi et al., 2022), and northeastern/north central (Ng et al., 2010; Crosbie et al., 2013; Rossman et al., 2014; Beigi and Tsai, 2015; Tillman et al., 2016; Bedaso et al. 2019) regions of the United States.

Several studies have shown that extreme (heavy) rainfall events are likely to become a significant contributor to groundwater recharge for a range of environments either by exceeding field capacity (pore-matrix flow) or due to bypass flow through macropores (Owor et al., 2009; Jasechko and Taylor, 2015; Liu et al., 2016; Thomas et al., 2016; Zhang et al., 2016; Asoka et al., 2018; Cuthbert et al., 2019; Bargués-Tobella et al., 2020; Goni et al., 2021; MacDonald et al., 2021; Boas and Mallants 2022; Guillaumot et al., 2022; Stigter et al., 2022; Taylor et al., 2022). For example, a Soil Water Balance Model (SWBM) estimated a significant increase in groundwater recharge driven by extreme precipitation from 1950 to 2010, which contributed more than 60% of the annual total groundwater recharge in the Northern High Plains (NHP) Aquifer, USA (Zhang et al., 2016). Extreme (heavy) rainfall creates temporary surface water bodies where focused recharge occurs (Crosbie et al., 2012; Taylor et al., 2013a; Reiss et al., 2019; Zheng et al., 2019; Arias et al., 2021; Seddon et al., 2021; Boas and Mallants, 2022; Taylor et al., 2022). The frequency and intensity of heavy (extreme) precipitation events are projected to continue to increase over the 21st century in response to climate change in New Jersey and other northeast states (Easterling et al., 2017; DeGaetano, 2021, Nazarian et al., 2022; Jong et al., 2023). Therefore, enhanced groundwater recharge is also anticipated in New Jersey, based on the projected increases in precipitation associated with climate change.

Recharge can alter the groundwater quality in various ways, including (i) diluting contaminants through water infiltrated from precipitation (Castaldo et al., 2021; Maroubo et al., 2021), (ii) transporting chemical compounds stored in the vadose zone to the water table (Nolan et al., 2003), (iii) introducing/delivering electron acceptors, organic carbon, and pollutants into aquifers (McGuire et al., 2005; McMahon et al., 2011; Schaefer et al., 2016; Arora et al, 2019; Curzio, 2019) and (iv) raising the groundwater level that limits the thickness of the vadose zone and mobilizes contaminants accumulated in vadose zone (Scholl et al., 2006; Lasagna et al., 2020).

Increased recharge may reduce contaminant concentrations through dilution or mixing with fresh water from precipitation and surface water sources. There is some evidence of nitrate attenuation by dilution from infiltrating groundwater recharge (Nolan et al., 2003; Taylor, 2003; Castaldo et al., 2021), though increased groundwater nitrates were attributed to increased recharge by the same studies. Similarly, Maroubo et al. (2021) observed an inverse relationship between heavy metal concentrations (cadmium and lead) and precipitation suggesting a dilution effect of increased recharge from precipitation. Other studies have shown lower concentrations of groundwater arsenic are associated with dilution and inhibition by oxic water stemming from high recharge (Aziz et al., 2008; Brikowski et al., 2014; Singh et al., 2020).

Increased recharge can adversely affect groundwater quality when recharge water causes downward mobilization or transportation of chemical compounds and pollutants stored in the vadose zone from past and present land use practices (Gurdak et al., 2007; Dragoni and Sukhija, 2008; Earman and Dettinger, 2011; Taylor et al., 2013; Klove et al., 2014; Oostrom et al., 2016; Zhou et al., 2016; Arora et al., 2019; Libera et al., 2019; Pétré et al., 2021). Enhanced recharge can cause the flushing of salts from the unsaturated zone (Amiaz et al., 2011) and increase groundwater salinity (Sugita & Nakane, 2007). For example, Małeckki and Matyjasik (2002) reported a nearly tenfold increase in the average total dissolved solids concentration as rainwater infiltrated through the ground surface into the groundwater supply. In Riverton, Wyoming, three years of groundwater monitoring data indicated that Na, SO₄, and Cl, present across the site in a silt layer, were introduced into the shallow aquifer during large recharge events (USDEOE, 2019).

Recharge events deliver electron donors, primarily DOC, and electron acceptors, such as O₂²⁻, NO₃⁻, SO₄²⁻, as well as anthropogenic organic pollutants into aquifers which can cause groundwater redox reactions and water quality changes (McGuire et al., 2005; McMahan et al., 2011; Arora et al., 2016; Schaefer et al., 2016; Yabusaki et al. 2017; Arora et al, 2019; Curzio, 2019). The redox process involves electron transfer reactions that release energy via the oxidation of labile organic carbon or inorganic compounds (electron donors) by microorganisms coupled to the reduction of electron acceptors. Therefore, redox processes control the release and/or sequestration of anthropogenic contaminants (Christensen et al., 1994; Kappler and Haderlein, 2003; Borch et al., 2010; Filippini, 2017; Curzio, 2019) and naturally occurring trace elements (As, Fe, Mn, P, S, Cd, Cr, Cu, Co, Se, Pb, Hg, and U), including their chemical speciation, bioavailability, toxicity, and mobility (Smedley and Kinniburgh 2002; McMahan et al., 2008; Du Laing et al., 2009; Borch et al., 2010; Arora et al., 2016; Wang et al., 2016; McMahan et al., 2018; Riedel and Kübeck, 2018; Kubier et al., 2019; Coyte and Vengosh, 2020; Hou et al., 2020; Dong et al., 2022; Huang et al., 2023a; Liu et al., 2023).

The most common electron donor driving redox processes in aquifers is DOC, which is either derived from sedimentary particulate organic matter in aquifers (Moore, 2018; Pracht et al., 2018; Baran et al., 2019); delivered into aquifers by recharge from vadose zones (McDonough et al., 2020a; Li et al., 2023a) and surface water bodies,

such as ponds, lakes, streams, and rivers (Chen et al., 2010; Farnsworth and Hering, 2011; Inamdar et al., 2012; Shen et al., 2014; Nghiem et al., 2019; Yang et al., 2020; Gao et al., 2023; Wen et al., 2023); or anthropogenic sources, such as petroleum hydrocarbon storage facilities, and industrial solvent disposal (Kelso, 2018; Podgorski et al., 2018; Li et al., 2021; Zhai et al., 2021; Cao et al., 2023). Natural DOC concentrations in surface water derived from decaying plant material have shown an increasing trend over the last few decades attributed to increasing temperatures related to climate change (Evans et al., 2005; Jennings et al., 2009; Naden et al., 2010; Larsen et al., 2011; SanClements et al., 2012; Pagano et al., 2014; Huntington et al., 2016; Ren et al., 2016; Lipczynska-Kochany, 2018; McDonough et al., 2020b; Wen et al., 2020; Tiwari et al., 2022; Hou et al., 2023; Ming et al., 2023) ultimately leading to an increased supply of DOCs into aquifers (McDonough et al., 2020b).

Peatlands located along coastal areas around the world (Yu et al., 2011; Page and Baird, 2016; Zhong et al., 2020; Loisel et al., 2021) and in New Jersey (Johnson, 1985; Lynn and Karlin, 1985; Karlin and Andrus, 1988; Mandernack et al., 2000) serve as a significant source of DOC in groundwater (Bartsch and Moore, 1985; Broder et al., 2017; Walpen et al., 2018; Sienkiewicz et al., 2019; Lim et al., 2022; Prijac et al., 2022; Rosset et al., 2022). It is expected that the DOC supply from peatlands into aquifers will be amplified due to increased decomposition of peat under elevated temperature (Kane et al., 2014; Ritson et al., 2014; Dieleman et al., 2016; Wang et al., 2018; Leng et al., 2019; McDonough et al., 2020a and 2020b; Xu et al., 2020). The surface-derived DOC in groundwater via recharge is likely to cause microbial reduction of metal (Fe and Mn) oxyhydroxides in aquifers, leading to mobilizations of redox-sensitive elements such as As, Mn, Fe, P, and Ni (Sharif et al., 2008; Du Laing et al., 2009; Mailloux et al., 2013; Lawson et al., 2016; Postma et al., 2017; Rinklebe and Shaheen, 2017; Nghiem et al., 2019; Coyte and Vengosh, 2020; Qiao et al., 2020; Tweed et al., 2020; Akintomide et al., 2021; Huang et al., 2023a; Liu et al., 2023). Arsenic-rich glauconitic sediment and metal oxyhydroxides are present in the aquifers of the inner coastal plain of New Jersey, which are likely the source of dissolved As in the groundwater and are mobilized due to reducing geochemical conditions driven by supplied DOCs (Vowinkel et al., 2001; Barringer et al., 2010 and 2013; Percy et al., 2011; Mumford et al., 2012). This might explain the exceedance of the New Jersey drinking-water maximum contaminant level for As observed in some private wells (Flanagan et al., 2016; Giri et al., 2022; Seliga et al., 2022).

Electron acceptors such as O_2^{2-} , NO_3^- , and SO_4^{2-} are also transported into aquifers via recharging water, promoting oxic geochemical conditions and groundwater quality changes (Kumar and Riyazuddin, 2012; Schaefer et al., 2016; Haugen et al., 2021). In oxic conditions, trace elements such as As, Mn, P, and Fe are stable due to the formation of minerals such as Fe-arsenates and Fe-hydroxides (Fortin and Langley, 2005). High levels of NO_3^- are observed in shallow groundwater throughout New Jersey due to recharge derived from agriculture areas (Böhlke, 2002a; Nolan et al., 2003), which can lead to oxidation of pyrite (Juncher Jørgensen et al., 2009; Torrentó et al., 2011; Zhang et al., 2012; Hu et al., 2020). The reduction of NO_3^- coupled with pyrite oxidation has

been documented as an important process limiting the movement of NO_3 in an agriculturally impacted glacial aquifer in Minnesota, as well as a coastal-plain aquifer in Maryland (Böhlke et al., 2002b and 2007). However, the oxidation of As-bearing sulfide minerals, including arsenopyrite can lead to mobilization of As in shallow groundwater (Welch et al., 2000; Hering and Kneebone, 2002; Foley and Ayuso, 2008; Garelick et al., 2009; Smedley and Kinniburgh, 2013; Houben et al., 2017; Asta et al., 2019; Erickson et al., 2021; Stolze et al., 2022; Zambito et al., 2022), which is a significant mineral source of high levels of As in the groundwater of New Jersey (Serfes et al., 2003 and 2010; NJGS, 2004). Also, geogenic metals such as Ni, Cd, and U can be released into groundwater during NO_3 reduction (Houden et al., 2017; van Berk and Fu, 2017; Riedel and Kübeck, 2018; Kubier et al., 2019; Riedel et al., 2022; Westrop et al., 2023). Westrop et al (2023) demonstrated that the influx of pore water NO_3 oxidized and mobilized naturally occurring U in High Plains alluvial aquifer in Central Nebraska. Conversely, SO_4 chemistry controls the availability of metals such as Cd, Cu, Ni, and Zn by forming metal sulfides (Du Laing et al., 2009), and depending on the geochemical conditions, SO_4 plays an important role in controlling As concentrations in groundwater. Arsenic concentrations are generally low when SO_4 levels are high (Aziz et al., 2017; Li et al., 2018), suggesting either oxidizing conditions or simultaneous SO_4 and Fe-reducing conditions that are favorable for sequestering As and Fe by forming arsenic sulfide minerals (O'Day et al., 2004; Buschmann & Berg, 2009; Kirk et al., 2010; Burton et al., 2014; Nghiem et al 2019). However, in oxic geochemical conditions, these precipitated sulfide minerals can be oxidized, becoming a source of As, Fe, and other metals (Battistel et al., 2021). Also, sulfides from sulfate reduction can indirectly mobilize arsenic by reducing Fe and As (Rochette et al., 2000; Poulton et al., 2004; Saalfeld & Bostick, 2009; Burton et al., 2014; Liu et al., 2022a; Zhou et al., 2022) and lead to the formation of complexes that dissolve arsenic (Suess and Planer-Friedrich, 2012; Feng et al., 2023a; Nghiem et al., 2023; Planer-Friedrich, 2023; Yang et al., 2023). Similarly, ammonium-generating bacteria can increase As concentration in groundwater by the reduction of Fe-oxides and As(V) to more mobile As(III) during nitrogen-metabolizing (NO_3 transformation) processes (Gao et al., 2021; Jiang et al., 2022, Feng et al., 2023a).

In general, groundwater levels respond quickly to the groundwater recharge rate. For shallow groundwater systems, recharge from precipitation influences the position of the water table (Viswanathan, 1983; Healy and Cook, 2002; van Gaalen et al., 2013; Hussain et al., 2022). The unsaturated vadose zone, located between the ground surface and the water table (Nimmo, 2006), controls the migration of contaminants to the water table due to occurrences of distinct physical and biogeochemical processes in this zone (Hopmans and van Genuchten, 2005; Holden and Fierer, 2005; Arora et al., 2019; Li et al., 2022). Therefore, changes in groundwater level that limit the thickness of the vadose zone can significantly influence the transport of DOC, electron acceptors, pollutants, (Hopmans and van Genuchten, 2005; Brusseau et al., 2013; Lusk et al., 2017; Voisin et al., 2018; Arora et al., 2019; Lebon et al., 2023), and microorganisms, including pathogenic bacteria, viruses, and protozoan

parasites, to aquifers (Brennan et al., 2010; Bradford et al., 2013; Mantha et al., 2017; Alegbeleye and Sant'Ana, 2020; Humphrey et al., 2021; Zhang et al., 2022).

The vadose zone tends to retain DOC because of biogeochemical processes occurring during vertical transport, and retention efficiency depends on the vadose zone thickness (VZT), where DOC retention increases with VZT. (Pabich et al., 2001; McMahon and Chapelle, 2008; Mermillod-Blondin et al., 2015). Therefore, the supply of DOC to the aquifer would increase when groundwater levels are raised via enhanced recharge, reducing the VZT and causing changes in groundwater geochemical (redox) condition and quality. Similarly, a thinner vadose zone could increase the risks of microbial and chemical contamination of aquifers (Tedoldi et al., 2016; Humphrey et al., 2021). Indeed, microbial contamination of groundwater and outbreak of waterborne diseases linked to recharge and vadose zone thickness have been observed in both low- and high-income countries (Taylor et al., 2009; Rao et al., 2013; Engström et al., 2015; Allen et al., 2017; Chuah and Ziegler, 2018; Alegbeleye and Sant'Ana, 2020). Some studies showed nitrate concentrations in groundwater were directly related to vadose zone thickness (Turkeltaub et al., 2018; Juntakut et al., 2019). Additionally, a rising groundwater level poses the threat of groundwater contamination by saturating the vadose zone, releasing contaminants that accumulate in unsaturated soil layers. This may include the mobilization of natural elements (Gurdak et al., 2007; Mills et al., 2011; McClain et al., 2019; Jarsjö et al., 2020), anthropogenic contaminants such as NO₃ (Zhou et al., 2016; Weitzman et al., 2022), hydrocarbons and other organic contaminants (Conrad and DePaolo, 2004; McGuire et al., 2005; Molins et al., 2010; Wellman et al., 2012; Brusseau et al., 2013; Mackay et al., 2018; Mattia et al., 2020), emerging contaminants such as Per- and polyfluoroalkyl substances (PFAS) (Brusseau et al., 2020; Rovero et al., 2021; Sharifan et al., 2021; Guo et al., 2020a; Lyu et al., 2022; Nickerson et al., 2023), microplastics (Ren et al., 2021; Liu et al., 2022; Feng et al., 2023; He et al., 2023; Ranjan et al., 2023), and pathogens/viruses (Chuah and Ziegler, 2018; Alegbeleye and Sant'Ana, 2020).

Available studies indicate that PFAS tends to accumulate primarily in the soil of the vadose zone or at the air-water interface because of their sorption and surfactant properties (Sepulvado et al., 2011; Xiao et al., 2015; Weber et al., 2017; Brusseau and Van Glubt, 2019; Nickerson et al., 2020; Sharifan et al., 2021; Gnesda et al., 2022; Lyu et al., 2022; McMahon et al., 2022). PFAS accumulated in the vadose zone can act as a reservoir for long-term groundwater contamination (Barzen-Hanson et al., 2017; Weber et al., 2017; Brusseau et al., 2020; Sharifan et al., 2021; Guo et al., 2020a; McMahon et al., 2022). Thus, PFAS concentrations in groundwater are closely related to rising and falling groundwater levels, indicating mobilization of accumulated PFAS to aquifers during recharge events (Cáñez et al., 2021). Therefore, enhanced recharge due to climate change makes the groundwater more susceptible to PFAS contamination. Furthermore, contaminated surface water bodies such as rivers/lakes serve as additional sources of PFAS in groundwater (Liu et al., 2019b; Mao et al., 2023). PFAS found

in New Jersey's soils, surface water, groundwater, and fish tissues (Procopio et al., 2017; McCord et al., 2020; Goodrow et al., 2020; Washington et al., 2020), are potential sources for groundwater contamination.

Microplastics (MPs), an emerging contaminant with the potential for toxic effects on human health and the environment (Chia et al., 2021; O'Kelly et al., 2022), have also been identified in groundwater systems (Kim and Lee 2020; Chia et al., 2021; Ren et al., 2021; Esfandiari et al., 2022; Samandra et al., 2022; Shi et al., 2022; Viaroli et al., 2022; Wu et al., 2022). Primary sources of MPs in groundwater include recharge from agricultural soils due to application of plastic mulches and biosolids (Wanner, 2021, Cha et al., 2023; Huang et al., 2023; Moeck et al., 2023), surface water bodies such as rivers/lakes (Severini et al., 2022; Gustavus, 2023; Shu et al., 2023), and urban infrastructure (stormwater management systems, septic tanks, wastewater treatment systems) (Liu et al., 2019; Liu et al., 2022; Koutnik et al., 2022; O'Kelly et al., 2022; Viaroli et al., 2022). The vertical movement of MPs through the vadose zone is facilitated by wetting–drying cycles associated with groundwater fluctuations driven by precipitation and infiltration/recharge (O'Connor et al., 2019; Chia et al., 2021; Liu et al., 2022; Cramer et al., 2022; Feng et al., 2023). Thus, MPs found in New Jersey's surface waters (Ravit et al., 2017; Bailey et al., 2021) are potential sources for groundwater contamination, resulting from increased precipitation and recharge under climate change conditions.

In summary, the enhanced recharge stimulated by altered precipitation patterns and influenced by climate change is likely to cause changes in physical and geochemical conditions of aquifers, leading to mobilization and transport of geogenic and anthropogenic contamination of groundwater.

3.3 Extreme Events (Flooding and Drought) and Groundwater Quality

Climate change is expected to cause changes in precipitation patterns in New Jersey, namely increased frequency and intensity of extreme events like floods and droughts (NJDEP 2020, Nazarian et al., 2022; Jong et al., 2023). Increased flooding contributes to the groundwater recharge of shallow aquifers (Doble et al., 2012; Zhang et al., 2017), and can therefore also impact the quality of groundwater by transporting DOC, electron acceptors, and pollutants (see Section 3.2), particularly in unconfined shallow aquifers with high hydraulic conductivity. Unconfined shallow aquifers with high hydraulic conductivity provide insignificant attenuation capacity for contaminants coming from polluted surface water bodies (Miotliński et al., 2012; Gowrisankar et al., 2017; Ramachandran et al., 2019; Aladejana et al., 2020). In urban areas, flooding can increase the loading of organic carbon and common urban contaminants, such as oil, solvents, and sewage, to groundwater (Green et al. 2011; Khatri and Tyagi, 2015).

A common and immediate threat to public health from flooding is microbial/pathogenic contamination of groundwater by infiltration of contaminated flood water and/or flushing of accumulated pathogens in the vadose

zone (Hunter, 2003; Funari et al., 2012; Andrade et al., 2018; Rowles III et al., 2020; Pieper et al., 2021; Dzodzomenyo et al., 2022; Mapili et al., 2022; Gitter et al., 2023). For example, groundwater contamination through nutrient and pathogen loading was reported in eastern North Carolina after the flooding caused by a hurricane (Humphrey et al., 2021). Furthermore, flooding decreases redox potential (Eh) and promotes reducing geochemical conditions in surface soils, causing changes in metal solubility (Du Laing et al., 2009; Kelly et al., 2020; Ponting et al., 2021; Tang et al., 2022). Also, metal solubility could increase due to the formation of complex ions (complexation) during the dissolution of soil organic matter (SOM) caused by flooding (Kelly et al., 2020; Tang et al., 2022). Therefore, flooding can mobilize geogenic contaminants like As, Fe, Mn, P, and other metals (Cr, Mo, V) in surface soils, which eventually are transported to shallow aquifers via infiltration/recharge (Burton et al., 2008; Weber et al., 2010; Burton et al., 2014; Shaheen et al., 2014; Connolly et al., 2021; Müller et al., 2022; Fang et al., 2023; Yang et al., 2023a).

Decreased precipitation and longer dry periods, projected to occur in the warmer months, may lead to more frequent and extended droughts or drought-like conditions (Trenberth, 2011; Krakauer et al., 2019; NJDEP, 2020). Droughts have the potential to significantly impact groundwater quality (Levy et al., 2021; Petersen-Perlman et al., 2022). For example, recent studies have observed post-drought increases in NO_3^- concentrations (Jutglar et al., 2021) and certain redox-sensitive ions (SO_4^{2-}) and metals such as Fe and Mn (Aladejana et al., 2020). During drier conditions, decreased groundwater recharge causes groundwater levels to decline (NJDEP, 2002; Sen, 2015; Schreiner-McGraw, et al. 2021). Lowered groundwater exposes arsenic-iron sulfide bearing minerals to oxygen introduced by diffusion through the unsaturated zone and causes oxidation of these sulfide minerals (Welch et al., 2000; Hering and Kneebone, 2002; Foley and Ayuso, 2008; Garelick et al., 2009; Smedley and Kinniburgh, 2013; Houben et al., 2017; Asta et al., 2019; Erickson et al., 2021; Stolze et al., 2022; Zambito et al., 2022). These minerals are significant sources of high levels of As in the groundwater of New Jersey (Serfes et al., 2003 and 2010; NJGS, 2004). Therefore, mobilization of arsenic by sulfide oxidation is likely to be enhanced during drought periods and provides an explanation for observed and predicted increases in As concentrations associated with droughts (Verplanck et al. 2008; García-Prieto et al., 2012; Pili et al. 2013; Tisserand et al., 2014; Bondu et al., 2016; Ventura-Houle et al., 2018; Lombard et al., 2021).

In summary, flooding may mobilize geogenic contaminants like As, Fe, Mn, and other metals (Cr, Mo, V) in soils which eventually are transported to shallow aquifers via infiltration/recharge. Similarly, droughts also have the potential to increase contaminant concentrations of NO_3^- and As, among other compounds.

3.4 Sea Level Rise and Groundwater Quality

Increasing rates of sea-level rise (SLR) have been observed in New Jersey and globally over the last several decades (Nicholls and Cazenave, 2010; NJDEP, 2020). In coastal areas, shallow groundwater is expected to

respond dynamically to SLR due to hydraulic connections with nearby high salinity coastal environments (Befus et al., 2020). Some of the most likely impacts of SLR on coastal shallow aquifers are seawater intrusion (Bosserele et al., 2022), rising groundwater levels (Rozell, 2020; Bosserele et al., 2022), and coastal flooding (Vitouseket et al., 2017; Hummel et al 2018; Rahimi et al., 2020).

Seawater intrusion (SWI), upward and landward movement (from hundreds of meters to several kilometers) of the seawater-freshwater interface (Andersen et al., 2005; Werner and Simmons, 2009; Werner et al., 2013; Knott et al., 2018; Jasechko et al., 2020), has the potential to be one of the leading causes of groundwater contamination resulting from rising sea-level (Alshehri et al., 2021; Jeen et al., 2021; Abd-Elaty et al., 2022). An immediate effect of SWI in New Jersey, as well as other parts of the USA, is the increase in groundwater salinity, known as salinization (Barlow and Reichard, 2010; Aynaz and Shrinidhi, 2020; Jasechko et al., 2020; Lotfata and Ambinakudige, 2020). For example, SWI has been observed up to 3 kilometers inland, forcing the closure of public- and industrial-supply wells and private wells in Cape May County, New Jersey since the 1940s (Lacombe and Carleton, 2002; Barlow and Reichard, 2010). Salinization of coastal aquifers by SWI can also involve a range of complex geochemical processes affecting water quality by mobilization of geogenic and anthropogenic contaminants. (Moore and Joye, 2021). The increasing salinity driven by SWI can significantly affect the mobility of metals including radioactive metals due to cation exchange, desorption by Cl (metal chloro-complexation), and interaction with organic complexes (Appelo and Postma, 2005; Brady and Weil, 2008; Acosta et al., 2011; Sun et al., 2015; Lazur et al., 2020; Zheng et al., 2021).

Some studies observed increases in metals (As, Cd, Co, Cu, Hg, Pb, Se, V) and other elements, such as B, Ba, Ca, Mg, Si, Sr, and radioactive metal radium (Ra) with increased groundwater salinity (Bolton, 2000; Du Laing et al., 2008 and 2009; Kiro et al., 2012; Vinson et al. 2013; Zhao et al., 2013; Sun et al., 2015; McNaboe et al., 2017; Liu et al., 2019a; Wen et al., 2019; Lazur et al., 2020; Mora et al., 2020; Papazotos et al., 2020; Moore and Joye, 2021). However, the effect of salinity on metals varies with chemical species, predominant geochemical phase, and the source of metals (Rumuri et al., 2021). An increase in Ra concentrations has been observed in drinking water wells installed in the Kirkwood–Cohansey aquifer in southern New Jersey due to increased salinity associated with road salt application (Lindsey et al., 2021). Ra concentration in groundwater increases with salinity because of competition for adsorption sites by Na⁺ ions, and the formation of RaCl⁺ and RaCl₂ complexes that enhance the solubility of Ra in saline waters (Langmuir and Riese 1985; Tamamura et al. 2014; McNaboe et al., 2017; Lindsey et al., 2021).

Other geochemical processes, such as redox reactions, sorption/desorption, co-precipitation/dissolution, and change in ionic strength, can also occur during SWI that affect the concentrations of major and trace elements in groundwater (Khadra et al., 2017; Mora et al., 2020; Lazur et al., 2020; Moore and Joye, 2021). Some studies showed that As desorption in aquifer sediments was higher due to increased groundwater salinity and alkalinity

(Morelli et al., 2017; Dehbandi et al., 2019; Yuan et al., 2023). Additionally, SWI inhibits the recrystallization of iron (hydr)oxides and facilitates the formation of iron sulfide leading to decreased adsorption potential of As (Yuan et al., 2023). In recent studies, the mobilization of some cations and As, Cu, Hg, Ni, Rb, and U in aquifers was driven by the desorption or dissolution of aquifer minerals during SWI (Appelo et al., 1990; Moore, 1999; Grassi and Netti, 2000; McFarland, 2010; Habtemichael et al., 2016; Khadra et al., 2017). Seawater, because of its high sulfate concentrations (Canfield and Farquhar, 2009) has stronger oxidizing capacity than freshwater, which causes oxidation of organic matter (Moore and Joye, 2021). Organic matter oxidation plays an important role in stimulating the reduction of iron and manganese oxides and the release of As, Fe, and Mn at seawater-freshwater boundaries (Snyder et al., 2004; Moore and Joye, 2021). Moreover, an increase in groundwater pH due to seawater mixing (Bozlee et al., 2008) would decrease the groundwater Eh (oxidation potential) promoting reducing conditions because of the inverse relationship between pH and Eh (Xyla et al., 1992).

Modeling studies have shown that SLR would cause the denser seawater (because of its salt content) to intrude under the fresh groundwater and force water levels to rise in coastal shallow aquifers (Masterson and Garabedian, 2007; Oude Essink et al., 2010; Bjerkli et al., 2012; Rotzoll and Fletcher, 2013; Masterson et al., 2014; Masterson et al., 2014; Walter et al., 2016; Hoover et al., 2017; Knot et al., 2019; Habel et al., 2020; Rozell, 2020). Rising groundwater levels over the past 50 years in coastal areas of Massachusetts (Douglas and Kirshen, 2022) and New England (Dudley and Hodgkins, 2013; Weider, 2014) are consistent with groundwater rise in response to the rising sea level. Factors that control groundwater rise in coastal areas are (a) groundwater recharge (Bjerkli et al., 2012), (b) groundwater discharge to tidal water bodies (SRPC, 2022), (c) the rate of SLR (Masterson & Garabedian, 2007), (d) the hydraulic connection of shallow aquifers to sea (Rotzoll and Fletcher, 2013; Befus et al., 2020), (e) the topographic/hydrographic settings (Knott et al., 2019), and (f) infrastructure systems (Habel et al., 2020). A groundwater numerical simulation of New Hampshire's coastal region has projected that mean groundwater rise (percentage of height) relative to SLR under high-emission scenario is 66% between 0 and 1 km, 34% between 1 and 2 km, 18% between 2 and 3 km, 7% between 3 and 4 km, and 3% between 4 and 5 km of distance from the coastline (Knott et al., 2019). Similarly, groundwater flow simulation indicated that the water tables in the unconfined shallow aquifer systems underlying the coastal areas of New Jersey (Fiore et al., 2018; Carleton et al., 2021) and New York (Misut and Dressler, 2021), are likely to rise in response to SLR. The rising groundwater level poses threats to groundwater by mobilizing natural elements and anthropogenic contaminants in vadose zones, as described in Section 3.2.

Many coastal communities, including in New Jersey, rely on on-site septic systems for treating wastewater. In general, a properly functioning septic system requires at least two feet of soil to provide filtration by removing solids and bacteria/pathogens from wastewater (USEPA, 2002). Therefore, the rising water table may lead to

septic failures and groundwater contamination through nutrient and pathogen loading (Mihaly, 2018; Cox et al., 2020).

Increased surface-water flooding due to storm surges and tides would be a common consequence of SLR (Nicholls, 2002; Strauss et al., 2012; Woodruff et al., 2013; Neumann et al., 2015; Moftakhari et al., 2017; Kulp and Strauss, 2019; Habel et al., 2020; Griggs and Reguero, 2021; Johnston et al., 2021; Thompson et al., 2021; Alarcon et al., 2022; Bilskie et al., 2022; Lopes et al., 2022; Reguero and Griggs, 2022; Wing et al., 2022) and the rising groundwater is likely to exacerbate coastal flooding hazards (Rahimi et al., 2020; Bosserelle et al., 2022; Su et al., 2022). Therefore, shallow aquifers in coastal areas are vulnerable to seawater infiltration due to coastal floodings and tidal inundations. The potential adverse effects of flooding on groundwater quality of shallow aquifers, which include transportation/infiltrations of DOC, electron acceptors, pollutants, and pathogens, are discussed in section 3.3. Also, seawater intrusion induced by coastal flooding can significantly affect the mobility of metals in soils/sediments due to cation exchange and the presence of Cl and organic complexes, as discussed above in this section. Many coastal sediments, including those in the United States, are significant sources of both anthropogenic and natural contaminants (Santschi et al., 2001; Barringer and Szabo, 2006; Vane et al., 2008; Barringer et al., 2013; Qian et al., 2015; Maruya et al., 2016; Sakizadeh, 2020; Paul et al., 2021). Therefore, coastal floodings in response to SLR can lead to the mobilization of geogenic contaminants like As and other metals in soils, which eventually are transported to shallow aquifers (LeMonte et al., 2017; Izaditame et al., 2022).

In summary, the increasing salinity driven by SWI can mobilize metals (As, Cd, Co, Cu, Hg, Pb, Se, V, and Ra) from soils/sediments to groundwater due to cation exchange and the presence of Cl and organic complexes. Similarly, rising groundwater and coastal flooding are likely to affect contaminant concentrations in shallow coastal aquifers.

3.5 Climate Change and Contaminant Transport and Remediation

Climate change could impact large numbers of contaminated sites as climate change and the associated extreme weather events can undermine the effectiveness of site remediation efforts (US EPA, 2013; Jesus et al., 2015; Libera et al., 2019; O'Connor et al., 2019a). This is due to shifts in hydrogeological and geochemical conditions of aquifers which can affect the contaminant's fate and transport, as well as long-term operations and management of remediation sites.

Substantial groundwater level fluctuations are likely to occur over the next century in response to variations in rainfall intensity and frequency expected under climate change conditions (Nygren et al., 2020; IPCC, 2022, Sections 3.2, 3.3 and 3.4). Furthermore, groundwater recharge is predicted to be enhanced under future climate

scenarios due to projected increases in precipitation (Section 3.2). These modified hydrogeological conditions may strongly influence contaminant mobilization, transport and fate, and the risk for receptors (Cavelan et al., 2022; Xu et al., 2022). For example, higher recharge would mobilize contaminants in the vadose zone and move them down to the aquifer (Zuo et al., 2021; Cavelan et al., 2022). Furthermore, increased recharge would augment hydraulic gradients by raising the groundwater level, resulting in enhanced plume mobility and mixing/dilution of contaminants in groundwater due to faster groundwater flow (Marshall et al., 2000). Given these conditions, increased recharge rates caused by climate change effects may enhance plume mobility and increase pollution load in downgradient locations (Gupta and Yadav 2017). Inversely, a decrease in recharge would result in decreased plume mobility and reduce the mixing and dilution of contaminants (Libera et al., 2019; Xu et al., 2022). Also, recharge and seasonal water level fluctuations in aquifers control the redox process and contribute to the natural attenuation of contaminants in aquifers (Böhlke, J.K., 2002a; McGuire et al., 2005; Scholl et al., 2006; Yadav et al., 2012; Rezanezhad et al., 2014; Van De Ven et al., 2021; Cavelan et al., 2022). Recharge delivers electron acceptors, such as O_2^{2-} , NO_3^- , SO_4^{2-} , and Fe^{3+} , that cause significant changes to anaerobic, contaminated aquifers and are important for natural attenuation of contaminants (McGuire et al. 2005; McMahon et al., 2011). The upward/downward movements of a contaminant plume, caused by water level fluctuations, can lead to the entrapment of pollutants in the form of isolated blobs (residual masses of contaminants) in pore spaces (Libera et al 2019). These residual pollutants trapped in the porous soils become lasting sources of pollution (Yadav and Hassanizadeh 2011; Libera et al 2019). Therefore, it is crucial to understand the response of contaminated unconfined aquifers to recharge events for the design and evaluation of effective bioremediation and natural attenuation strategies. As discussed earlier, more extreme droughts and heavy precipitation events (and associated floods) are likely to occur more frequently due to climate change and will likely influence soil moisture content. Several studies have shown that microbial respiratory activity that controls the degradation of contaminants in the soil also depends strongly on moisture (Xiang et al., 2008; Butcher et al., 2020; Vu et al., 2022).

An increase of a few degrees Celsius in the mean surface and groundwater temperature is expected (IPCC, 2022). Higher temperatures associated with climate change are likely to stimulate microbial activity, thus potentially enhancing biodegradation of organic and other pollutants (Anthony, 2006; Ali, 2010; Baczynski et al., 2010; Zeman et al., 2014; Bajaj and Singh, 2015; Bargiela et al., 2015; Liao et al., 2016; Alkorta et al., 2017; Westphal et al., 2017; Gaur et al., 2018; McAlexander and Sihota, 2019; Riedel et al., 2019; Retter et al., 2021; Cavelan et al., 2022). Thus, this phenomenon may affect temperature-dependent mobilization and biodegradation processes which would greatly shift the effectiveness of some remediation systems, including bioremediation and monitored natural attenuation (MNA). In general, solubility increases with temperature for most organic compounds and metals (Delle Site, 2001; Wang et al., 2010; Ossai et al., 2020; Cavelan et al., 2022). Furthermore, organic matter enhances the solubility and mobility of metals and other pollutants through complexation and binding (Murphy and Zachara, 1995; Weng et al., 2002; Wang et al., 2007; Aiken et al., 2011;

Daugherty et al., 2017; Lipczynska-Kochany, 2018; Adusei-Gyamfi et al., 2019; Tang et al., 2022a). Therefore, climate change is likely to play a significant role in the fate and transport of pollutants by changing the structure and quantity of organic matter and modifying interactions between organic matter, metals, and sediments in aquifer (Lipczynska-Kochany, 2018; Cavelan et al., 2022).

Sea-level rise is also expected to impact many contaminated sites. This is an important consideration for New Jersey as the state has the second highest number of Superfund sites in the United States (Gómez, 2021; Kiaghadi et al., 2021). Many of these sites are in low-lying coastal regions and have the potential to release hazardous contaminants that could pose risks to public health into aquifers. For example, the rising groundwater due to heavy precipitation and flooding in response to SLR can remobilize pollutants dormant in the soil at contaminated sites. Moreover, SLR is likely to reduce the hydraulic gradient in inland and coastal areas which would affect the transport and dispersion of contaminants (O'Connor et al., 2019a; Guo et al., 2020).

To summarize, the effects of climate change create both risks and opportunities, which should be considered during remedial design and long-term planning.

4.0 Conclusions and Knowledge Gaps

This review examines the present and future impacts of climate change on groundwater quality based on our current knowledge. The review of available scientific literature suggests the effects of climate change on groundwater quality may be a slow and long-term process that will affect groundwater quality by modifying hydrogeological processes such as precipitation, groundwater recharge, discharge, aquifer storage/water level, and seawater intrusion. These modifications influence biogeochemical reactions, chemical fate, and the transport of contaminants. Changes in hydrogeological and biogeochemical processes induced by climate change are likely to cause long-term and seasonal variability both in anthropogenic and geogenic contaminants (Levitt et al., 2019; Degnan et al., 2020; Henri et al., 2020; Lasagna et al., 2020; Lee and Murphy, 2020; Guo et al., 2021; Mailloux et al., 2021). Thus, it is crucial to understand the time scales of change in contaminant concentrations in drinking water wells for contaminant regulation, mitigation, and, eventually, the reduction of human exposure to harmful contaminants (Degnan et al., 2020). For example, some studies showed that geogenic contaminants, such as As and other anthropogenic contaminants, have variable seasonal and long-term concentrations, related to water level fluctuations and climate variability (Goovaerts et al., 2005; Ayotte et al., 2015; Levitt et al., 2019; Jarsjö et al., 2020; Lasagna et al., 2020; Degnan et al., 2020; Spaur et al., 2021). Furthermore, the increasing salinity driven by SWI is likely to mobilize As in coastal areas (Section 3.4), which can lead to considerable and systematic underestimation of As mobilization and future changes. Therefore, this warrants the collection of multiple samples throughout the year to avoid failure to document potential exceedances of a human health standard when

wells may be sampled under low concentration conditions (Degnan et al., 2020; Mailloux et al., 2021). This more frequent sampling approach may be particularly important for private wells because of their lesser testing frequency than public-supply wells (Degnan et al., 2020; O'Neill et al., 2022; Seliga et al., 2022).

As the review indicates, climate change will affect groundwater quality in several different ways, however, quantification of the impacts is difficult because of uncertainty in climate projections, global socio-economic developments, and the complex responses of the hydrogeological system to climate variability (Clark et al., 2016; Refsgaard et al., 2016; Reinecke et al., 2021; Mensah et al., 2022; Huggins et al., 2023). Long-term monitoring studies which include a denser network of groundwater observation points are necessary to improve the quantification of hydrogeological systems that are likely to be affected by climate change at local to regional scales.

Groundwater is likely to play a critical role in meeting the increased water demand in the future because of substantial declines in surface water availability due to climate change and other natural and anthropogenic impairing factors (MacDonald, 2010; Taylor et al., 2013; Wada et al., 2013; Elliott et al., 2014; Haddeland et al., 2014; Flörke et al., 2018; Bierkens and Wada, 2019; Brown et al., 2019; Allan et al., 2020; Sanchez et al., 2020; Akhtar et al., 2021; Abbas et al., 2022; Swain et al., 2022; Warziniack et al., 2022; Qiu et al., 2023). Therefore, the likely increased groundwater extraction needed to meet the future water demand could aggravate declining water tables, causing a loss of groundwater storage, and decreasing water quality in many already stressed aquifer systems. Impacts associated with climate change, are expected to increase over time and are already in motion because of historic releases of CO₂ and the inertia of the climate system (Meehl et al., 2005; Mertz et al., 2009; Brown et al., 2019), thus mitigation and adaptation strategies will need to be identified in order to adequately address the consequential alterations of groundwater quantity and quality (Berrang-Ford et al., 2011; Brown et al., 2019; Hanifehlou et al., 2022; Stigter et al., 2022; Maheshwari, 2023; Scanlon et al., 2023).

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