Climate Change Impacts on Groundwater in MAPC Communities

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University of Massachusetts Boston
School for the Environment

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ACKNOWLEDGMENTS

When the inaugural Boston Research Advisory Group (BRAG) report was released in 2016, it was recommended that the scientific consensus on climate change risk factors for Boston be updated every three to five years. The Barr Foundation made this update possible. Darci Schofield of the Metropolitan Area Planning Council (MAPC) had noted that the BRAG report offered essential information that was useful to and utilized by many cities and towns outside of the City of Boston and recommended a compilation of more localized information in the update. Subsequent discussions with Bud Ris, Mary Skelton Roberts, Emily Sidla, and Kalila Barnett of the Barr Foundation led to the expansion of the study area to include the 101 towns and cities within the MAPC region. We also acknowledge Ms. Schofield for helping to recruit members of the GBRAG steering committee and to organize our outreach activities within the MAPC domain. This special report is a result of those outreach activities as the impacts of climate change on groundwater resources were mentioned as a major concern across the region.

We acknowledge and appreciate the guidance and support we received from John Cleveland and Amy Longsworth of the Green Ribbon Commission in launching the GBRAG. We gratefully acknowledge the Barr Foundation for funding the GBRAG activities and reporting. We also gratefully acknowledge the competent and steadfast administrative efforts of Kimberly Starbuck of the Urban Harbors Institute at UMass Boston, who organized and managed GBRAG meetings, communications, and the GBRAG report. We further acknowledge the members of the GBRAG Steering Committee for their time and thoughtful feedback during this process. We gratefully acknowledge the thorough review of this report and detailed feedback provided by David Bjerklie, New England Water Science Center, United States Geological Survey.

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Executive Summary

The inaugural Boston Research Advisory Group (BRAG) report for Climate Ready Boston (Douglas et al., 2016) was an initiative to help develop resilient solutions to prepare Boston for climate change. The BRAG report covered sea level rise, coastal storms, extreme precipitation, and extreme temperatures. Groundwater was not considered in that report because the effects of climate change on groundwater are complex and broad. As part of the recent Greater Boston Research Advisory Group's (GBRAG's) initiative, a special report devoted solely to climate change effects on groundwater in the Metropolitan Area Planning Council (MAPC) region has been prepared. Groundwater is often overlooked in climate change discussions because it is out of sight and slow moving. Groundwater changes are not as obvious as other risk factors, such as more intense precipitation or rising temperatures, for example, but are equally important and should not be overlooked. This report is broken into four chapters that describe some of the climate parameters that influence groundwater levels, how these parameters might change, and the potential effects of these changes.

Chapter 1
Introduction

Groundwater is the world's largest distributed source of fresh water and is important for both ecosystems and human consumption (Taylor et al., 2013). Groundwater levels are controlled by groundwater recharge, the amount of water from precipitation or surface-water bodies infiltrating the ground surface and percolating through the soil to the groundwater table, and groundwater losses. These include groundwater discharge to surface-water bodies either naturally or through underground infrastructure and groundwater withdrawals. The amount of groundwater recharge is affected by precipitation, temperature and evapotranspiration, land cover and land use, soil moisture, and topography (Boutt et al., 2019). Precipitation intensity, temperature, and sea levels are all projected to increase in this region with climate change. These changes can result in seasonal and long-term changes in groundwater levels with implications for drinking water, buildings and infrastructure, ecosystems, and water quality.

Chapter 2
Regional Hydrogeology

The study area is the MAPC region consisting of 101 communities in eastern Massachusetts (MA). The aquifers (geologic materials with sufficient water for wells) in this region are glacial deposits consisting of layered and sorted sand, gravel, silt, and clay (stratified drift) that lie along rivers and streams in north-south trending bedrock valleys (Boutt et al., 2010; Desimone, 2004; D.A. Masterson et al., 2009). Glacial till and bedrock make up most of the remaining area. Wells typically draw water from the stratified drift deposits or from fractured bedrock.

The climate of MA is humid and temperate with a wide range of temperatures and relatively uniform precipitation throughout the year (Bjerklie & Sturtevant, 2018). The average precipitation in the region is approximately 109 cm/year (43 in/year) (National Climatic Data Center, n.d.). Evapotranspiration is the sum of evaporation losses from surface-water bodies, soils surfaces, and transpiration from plants (Freeze & Cherry, 1979) and varies spatially depending on the percentage of vegetated cover versus impervious surface. It ranges from 48 to 62 cm/year (19 to 24 in/year) in the suburban areas outside of the City of

Boston (U.S. Geological Survey, 2021). Aquifer recharge is affected by evapotranspiration, dropping to zero in the summer months during the growing season when evapotranspiration is the highest.

The Massachusetts Water Resources Authority (MWRA) provides water service to approximately 30% of the study area leaving approximately 70% of the area relying on local surface water and groundwater sources for their water supply. Groundwater withdrawals are the largest local source; they average over 50% of water withdrawals in Middlesex, Norfolk, and Plymouth Counties. Groundwater withdrawals account for less than 10% of the water withdrawals in Essex County due to the relative scarcity of sand and gravel aquifers in this region (U.S. Geological Survey, 2021).

Chapter 3
Climate Change and Groundwater

The key findings are:

- Groundwater is important for water supply, stream baseflows, wetlands, and stormwater management.
- Groundwater levels in the MAPC area have shown increasing elevation trends in long-term monitoring wells. This is due to precipitation increases in the northeast currently out pacing increases in evapotranspiration.
- Climate change will affect groundwater levels through changes in aquifer recharge associated with changing precipitation, temperature, and snowmelt.
- Prior to 2070, seasonal groundwater recharge and stream baseflows are projected to increase in the late fall and early winter with increases in precipitation. Annual groundwater recharge is projected to decrease after year 2030 due to reduced snowpack and evapotranspiration increases in vegetated areas both from rising temperatures and longer growing seasons.
- Annual groundwater recharge could decrease by 3% to 28% by the end of the century (year 2100) depending upon the greenhouse gas emission scenario and location.
- Groundwater levels will rise near the coast due to Relative Sea Level Rise (RSLR). Rising groundwater will occur farther inland than tidal inundation.

Chapter 4
Potential Impacts of Changing Groundwater Levels

The key findings are:

- Assets and natural resources in areas where groundwater is currently shallow may be vulnerable to weakened soil conditions or groundwater inundation (flooding) with future increases in groundwater levels.
- Potential impacts from declining aquifer recharge in the spring and declining groundwater levels in the summer include reductions in annual water supply and summer stream baseflows.
- Potential impacts from rising groundwater at the coast include premature infrastructure failure, water quality degradation, wetland expansion and/or transition, and flooding.

Chapter 5
Summary, Open Questions, and Data Gaps

The importance of groundwater for drinking water, natural resources, and streamflow in the northeast is well documented. Groundwater levels are also important considerations in maintaining proper functioning of pavements, underground infrastructure, building foundations, and waste disposal areas as well as other grey and green infrastructure. Climate change induced groundwater changes in the northeast are complex and vary both temporally and spatially. Increases in precipitation and sea levels may result in higher groundwater tables and more flooding, while lengthening growing seasons with more evapotranspiration and less snowmelt may result in declining aquifer recharge and lower groundwater tables. Spatially, coastal communities have different challenges than inland communities, and rural communities have different challenges than urban areas. Groundwater flow and transport models coupled with groundwater monitoring are powerful tools for simulating complex hydrologic interactions with a changing climate and changing land use. They can integrate changing aquifer recharge, evapotranspiration, water withdrawals, groundwater/surface water interactions, sea level, and contaminant transport (Habel et al., 2020; Knott, Jacobs, et al., 2019). Regional groundwater models and long-term monitoring wells are needed in the MAPC region to assess and predict the combined effects of precipitation, temperature, sea level, and land use changes on groundwater levels. In addition, studies combining climate model output with rainfall-runoff models calibrated with field studies are needed to assess the effect of more extreme rainfall events on groundwater recharge.

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1. Introduction

Groundwater is the world's largest distributed source of fresh water and is important for both ecosystems and human consumption (Taylor et al., 2013). Groundwater levels are affected directly by recharge (water infiltrating the ground surface and moving into the groundwater system) and water losses through groundwater discharge to surface-water bodies and groundwater withdrawals from aquifers. Many factors influence the amount of groundwater recharge that occurs. These include precipitation, temperature, evapotranspiration, land cover and land use, soil moisture, and topography (Boutt et al., 2019). Climate change is affecting the global water cycle by increasing rates of ocean evaporation, terrestrial evapotranspiration, and precipitation (Huntington et al., 2018; Kramer et al., 2015). Precipitation, temperatures, and sea levels are all projected to increase in the northeast due to climate change. These factors can result in long-term and seasonal changes in groundwater levels potentially impacting drinking water supplies, water quality, the useful life of pavements and underground infrastructure, and flooding.

2. Regional Hydrogeology

2.1 STUDY AREA

The 101 communities served by the Metropolitan Area Planning Council (MAPC) occupy 3732 km^2 (1441 mi^2) in the Greater Boston area. The MAPC planning area is divided into eight major subregions shown in Figure 2.1 including (from the north to the south) the North Shore Task Force (NSTF), the North Suburban Planning Council (NSPC), the Minuteman Advisory Group on Interlocal Coordination (MAGIC), the Inner Core Communities (ICC), the MetroWest Regional Collaborative (MetroWest), South West Advisory Planning committee (SWAP), the Three Rivers Interlocal Council (TRIC), and the South Shore Coalition (SSC) (Metropolitan Area Planning Council, 2021).

The topography of the area ranges from 192 m (630 ft) in the northwestern part of the study area to mean sea level (–0.092 m or –0.302 ft NAVD88) at the coast (National Oceanic and Atmospheric Administration, 2021; MassGIS 2020a).

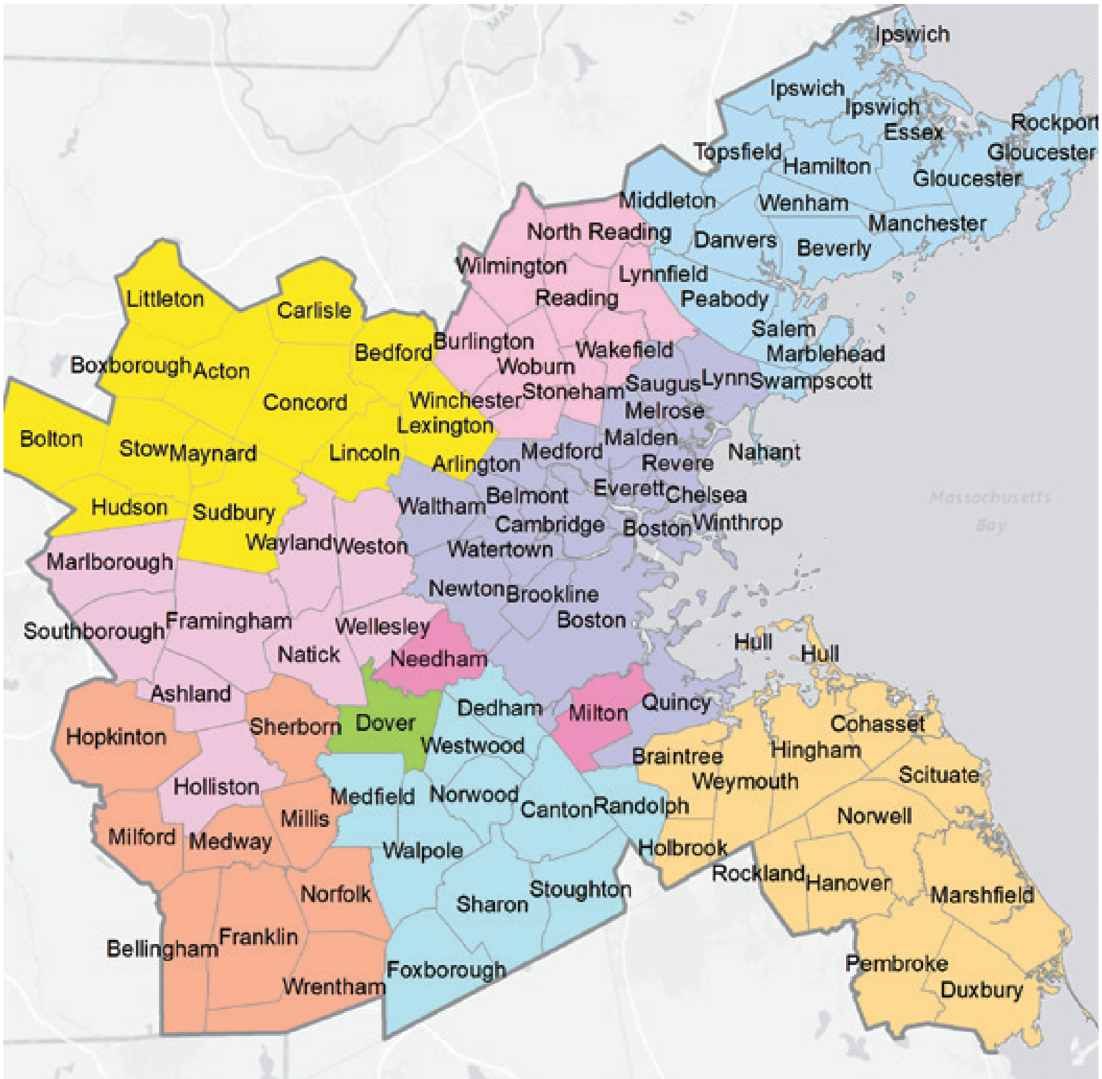
2.2 GEOLOGY

The surficial geology in the study area, shown in Figure 2.2, consists of unconsolidated glacial sediments deposited in north-south trending valleys in fractured metamorphic and crystalline bedrock (Boutt et al., 2010; Desimone, 2004; Kirshen et al., 2014; D. A. Masterson et al., 2009). The major water bearing deposits that form aquifers (geologic materials with sufficient water to supply water to wells) are stratified drift glacial deposits composed of layered and sorted sand, gravel, silt, and clay that lie along the rivers and streams. These deposits typically range from approximately 11 to 49 m (36 to 161 ft) thick in the western part of the study area (e.g., the Assabet River basin) with coarser-grained sediments overlying thick layers of fine sand, silt, and clay (Brackley & Hansen, 1985; Desimone, 2004; Randall, 2001). The hydraulic conductivity of these deposits ranges from 24 to 206 m/day (79 to 676 ft/day) averaging approximately 58 m/day (190 ft/day) (Desimone, 2004). Fractured bedrock also contains groundwater but the quantities available for use are typically limited and are less connected to surface water bodies.

Transmissivity, defined as the hydraulic conductivity multiplied by the aquifer thickness, can be used to characterize an aquifer's potential for water supply. In eastern Massachusetts, aquifers with transmissivities less than 130 m^2/day (1400 ft^2/day) are associated with low-yield aquifers (<50 gpm); transmissivities ranging from 130 m^2/day to 372 m^2/day (1400 to 4000 ft^2/day) are associated with medium-yield aquifers (100 to 300 gpm); and transmissivities greater than 372 m^2/day (4000 ft^2/day) are associated with high-yield aquifers (>300 gpm) (MassGIS, 2020b). The transmissivity of the Assabet River basin stratified drift aquifers ranges from 260 m^2/day (2,799 ft^2/day) to over 2,200 m^2/day (23,681 ft^2/day). In the southeastern part of the study area, stratified drift deposits in valleys underlain by bedrock have transmissivities ranging from 120 m^2/day (1,292 ft^2/day) to more than 370 m^2/day (3,983 ft^2/day) (B. P. Hansen & Lantham, 1992; D. A. Masterson et al., 2009; Persky, 1993). Low permeability glacial till is also prevalent in the study area and can occur either as a thin 3 to 5 m (10 to 16 ft thick) layer covering uplands (Boutt et al., 2019) or as drumlins which are 15 to 24 m (49 to 79 ft) thick elongated hills typically oriented in a north

Figure 2.1

The study area consisting of the Metropolican Area Planning Council (MAPC) area with MAPC Subregions and town names.



MAPC Subregions

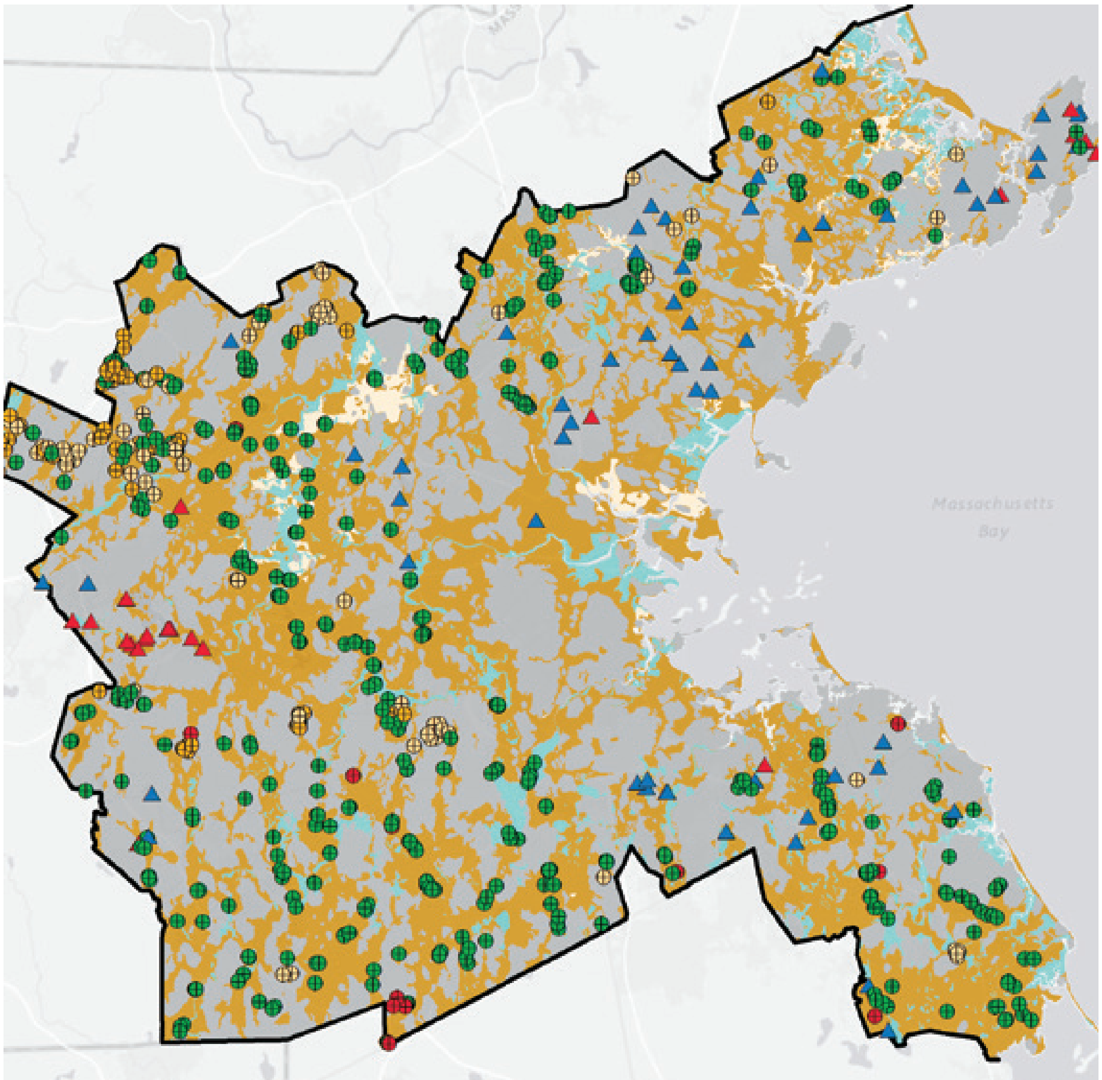
- INNER CORE
- INNER CORE/TRIC
- MAGIC
- METROWEST
- NSPC
- NSTF
- SSC
- SWAP
- SWAP/TRIC
- TRIC

0 10 20 km

Source: Esri, HERE, Garmin. © OpenStreetMap contributors and the GIS user community.

Figure 2.2

Generalized surficial geology with public water supply sources in the study area. NTNC is Non-Transient, Non-Community (e.g., schools, hospitals, factories, etc.) water supply; TNC is Transient Non-Community (e.g., gas stations, campgrounds, etc.) water supply.



Surficial Geology & Public Water Supply

- Sand & Gravel
- Till or Bedrock
- Fine Grained Deposits
- Floodplain Alluviam
- Surface Water
- Emergency Surface Water
- Groundwater Withdrawal Location
- Proposed Well
- NTNC
- TNC

0 10 20 km

Source: Esri, HERE, Garmin. © OpenStreetMap contributors and the GIS user community.

south direction (Desimone, 2004; W. R. Hansen, 1956; Koteff, 1966). Due to low transmissivity, till deposits are typically not used for public water supply (Boutt, 2017).

Underlying the unconsolidated surficial deposits is crystalline bedrock. The Nashoba and Avalon Terranes, oriented from southwest to northeast, dominate the bedrock geology in the study area (Arvin, 2010). The Nashoba Terrane is in the western part of the study area and consists of metamorphic rocks to the east and metasedimentary rocks to the west. The Avalon Terrane is in the eastern and southeastern part of the study area and consist primarily of granites with some gneisses and quartzites. The Bloody Bluff Fault Zone separates the Nashoba and Avalon Terranes. Fractured bedrock is becoming increasingly important for water supply in this area. (Arvin, 2010; Boutt et al., 2010).

2.3 HYDROLOGY AND HYDROGEOLOGY

The climate of Massachusetts (MA) is humid and temperate with a wide range in diurnal and seasonal temperatures and relatively uniform precipitation throughout the year (Bjerklie & Sturtevant, 2018; DeSimone et al., 2002). According to precipitation and temperature records at Logan Airport in Boston, the annual average precipitation was 109 cm/year (43 in/year) with a standard deviation (S.D., a measure of the spread of values around the mean) of 18 cm/year (7 in/year) from 1970 to 2019. The monthly mean temperature over the same period ranges from 1°C (29°F) in January with a S.D. of 2.3°C (4.2°F) to 23°C (74°F) in July with a S.D. of 1.2°C (2.1°F) (National Climatic Data Center, 2021). Summer air temperatures have increased 1.1°C (2.0°F) in New England from 1950 through 2006; a 0.19°C increase per decade (Hodgkins & Dudley, 2011); however, winter air temperatures have warmed approximately 0.7°C per decade, more than three times faster (Hayhoe et al., 2007).

Evapotranspiration is the sum of evaporation losses from surface-water bodies, soil surfaces, and transpiration from plants. Potential evapotranspiration (PET) is the amount of evapotranspiration that can occur if it is not limited by the availability of water. Actual evapotranspiration (AET) is a portion of PET that occurs under the constraints of the soil moisture supply. AET approaches PET in areas where groundwater is discharging to surface-water bodies such as streams, ponds, and lakes; wetlands, where soils remain saturated due to high groundwater levels; and open water surfaces including the ocean. AET is usually substantially less than PET in recharge areas—areas where water infiltrates the ground surface and moves into the groundwater system (Freeze & Cherry, 1979). The average AET in the study area for the period of 2000 through 2013 varies spatially. AET is the lowest (0 to 48 cm/year, 0 to 19 in/year) in the areas of low vegetation cover and where the impervious surface percentage is high as in the ICC subregion and other urban areas within the MAPC subregions. In the suburban areas outside of the city, AET ranges from 48 to 62 cm/year (19 to 24 in/year). The highest amount of AET occurs in the wetlands and rural areas where vegetation is prevalent and there is ample water supply. AET in these area ranges from 62 to 78 cm/year (24 to 31 in/year) (U.S. Geological Survey, 2021). Aquifer recharge and streamflows are affected by AET, decreasing when evapotranspiration increases in the spring and summer, with the lowest stream flow usually occurring in August and September. AET decreases in the fall and the winter months (Hodgkins & Dudley, 2011).

The region has eight major river watersheds either all, or mostly, within the study area (Figure 2.3). The largest watershed is the Charles River watershed followed by the Concord River watershed with drainage areas of 823 km^2 (318 mi^2) (22%) and 726 km^2 (280 mi^2) (19%) of the MAPC area, respectively. In addition to approximately 730 km (454 mi) of streams, 395 km^2 (153 mi^2) (11%) of the MAPC area is covered by ponds and other surface water bodies. These wet areas are described in Table 2.1.

The largest area of flooded land, approximately 207 km^2 (80 mi^2) (or 52% of the wet area) is classified as freshwater wetlands, areas where the fresh groundwater table is at or near the ground surface. Another 50 km^2 (19 mi^2) or 13% is classified as saltwater wetlands. Areas with high percentages of wetlands and

Figure 2.3

Major watersheds in the study area with major streams and lakes.

Source: Esri, HERE, Garmin. © OpenStreetMap contributors and the GIS user community.

Table 2.1

Hydrography of the MAPC area; 1 km² = 0.386 mi². Percent is the percent of wet area. Inundated area is flooded land that is not otherwise classified. (MassGIS, 2020c)

Description	Area (km²)	Percent
Wetland	206.9	52%
Pond, Lake	83.2	21%
Salt Wetlands	50.3	13%
Reservoir	33.8	9%
Tidal Flats, Shoals	10.6	3%
Cranberry Bog	3.8	1%
Submerged Wetland	3.7	1%
Bay, Ocean	2.7	1%
Inundated Area	0.1	0%
Total	395.1	100%
Total MAPC Area	3,732.0	
Water Bodies in MAPC Area	395.1	11%

open water have been associated with higher amounts of evapotranspiration which may increase with increasing temperatures (Hodgkins & Dudley, 2011).

Groundwater typically discharges to streams in the study area and is an important component of stream baseflows. In the upper Charles River drainage basin in the western part of the study area, 95% of the streams are gaining streams, i.e., streams that receive water from groundwater, and only 5% of streams are losing streams, i.e., streams that lose water to underlying aquifers (DeSimone et al., 2002; Eggleston, 2003). Stream baseflows are important for maintaining river flow and aquatic habitat during periods of low precipitation as well as for maintaining water quality in rivers where wastewater effluent is discharged (Price, 2011).

Aquifer recharge is the water that infiltrates the ground surface and travels through the unsaturated zone to the saturated zone that comprises the aquifer. The amount of aquifer recharge that an area receives depends on the climate, soils, topography, land use, and proximity to discharge areas. A substantial portion of the water that infiltrates the ground surface is lost through evapotranspiration from plants. This lost water is not considered recharge since it does not reach the water table (saturated zone). The amount of evapotranspiration depends on the vegetative cover, the local air temperature and relative humidity, and the amount of available water. Consequently, aquifer recharge is seasonally dependent. Most recharge occurs during the late fall, winter, and early spring when plants are dormant and evaporation rates are small (Heath, 1983; D. A. Masterson et al., 2009). The annual average groundwater recharge rate in the western part of the study area (the Assabet River valley area) ranged from 43 to 53 cm/year (17 to 21 in/year), or 37 to 44% of annual average precipitation for the period of 1964 through 2002. The average monthly recharge rate during this same period ranged from a maximum of just over 11 cm (4 in) in March to a minimum of less than 1 cm (0.4 in) in June, July, and August (Desimone, 2004). In the southeastern part of the study area the recharge rate in the stratified drift deposits is estimated to be 68 cm/year (27 in/year) or 57% of the total precipitation (D. A. Masterson et al., 2009).

The amount of impervious surface and vegetated cover as well as the topography all affect evapotranspiration, runoff, and ultimately aquifer recharge. Areas with high impervious area percentage will have lower recharge resulting in lower baseflows (low flows) to streams and higher-peak stream flows due to the increased runoff potential (Bjerklie & Sturtevant, 2018). Areas with more vegetated cover will also experience lower amounts of aquifer recharge during the growing season. Groundwater recharge is also influenced by topography and soil type with the greatest amount of recharge occurring in relatively flat coastal plains and alluvial deposits (Schilling, 2009).

In the absence of recharge projections for MA, research in neighboring New Hampshire (NH) was used to investigate recharge trends in the MAPC area. NH was modeled with the USGS Precipitation-Runoff Modeling System (PRMS) to assess the impacts of climate change on recharge, streamflow, and snowfall (Bjerklie & Sturtevant, 2018). This analysis was done at a Hydrologic Response Unit (HRU) spatial scale. This is defined by Bjerklie & Sturtevant (2018) as the smallest hydrologically homogeneous (with respect to landcover, topography, soil, and geology) sub-watershed area modeled with PRMS. The HRU is approximately the same scale as the 12-digit hydrologic unit code (HUC) watersheds (Seaber et al., 1987) but is not the same and can be smaller. While the PRMS modeling results were published only for NH, HRU statistics for some areas in MA are available from this study. Additionally, the characteristics of the NH HRUs can be compared with the general characteristics of the MAPC subregions, as shown in Table 2.2.

The ICC subregion, including Boston, has the largest mean area of impervious surfaces (approximately 46%) and the lowest vegetation density (less than 13%). The NSPC and the TRIC subregions have the second and third highest mean impervious surface area, respectively, each with more than 20%. The

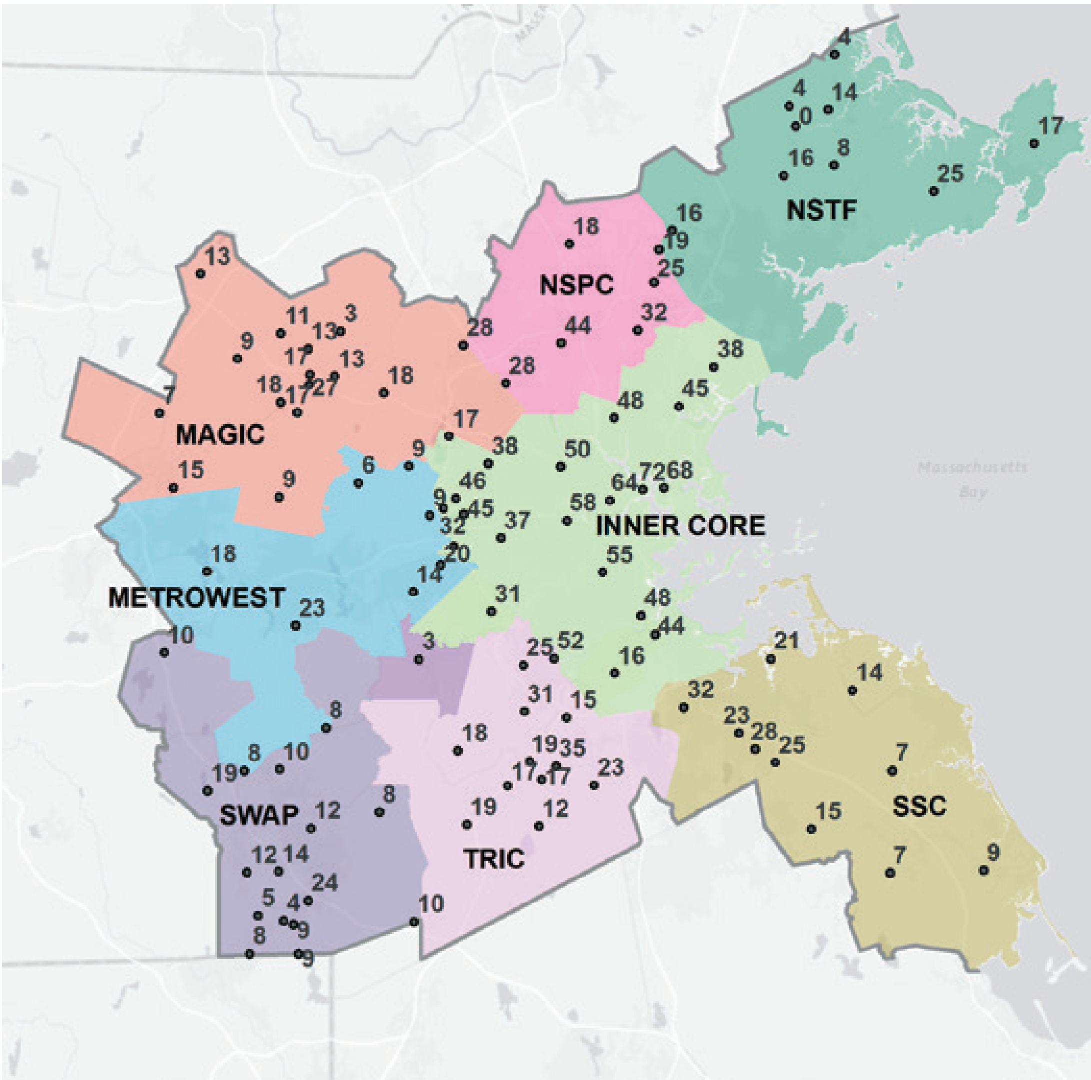
Table 2.2

Land use and topography statistics calculated from HRUs within each MAPC subregion. N is the number of HRUs in each subregion; S.E. is Standard Error in units shown; 1 m = 2.381 ft.

MAPC Subregion		Impervious Area (%)		Summer Vegetation Density (%)		Winter Vegetation Density (%)		Slope		Elevation (m)	
	N	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
ICC	20	46.3	3.4	12.7	3.7	1.9	0.8	0.047	0.003	22.2	3.7
NSPC	6	27.0	3.9	41.2	5.3	7.4	2.2	0.049	0.004	34.3	5.4
TRIC	12	21.8	3.3	45.5	4.8	10.3	1.4	0.044	0.003	44.2	5.7
NSTF	9	18.2	2.7	39.3	5.2	9.4	3.6	0.054	0.005	17.2	2.4
SSC	10	16.2	2.9	49.4	3.5	15.1	2.6	0.038	0.002	27.9	4.1
METROWEST	9	14.0	2.1	51.4	2.8	13.4	1.8	0.056	0.004	59.8	4.7
MAGIC	15	12.2	1.7	51.7	2.8	15.0	1.6	0.047	0.003	56.7	4.1
SWAP	15	10.7	1.4	54.9	2.2	16.0	1.5	0.055	0.003	80.7	3.5

Figure 2.4

Percent impervious area in the MAPC subregions calculated for USGS Hydrologic Response Units (HRUs) (created using data from Bjerklie & Sturtevant, 2018).



● Percent Impervious **MAPC Boundary** **0 5 10 20 km**

Source: Esri, HERE, Garmin. © OpenStreetMap contributors and the GIS user community.

SWAP subregion is the farthest west in the study area with the highest average ground surface elevation. It has the lowest mean impervious surface area and the highest vegetation density. The winter vegetation density is less than one-third of the summer vegetation density in all the subregions. Figure 2.4 shows the spatial distribution of impervious surface area.

Based on this qualitative characterization, the recharge potential appears to be highest in the NSTF and SSC subregions with mean impervious area less than 20%, the summer vegetative cover less than 50%, and it is relatively low topography. These factors will be tempered by the higher ET rates in the wetland areas of these subregions, however.

Four HRUs in southern NH and northern MA with similar land use and topographic characteristics were chosen from the NH PRMS study (Bjerklie & Sturtevant, 2018) to estimate recharge rates in the MAPC study area. These four HRUs include the City of Portsmouth, which is similar to the ICC sub-region's communities with a high percentage of impervious surface and low vegetation density; Hampton, NH, a coastal community similar to the NSTF and SSC communities with wetlands and low topography; Dracut, MA, an inland community with moderately high percentage of impervious surface and vegetated areas like the NSPC communities; and Pepperell, MA, similar to higher elevation and rural MAGIC communities. The statistics for these HRUs are presented in Table 2.3.

Table 2.3

Land use and topography statistics for four HRUs in Portsmouth and Hampton, NH; and Dracut and Pepperell, MA; 1 m = 3.281 ft.

Town	Impervious Area (%)	Summer Vegetation Density (%)	Winter Vegetation Density (%)	Slope	Elevation (m)
Portsmouth	36.5	15.3	0.9	0.02	13
Hampton	11.4	41.6	12.8	0.02	11
Dracut	18.5	39.4	8.5	0.05	40
Pepperell	3.9	61.1	29.0	0.06	69

The average monthly recharge rates calculated for the baseline period of 1981 through 2000 are presented in Figure 2.5. All four HRUs show a typical New England pattern (Bjerklie & Sturtevant, 2018; Desimone, 2004; Mack, 2009; D. A. Masterson et al., 2009) of high recharge rates in the spring and fall dropping to zero from June through August when temperatures and evapotranspiration are highest. Within this pattern there are spatial differences in the maximum rate of recharge and the duration of zero or very small recharge rates. The highest early spring and late fall recharge rate occurs in the Hampton and Pepperell HRUs with the lowest percent of impervious area. Dracut has the lowest early spring and late fall recharge rate along with the longest period of less than 1 cm/month (0.4 in/month) recharge.

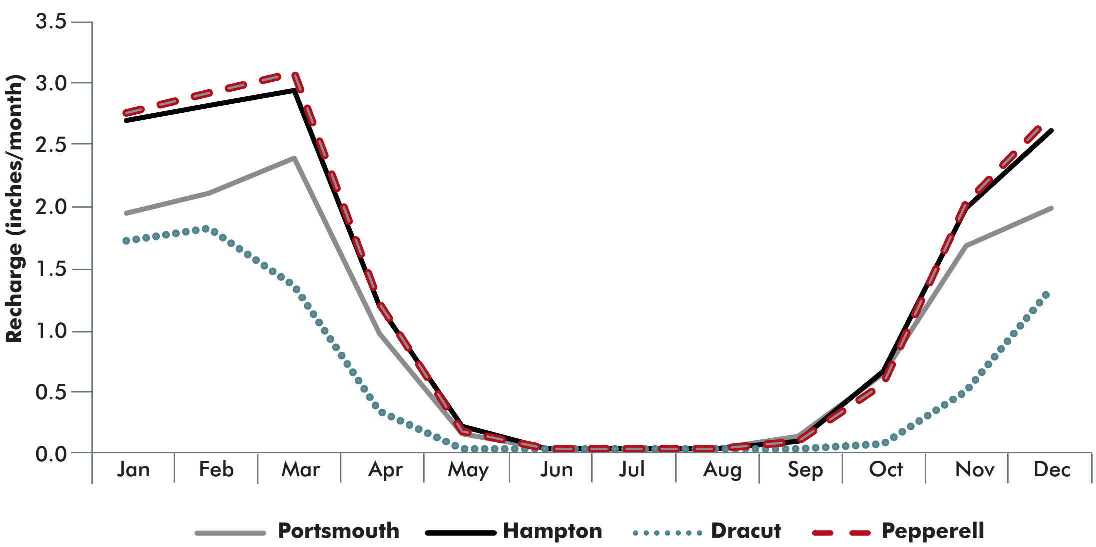
This may be due to a combination of high impervious surface area (reducing winter and spring recharge), coupled with a high percentage of summer vegetative cover (lengthening the period of no recharge) and a relatively high slope (more runoff), all resulting in less aquifer recharge. The recharge variation in the four HRUs described above, as it relates to the pertinent physical characteristics (Table 2.2) of the HRUs, provide insight into the process-based recharge expectations in other areas within the MAPC as a function of similar physical characteristics.

2.4 WATER USE AND WATER SOURCES

Aquifers, public water supply sources, and areas served by the MA Water Resources Authority (MWRA) in the study area are shown in Figure 2.6. The MWRA provides both water and sewer service to 23% of the study area, mostly in the ICC subregion. MWRA provides water without sewer service to 7% of the area and sewer without water service to 11% of the area. Nearly 70% of the MAPC region does not have MWRA water service and relies on local surface water and groundwater sources for their drinking water.

Figure 2.5

Monthly recharge rates for Portsmouth and Hampton, NH, and Dracut and Pepperell, MA, 1981–2000.



Approximately 65% of the area relies on municipal wastewater treatment plants or on-site wastewater treatment systems (OWTSs), i.e., septic systems. Septic systems return much of the wastewater back to the aquifer. Wastewater collected by sewer systems is treated at a wastewater treatment plant and discharged to surface water bodies. Thus, groundwater withdrawn for domestic, commercial, and industrial use is typically lost from the aquifer system in areas served by sewer systems.

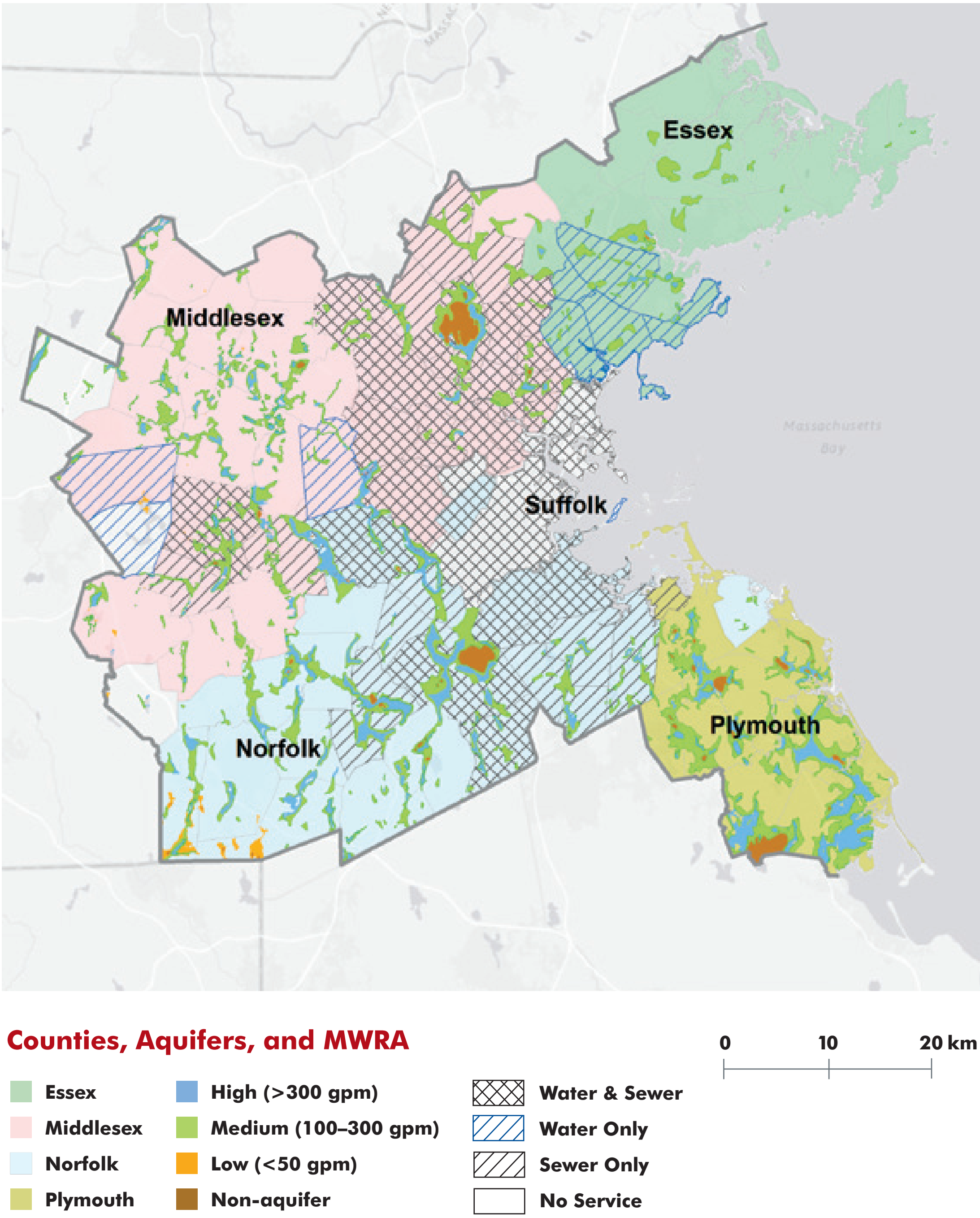
The communities not served by the MWRA that primarily rely on local surface water for their drinking water are in the NSTF subregion where sand and gravel aquifers are scarce and bedrock and glacial till dominate the shallow geology. Communities served by MWRA for emergency use only also rely primarily on local surface-water sources. The remaining MAPC communities rely on groundwater for their drinking water. Aquifers capable of producing greater than 545 m^3/day (100 gpm) make up only 17% of these communities and some aquifers primarily along the south shore, may be vulnerable to saltwater intrusion. Many communities in eastern MA are now using fractured bedrock for domestic (individual) and public (community) drinking water supplies (Arvin, 2010; Boutt et al., 2010). Less than 3% of all fractures conduct groundwater flow and the most transmissive zones occur within the upper 50 m (164 ft), decreasing significantly below 165 m (541 ft) (Boutt et al., 2010; Shapiro & Hsieh, 1998). Groundwater in bedrock roughly follows the topography with the depth to water in a bedrock well averaging 8 m (26 ft) below the ground surface (Boutt et al., 2010). The source of groundwater is dependent, however, on the nature of the fracturing in the bedrock which can be local or regional, and at varying depths.

Population and water-use data for four counties (Essex, Middlesex, Norfolk, and Plymouth) from 1990 through 2015 are presented in Figure 2.7 (U.S. Geological Survey, 2021). The population has increased by 16% in Essex County, 13% in both Middlesex and Norfolk Counties, and 17% in Plymouth County over these 25 years.

Approximately 90% to 95% of the population was served by public water supply in these communities in 2015. The population served by local water withdrawals was the greatest in Middlesex County followed by Essex, Norfolk, and Plymouth Counties. Essex and Plymouth counties had the highest percent of domestic versus commercial and industrial water use of the four counties considered during this period.

Figure 2.6

MWRA service and aquifers in the MAPC study area.

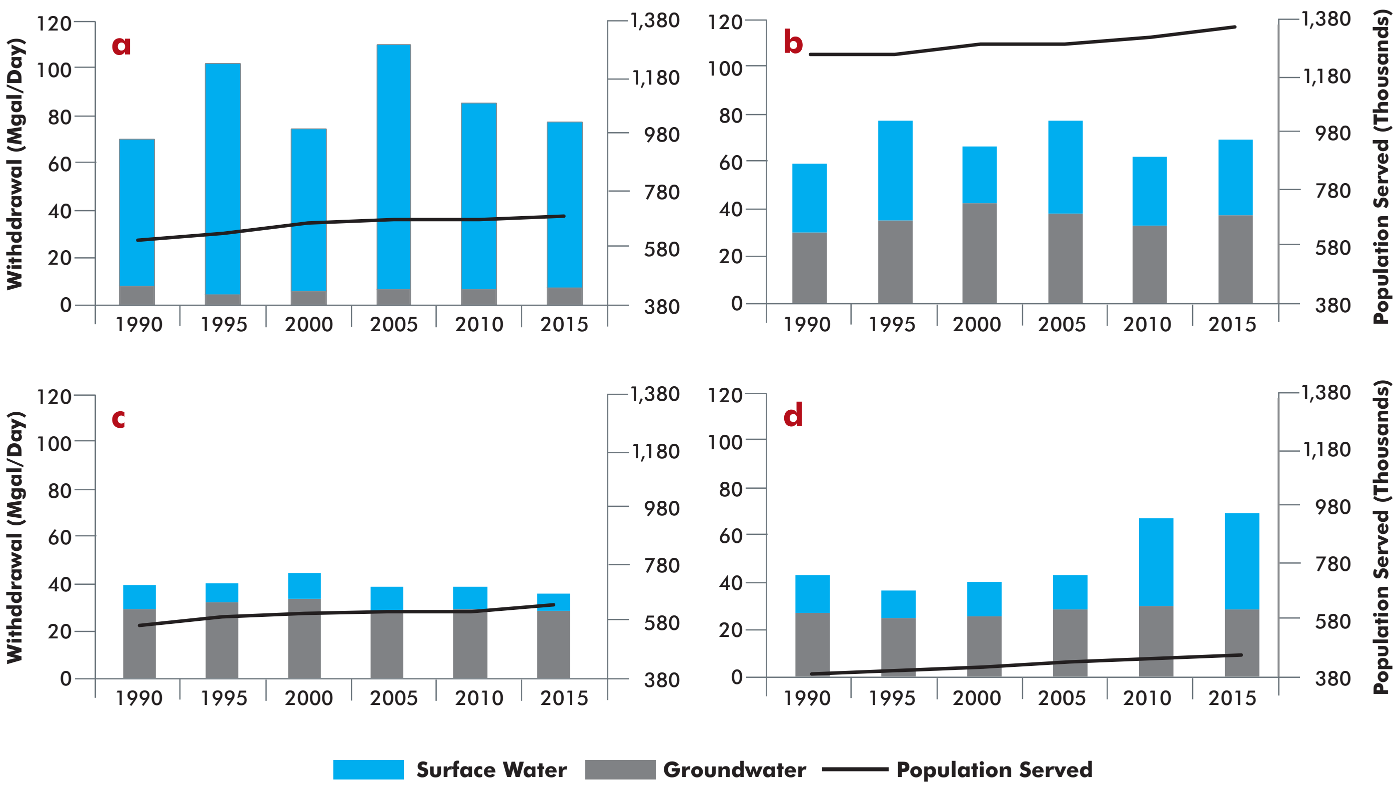


Note: Some communities are served by both local sources and MWRA sources (MassGIS, 2020d).

Source: Esri, HERE, Garmin. © OpenStreetMap contributors and the GIS user community.

Figure 2.7

Water withdrawals for public drinking water supply in a) Essex County, b) Middlesex County, c) Norfolk County, and d) Plymouth County. Groundwater and surface water withdrawals are shown with the total withdrawal indicated by the height of the bar.



Source: Data used in this figure is from USGS (USGS Water Use Data for Massachusetts, accessed October 16, 2019).

The 10-year average population served in the four counties increased 4 to 11% from the period between 1990 and 2000 to the period between 2005 and 2015 but water withdrawals remained relatively constant or declined in Middlesex and Norfolk Counties. This may be due to conservation measures (Tsai et al., 2011) that resulted in a 1% reduction in average per capita use in Middlesex County and a 13% reduction in Norfolk County over the analysis period. In contrast, water withdrawals increased 11%, slightly more than the population increase of 8% in Essex County and 50% in Plymouth County, an increase sharply higher than the population increase of 11%. These counties have lower population densities than the other two counties and the increases were accompanied by a corresponding increase in per capita water use of 3% and 34%, respectively. According to the Massachusetts Department of Environmental Protection's (MADEP) Residential Gallons per Capita Day (RGPCD) records, Plymouth Water Company (in Plymouth County) reported RGPCDs ranging from 82 to 127 gallons per person per day over the period of 2013 through 2019, which is one of the highest RGPCDs in the state (MADEP, 2021). The Massachusetts standard is currently 65 gallons per person per day. Unaccounted for water (UAW) can also be a reason for increased water withdrawals. Some of the water departments in Essex County (Lynn, Manchester, Gloucester) have the highest UAW in the MAPC area, ranging from an average of 29 to 34%. The Massachusetts standard for UAW is 10%. The increases in water withdrawals were observed primarily in the surface-water withdrawals. The groundwater withdrawals remained relatively constant (U.S. Geological Survey, 2021).

Groundwater and surface water withdrawals are shown with the total withdrawal indicated by the height of the bar (U.S. Geological Survey, 2021) (Figure 2.7). Groundwater withdrawals on average accounted for less than 10% of the total water withdrawals in Essex County from 1990 to 2015 possibly due to the relative scarcity of sand and gravel aquifers in this region. Groundwater withdrawals accounted for a much higher percentage in the other counties where sand and gravel aquifers are more prevalent (see Figure 2.6), averaging 52%, 76%, and 57% of the total water withdrawals in Middlesex, Norfolk, and Plymouth Counties, respectively. As mentioned above, the groundwater withdrawals have been more constant than the surface water withdrawals. The groundwater withdrawal coefficient of variation (COV, a measure of relative variability, i.e., the ratio of the standard deviation to the mean) ranged from a minimum of 7% in Plymouth and Norfolk Counties to a maximum of 18% in Essex County during the study period versus the surface-water withdrawal COV that ranged from a minimum of 17% in Norfolk County to a maximum of 57% in Plymouth County.

3. Climate Change and Groundwater

3.1 REVIEW OF EXISTING SCIENCE

Groundwater levels are controlled by many factors including climate, land use, hydrogeologic properties of the aquifer, and the proximity of discharge areas. Groundwater levels are highly sensitive to precipitation and temperature. Factors that decrease evapotranspiration and increase infiltration will increase recharge and groundwater levels, and factors that increase evapotranspiration and decrease infiltration will decrease recharge and groundwater levels (Price, 2011). Increasing groundwater levels will increase water available to wells and to stream baseflow and decreasing groundwater levels will reduce water available to wells and to stream baseflow.

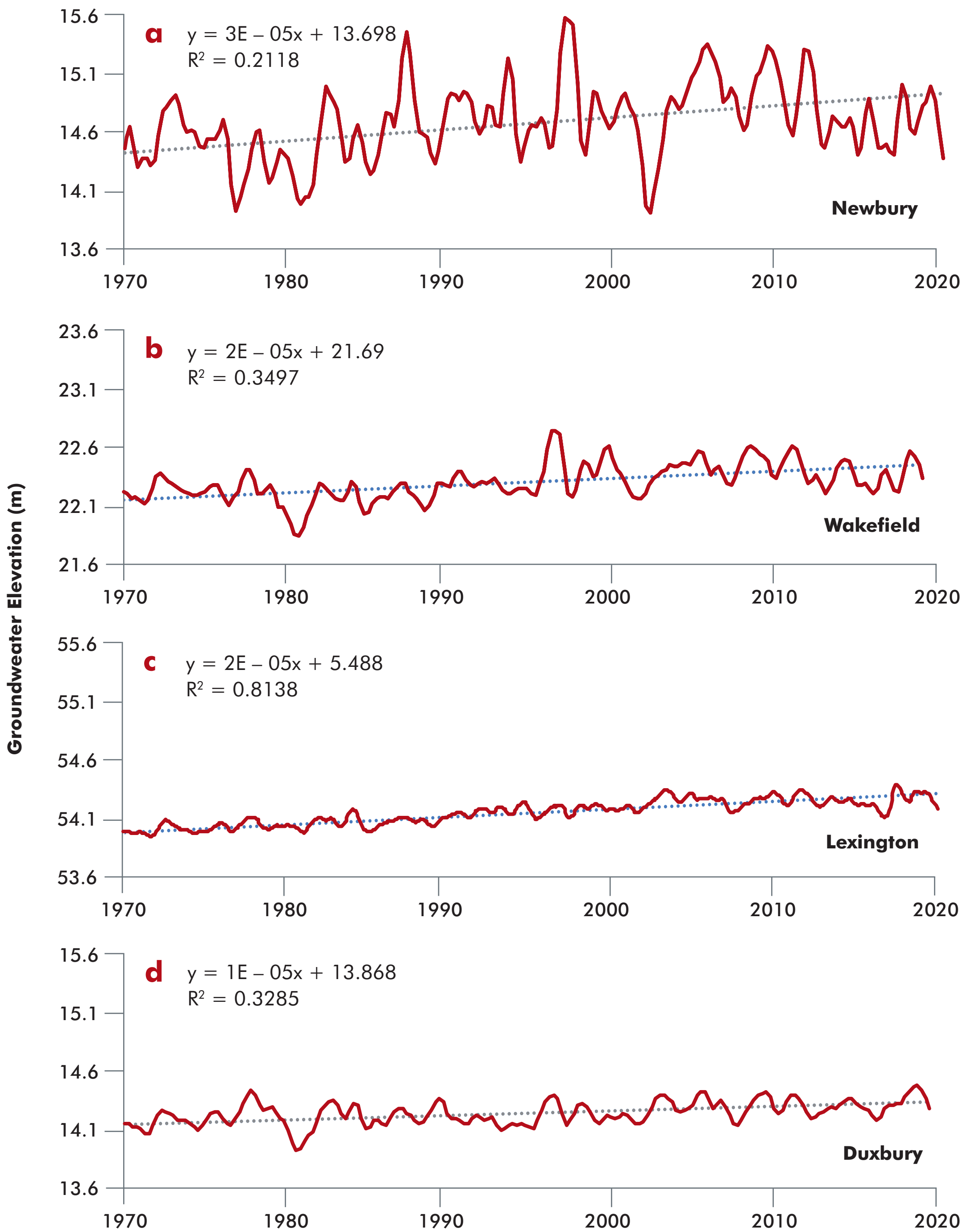
Groundwater elevations from USGS long-term monitoring wells, located away from public water supply withdrawals, were plotted to observe natural trends in four communities in MA, each from a different MAPC subregion: Newbury (NSTF), Wakefield (NSPC), Lexington (MAGIC), and Duxbury (SSC). These USGS monitoring wells are all less than 10 m (33 ft) deep and screened in sand and gravel deposits, except the Newbury well which is screened in till. The Newbury well is approximately 560 m (~1840 ft) from the Parker River and the other wells are between 150 and 190 m (~490 and 620 ft) from minor rivers and wetlands. The time chosen was from 1970 through the present to minimize the influence of the 1960s drought (Bradbury et al., 2002; Hodgkins et al., 2017). The average water table is shallow in the unconsolidated deposits at these locations, with groundwater depths ranging from 0.7 m (2.3 ft) (S.D. of 0.2 m or 0.7 ft) at the Lexington well to approximately 2.5 m (8.2 ft) (S.D. of 0.2 m or 0.7 ft) at the Duxbury well. Seasonality was removed from the data by decomposing the time series into three components: seasonal, trend, and remainder (the residuals from the seasonal plus trend fit) by LOESS (Locally Estimated Scatter Plot Smoothing) using the STL function in R (https://www.rdocumentation.org/packages/stats/versions/3.6.2/topics/stl). The groundwater elevation trends in these wells are shown in Figure 3.1. All four long-term

KEY FINDINGS

- Groundwater is important for water supply, stream baseflows, wetlands, and stormwater management.
- Groundwater levels in the MAPC area have shown increasing trends in USGS long-term monitoring wells. This is due to precipitation increases in the northeast currently out pacing increases in AET.
- Climate change will affect groundwater levels through changes in aquifer recharge associated with changing precipitation, temperature, and snow melt. Seasonal groundwater recharge and stream baseflows are projected to increase in the late fall and early winter with increases in precipitation. Annual average recharge is projected to decrease after year 2030 due to reduced snowpack and evapotranspiration increases in vegetated areas both from rising temperatures and longer growing seasons.
- Assets and natural resources in areas where groundwater is currently shallow may be vulnerable to weakened soil conditions or groundwater inundation (flooding) with future increases in groundwater levels.
- Potential impacts from declining recharge in the spring and declining groundwater levels in the summer include reductions in water supply and summer stream baseflows.
- Groundwater levels will rise near the coast due to Relative Sea Level Rise (RSLR). Groundwater rise will occur farther inland than tidal inundation.
- Potential impacts from rising groundwater at the coast include premature infrastructure failure, water quality degradation, wetland expansion and/or transition, and flooding.

Figure 3.1

Trends in groundwater levels from 1970 to present in USGS monitoring wells: (a) Newbury (NSTF), (b) Wakefield (NSPC), (c) Lexington (MAGIC), and (d) Duxbury (SSC). These wells all show increasing trends consistent with the findings of Boutt (2017); 1 m = 3.281 ft.



monitoring wells show increasing groundwater level trends from 1970 through the present. Water levels in till (Newbury) exhibited the largest variability and the largest increase (1.5 mm in 50 years).

A similar trend of increasing water levels over the same period is found in a nearby bedrock well in Duxbury (SSC), although the groundwater elevations are approximately 1 m (3 ft) lower than in the overlying unconsolidated deposits (shown in Figure 3.1).

Huntington et al. (2018) investigated changes in the Water Cycle Intensity (WCI), defined as the sum of precipitation and actual evapotranspiration (AET) over a specific landscape and time of interest over two 30-year time periods, 1945 to 1974 and 1985 to 2014. They found that WCI is increasing in the northeast driven largely by increases in precipitation. Average annual AET driven by canopy greening, i.e., vegetation growth in areas previously limited by soil moisture, and potential evapotranspiration (PET) driven primarily by temperature, are also increasing; however, in the humid northeast, the precipitation magnitude increased substantially more than PET (Huntington et al., 2018; Kramer et al., 2015). This may explain the increasing groundwater table trends observed over the last 50 years. These trends may not continue, however, as temperatures continue to rise.

The groundwater system serves to store water and release it slowly to streams and other discharge areas. Stream baseflows, seasonal low flows during periods of low or no precipitation, are sustained by soil moisture in the unsaturated zone and groundwater discharge. They also are affected by antecedent precipitation, drainage areas, slope, soil type, and groundwater withdrawals (Dudley et al., 2020; Hodgkins & Dudley, 2011; Kam & Sheffield, 2016; Price, 2011; Sadri et al., 2016). The northeast region has experienced an increasing trend in low flows over the past few decades, consistent with increasing trends in precipitation and soil moisture (Douglas et al., 2000; K. Hayhoe et al., 2006; Hodgkins & Dudley, 2011; Sadri et al., 2016; USGCRP, 2018).

Kam and Sheffield (2016) determined that the 7-day low flow (Q7) showed maximum correlation with a 90-day cumulative antecedent precipitation (AP-90). Q7 is generally considered to be a measure of low-flow persistence and sustained low groundwater and baseflow. They also report an increase in AP-90 in the northeastern U.S. over the period of 1962 through 2011 with a corresponding, though weaker, Q7 increasing trend during this period (Kam & Sheffield, 2016). In a study of annual Q7 stream flows at 2482 U.S. Geological Survey stream gages in the United States, increases in Q7 over the past 100-, 75-, and 50-year time periods were observed in the northeast (Dudley et al., 2020). The percent of streamgages with Q7 increases and decreases for a 75-year period and a 50-year period in MA streamgages is presented in Table 3.1 (Dudley et al., 2020). Statistical significance ($p \leq 0.05$) was determined for three autocorrelation assumptions: independence (INDE), short-term persistence (AR1), and long-term persistence (LTP) of the time-series data. Independence tends to overestimate the number of significant trends and is the least likely. A decreasing number of trends is evident with ARI and LTP (Dudley et al., 2020). Increasing trends in Q7

Table 3.1

Percent of MA stream gages with trends in 7-day low stream flows for 1941 to 2015 (75 years) and 1966 to 2015 (50 years). Statistical significance was determined for independence (INDE), short-term persistence (AR1), and long-term persistence (LTP); N = number of gages; All = all trends regardless of significance (modified from Dudley et al., 2020)

			Increases (%)				Decreases (%)			
Years	Period	N	ALL	INDE	AR1	LTP	ALL	INDE	AR1	LTP
1941–2015	75–Year	35	68.6	25.7	25.7	20.0	28.6	8.6	5.7	2.9
1966–2015	50–Year	44	75.0	11.4	9.1	4.5	25.0	2.3	4.5	0.0

dominate for both time periods with significant increases more evident over the 75-year period than the 50-year period. The more significant trend in the 75-year record is likely due, at least in part, to the inclusion of the extreme drought experienced throughout New England in the early 1960's (Bjerklie et al., 2011).

In the MAPC area, seasonal patterns in aquifer recharge have been found to influence streamflow throughout the year. In Kingston, just south of the SSC subregion, average monthly streamflow in the Jones River measured during the period from 1966 through 2006 followed aquifer recharge with a peak in the spring, a minimum flow in the summer, and a rebound in the fall unlike precipitation which was relatively constant throughout the year (D. A. Masterson et al., 2009). Urban development can either increase or decrease stream baseflow depending on the development specifics in the basin (Dudley et al., 2020; Price, 2011). The Ipswich River basin in the northeast part of the study area is sensitive to water use and has experienced extreme summer low flows detrimental to river ecosystems, wetlands, and its estuary. Drinking water withdrawals and the exportation of sewer water out of the basin (diversions) were found to represent 15 to 20% of streamflow in the 1980s and increases in population and water use would be expected to increase diversions and reduce streamflow (Canfield et al., 1999). More development can also increase impervious area resulting in more runoff and peak stream flows while decreasing aquifer recharge and summer baseflows (Bjerklie & Sturtevant, 2018).

3.2 CLIMATE CHANGE PROJECTIONS

Changes in Aquifer Recharge

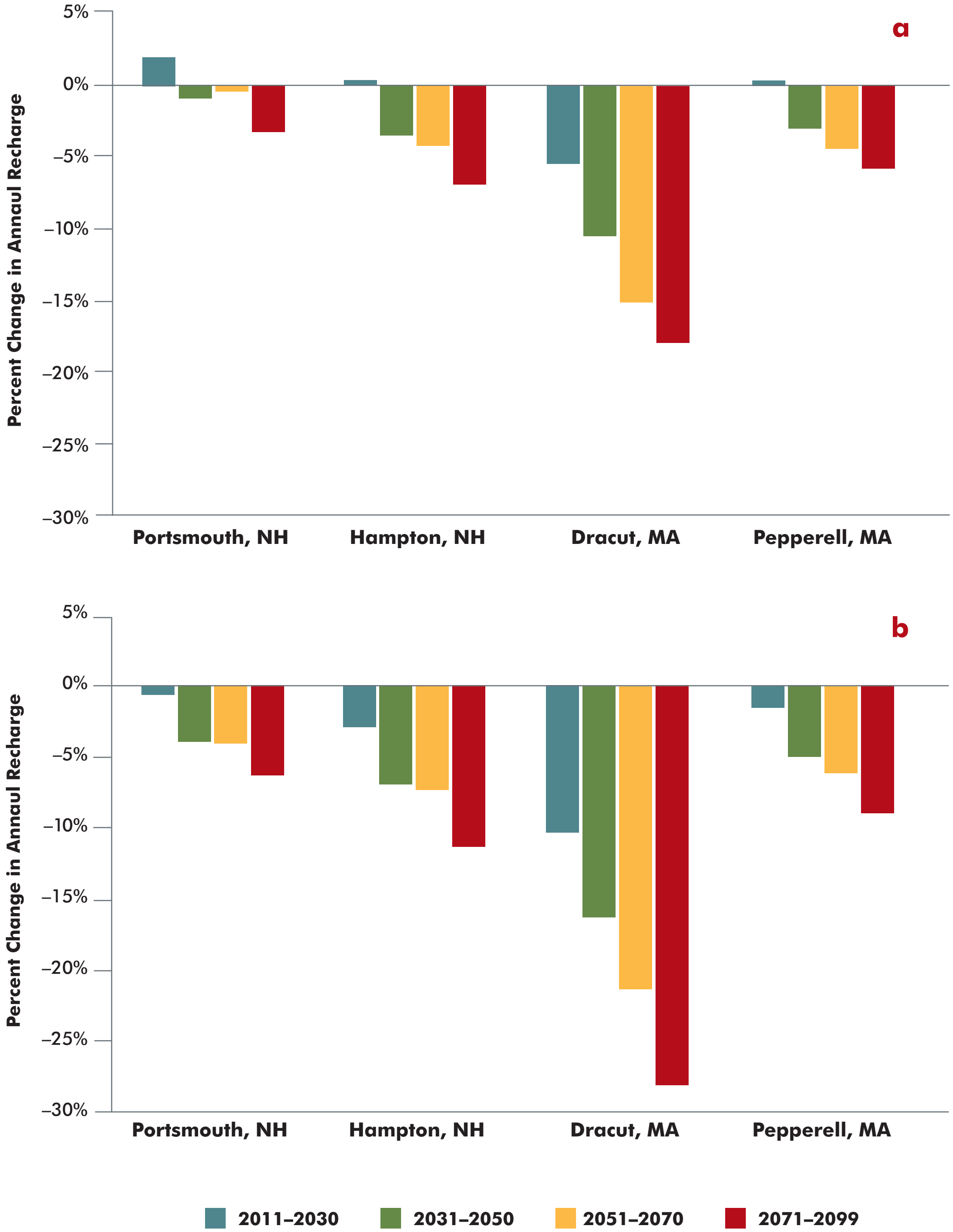
Climate change can influence groundwater availability both through changes in aquifer recharge and water use (Taylor et al., 2013). Precipitation, temperatures, and sea levels are projected to increase in the northeast due to climate change. These factors can result in long-term and seasonal changes in groundwater levels potentially impacting ecosystems, drinking-water supplies, water quality, roads and underground infrastructure, and flooding (Knott, Jacobs, et al., 2018; Knott, Sias, et al., 2019). Difficulties in quantifying changes in aquifer recharge come, in part, from a limited number of studies integrating climate projections with rainfall-runoff models used to estimate recharge, recharge changes related to topography and snow melt, and the difficulty of predicting recharge from changes in the frequency and intensity of extreme precipitation events (Bjerklie & Sturtevant, 2018; Meixner et al., 2016). Despite these challenges, studies of climate change effects on recharge have become more prevalent in recent years. In NH, the USGS Precipitation Runoff Modeling System (PRMS) was used with five general circulation models (GCMs) and two emissions scenarios (RCP 4.5 and RCP 8.5) to simulate changes in streamflow, snow melt, and aquifer recharge from the present (1981 to 2000) to the end of the century (2081 to 2100) (Bjerklie & Sturtevant, 2018). RCP stands for Representative Concentration Pathway referring to a greenhouse gas concentration trajectory adopted by the IPCC (Intergovernmental Panel on Climate Change). RCP 4.5 represents an intermediate scenario where emissions peak around 2040 then decline. RCP 8.5 represents a scenario where emissions continue to rise throughout the 21st century (Stocker et al., 2013).

Annual and monthly recharge projections from the PRMS model were analyzed for the four HRUs (Portsmouth, Hampton, Dracut, and Pepperell) described in Section 2.3 (Table 2.3). The percent change in annual recharge for 20-year to 30-year future periods 2011 to 2030, 2031 to 2050, 2051 to 2070 and 2071 to 2100 relative to the baseline period (1981 to 2000) at these locations for two RCP scenarios (RCP 4.5 and RCP 8.5) is shown in Figure 3.2.

All locations exhibit future decreasing trends in annual average recharge rates with both emissions scenarios. Under the RCP 4.5 emissions scenario, increases in precipitation dominates the recharge response until 2030 with increasing recharge, after which increases in AET dominate with decreasing recharge. Under the RCP 8.5 emissions scenario, recharge is projected to decline at all locations with the magnitude of the decrease increasing substantially after year 2030. The magnitude of the change varies

Figure 3.2

Projected percent change in annual average recharge in Portsmouth and Hampton, NH and Dracut and Pepperell, MA for two emissions scenarios: (a) RCP 4.5 and (b) RCP 8.5.



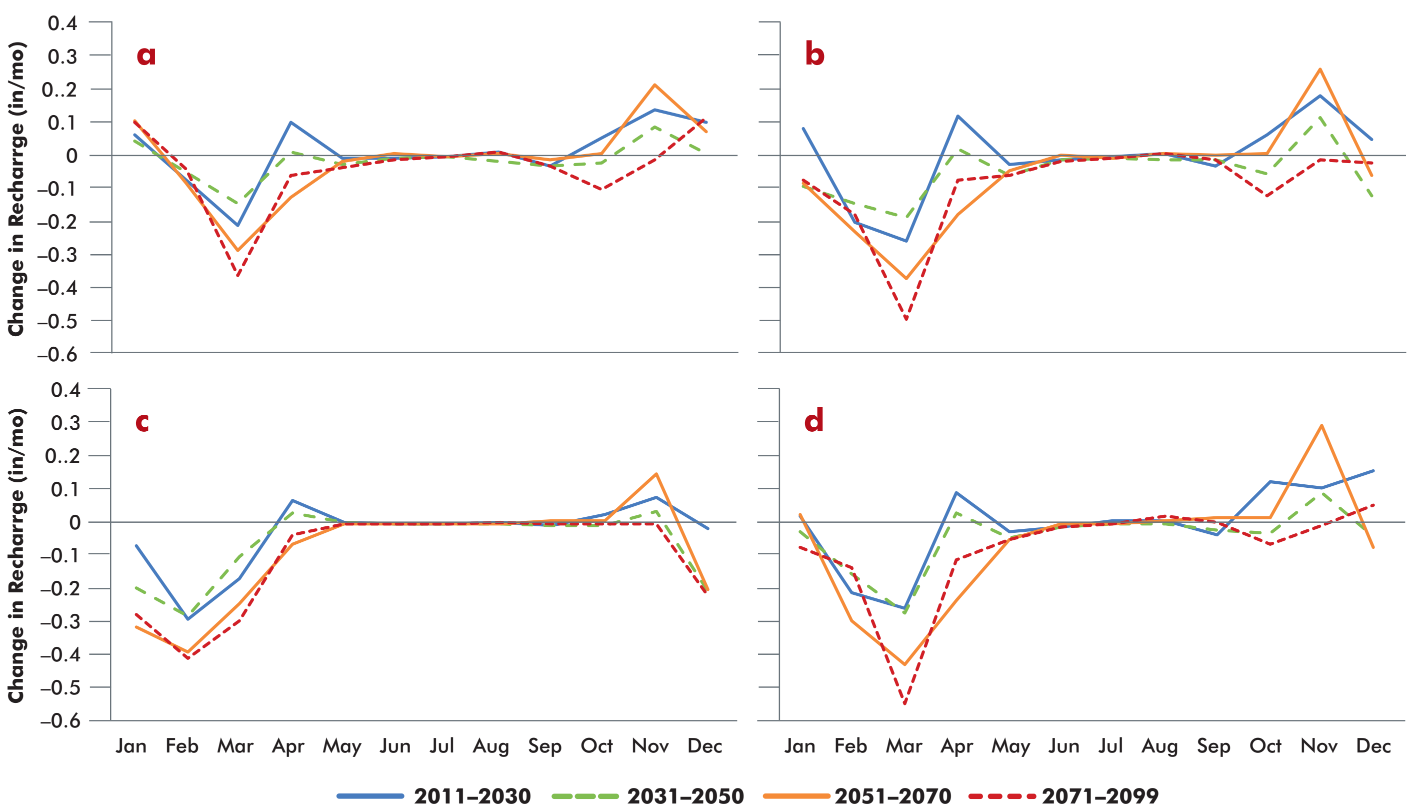
between the sites demonstrating that recharge rates are influenced by local variations in precipitation, land use, geology, and topography. In Dracut, recharge-rate decreases (as a percent of the baseline) are higher than in the other locations because baseline recharge rates are generally lower in this HRU, from a combination of high summer vegetation density and relatively high impervious surfaces and slopes (Table 2.3). Here, annual average recharge-rate decreases are projected throughout the century with a maximum projected decrease of more than 15% at the end of century. In Portsmouth, Hampton, and Pepperell, under the RCP 4.5 emissions scenario, annual average groundwater recharge is projected to increase slightly or stay the same from 2011 through 2030. Beyond 2030, projected recharge-rate decreases are less than 5% before mid-century but increase in magnitude towards the end of the century. The changes in recharge are the smallest for the City of Portsmouth, which has the largest impervious surface percentage and the smallest vegetative cover. Projected recharge-rate declines are the greatest in the HRUs with the smallest percent of impervious area and the largest density of summer vegetation, suggesting that increases in annual AET with increasing temperatures dominate these recharge proejctions, either due to a lengthening growing season or an increasing rate of AET. The magnitude of the projected reduction in aquifer recharge increases at all locations under the RCP 8.5 emissions scenario.

Aquifer recharge varies significantly throughout the year in the northeast. The projected monthly changes in recharge rates for the RCP 4.5 emissions scenario relative to the baseline period (2081–2000) are shown in Figure 3.3.

While the magnitude of change varies between locations and time of year, certain trends are common

Figure 3.3

Projected changes in monthly recharge for four 20-year periods relative to the baseline period (1981–2000) under the RCP 4.5 emissions scenario for a) Portsmouth, b) Hampton, c) Dracut, and d) Pepperel.



in the projections. In the first half of the century, small increases in aquifer recharge are projected in late spring (April through May) prior to the summer period during which all potential recharge is lost to evapotranspiration. These late spring increases become losses after mid-century. A steep decline in aquifer recharge is projected in late winter and early spring (January through March) with the magnitude increasing from mid- to end-of-century. An increase in recharge is predicted at all four locations in late fall and early winter (October through early December) through 2070 and beyond in some locations.

Changes in the duration of climatic seasons (as opposed to seasons defined by the solar calendar) will also influence aquifer recharge. A shortening of the frozen period coupled with increasing precipitation would be expected to increase aquifer recharge during late fall, the winter, and early spring. On the other hand, an increase in the length of the growing season is projected to result in more spring and fall AET coupled with a smaller snowpack reducing aquifer recharge (Ehsani et al., 2017; Knott, Sias, et al., 2019). Increasing temperatures increase PET, but AET depends on water availability. In the typically wet northeast, water availability is usually not a limiting factor, except in late summer or during periods of drought, suggesting that AET will increase with a warming climate (Condon et al., 2020). Increasing temperatures may also increase water use and water withdrawals from aquifers, lowering groundwater levels during the summer months.

This analysis is a preliminary investigation of potential aquifer recharge changes caused by climate change in the MAPC region. The analysis has many limitations. First, only four HRUS near, but not in, the MAPC region were investigated. Second, the PRMS model is calibrated to daily streamflow records and estimated recharge changes are based on daily precipitation and temperature projections. The recharge rate is then averaged over a 20-year period. Extreme precipitation events on hourly timescales are not included in the study. Water withdrawals and returns, frozen ground effects, geologic heterogeneity at spatial scales finer than HRUs, and local weather data are all not included (Bjerklie & Sturtevant, 2018). Considering these limitations, the relative changes in groundwater recharge indicate that groundwater availability to wells and for sustaining baseflows to streams will decline after mid-21st century.

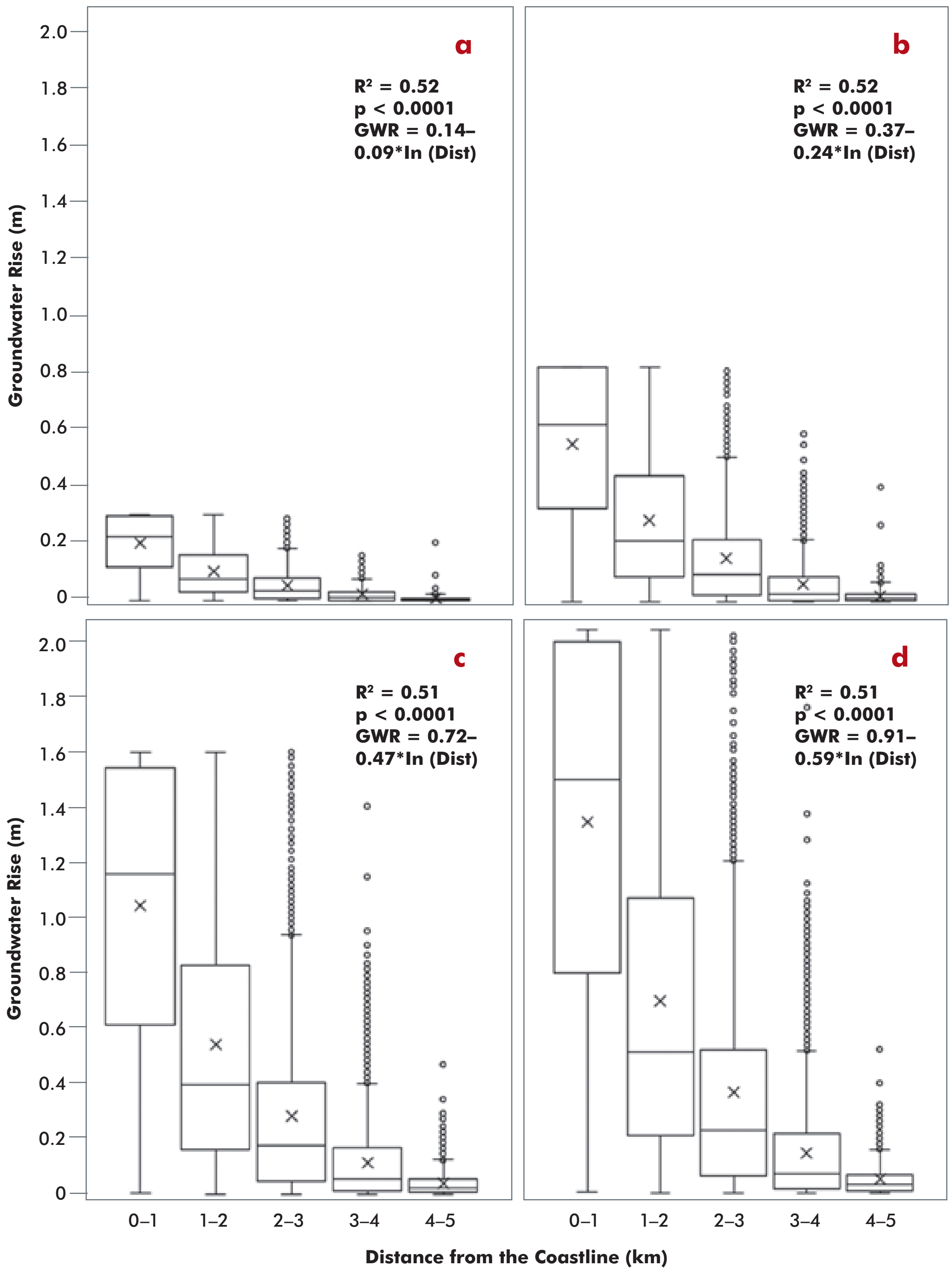
RSLR-Induced Groundwater Rise

Coastal groundwater levels in New England have been projected to rise with RSLR farther inland than tidal-water inundation. Groundwater rises with RSLR because of increasing hydraulic head at the coast (Bjerklie et al., 2012; Knott, Jacobs, et al., 2018; Walter, et al., 2016). The groundwater-rise zone (GWRZ), the area along the coast that will experience groundwater elevation increases caused by RSLR, and the magnitude of the groundwater elevation increases can be determined using groundwater modeling (S. Habel et al., 2017; Knott et al., 2018; Masterson & Garabedian, 2007b; Oude Essink et al., 2010; Walter et al., 2016). Based on groundwater modeling in coastal NH, groundwater levels are projected to rise at distances up to 5 km (3 mi) inland from the shoreline through the end of the century as shown in Figure 3.4 (Knott, Jacobs, et al., 2018). In comparison, tidal-water inundation is projected to extend approximately 1.5 km (0.9 mi) inland from the shore (Rockingham Planning Commission, 2015). By the end of the century, mean groundwater rise relative to RSLR is projected to be 66% between 0 and 1 km (0 and 0.6 mi), 34% between 1 and 2 km (0.6 and 1.2 mi), 18% between 2 and 3 km (1.2 and 1.9 mi), 7% between 3 and 4 km (1.9 and 2.5 mi), and 3% between 4 and 5 km (2.5 and 3.1 mi) from the coastline in the NH Seacoast region (Knott, Jacobs, et al., 2018). There is a large variability around the mean due to factors such as the properties of the geologic materials (Oude Essink et al., 2010), proximity to surface-water discharge areas, and groundwater withdrawals suggesting that mean values should be used with caution (Knott, Jacobs, et al., 2018; Mullaney et al., 2012; Walter, et al., 2016).

Sea level in Boston is projected to rise 0.3 m (1.0 ft) by 2030, 0.5 m (1.6 ft) by 2050, 0.8 m (2.6 ft)

Figure 3.4

Projected groundwater rise with RSLR in coastal New Hampshire for four sea level rise scenarios: a) 0.3 m, b) 0.8 m, c) 1.6 m, and d) 2 m. Each box shows the mean (x), median, interquartile range, and outliers for each 1.0-km distance interval from the coast.



by 2070, 1.5 m (4.9 ft) by 2100, and 3.8 m (12.5 ft) by 2200 with a 0.17 exceedance probability under the RCP 8.5 emissions scenario. Higher RSLRs of 0.4 m (1.3 ft) by 2030, 0.8 m (2.6 ft) by 2050, 1.4 m (4.6 ft) by 2070, 2.7 m (8.9 ft) by 2100, and 9.0 m (29.5 ft) by 2200 are projected with a 0.01 exceedance probability (DeConto et al., 2021; DeConto, personal communication). Scaling with the mean groundwater rise projections from NH with the 0.17 and 0.01 exceedance probability RSLR projections, mean groundwater levels in the MAPC coastal zone have the potential to rise from 1.0 to 1.8 m (3.3 to 5.9 ft) between 0 and 1 km (0 and 0.6 mi), 0.5 to 0.9 m (1.6 to 3.0 ft) between 1 and 2 km (0.6 and 1.2 mi), 0.3 to 0.5 m (1.0 to 1.6 ft) between 2 and 3 km (1.2 and 1.9 mi), and 0.1 to 0.2 m (0.3 to 0.7 ft) between 3 and 4 km (1.9 and 2.5 mi) from the shoreline by the end of the century under the RCP 8.5 emissions scenario. This is an estimate based on the NH modeling work. Since the MA coastline is complex and different from coastal NH with varying surficial geology, coastal geometry, wetlands, stream networks, and land use, actual projections for the MA coastline are needed for adaptation planning.

The magnitude and inland extent of groundwater rise is greatest on peninsulas or islands with tidal water on three or more sides. Groundwater rise will also occur farther inland along tidal estuaries, such as the Charles and Mystic Rivers in the ICC subregion and the Ipswich River in the NSTF subregion. The magnitude and inland extent of the projected groundwater increases is influenced by distance from the coast, the permeability and thickness of the geologic materials, and the proximity to groundwater discharge areas and/or large groundwater withdrawals (Bjerklie et al., 2012; Habel et al., 2014; Knott, Jacobs, et al., 2018; Oude Essink et al., 2010; Walter, et al., 2016).

Areas and assets most vulnerable to RSLR-induced groundwater rise are low-lying areas where the groundwater is already shallow and the separation between the assets (infrastructure, natural resources, basements, OWTSs) and water table is small (Habel et al., 2020; Knott, Daniel, et al., 2018; Walter, McCobb, et al., 2016). RSLR-induced groundwater rise is dampened near streams. The surface-water elevations are controlled by the bank height, resulting in an increased gradient between the rising groundwater and the relatively fixed surface-water elevation. The increased gradient will drive more groundwater discharge to the stream, resulting in increased streamflow and wetland expansion within the GWRZ. On Cape Cod, the groundwater discharge to freshwater streams and wetlands is projected to increase from 50% to 60% of the total outflow with 1.8 m RSLR (Walter, et al., 2016). A 34% increase in streamflow is projected in New Haven, Connecticut with 0.9 m RSLR (Bjerklie et al., 2012), and in coastal NH a 14% increase in net discharge to streams with a corresponding 32% decrease to coastal discharge areas is projected with 2 m of RSLR (Knott, Jacobs, et al., 2018).

4. Potential Impacts of Changing Groundwater Levels

4.1 INLAND COMMUNITIES

Drinking Water Supplies and Aquifer Recharge

Water for drinking, irrigation, and other uses from groundwater sources depends on adequate aquifer recharge to sustain water withdrawals. Most communities in the MAGIC subregion (see Figure 2.1) have private drinking water wells or municipal supplies that include public groundwater sources and some supplementary private wells. Drinking water infrastructure in this subregion has been assessed as highly vulnerable to drought and changes in precipitation. In 2016, nearly 33% of the watershed sub-basins of this subregion were classified as net depleted according to the Sustainable Water Management Initiative (SWMI), meaning that the groundwater withdrawal rates exceed groundwater recharge (MAPC, 2017). A warming climate can exacerbate this condition through reductions in aquifer recharge and declining groundwater levels, either through increasing rates of AET and/or decreasing rates of infiltration, especially in the long term. The decrease in annual recharge rates predicted by the PRMS model suggests that aquifers in these communities may be adversely affected by a warming climate after 2030. This projected reduction in natural recharge—coupled with population and water-demand increases, more impervious area, and higher temperatures—may result in less water available for drinking and other uses if mitigating policies (water conservation and recharge or infiltration basins) are not implemented.

Surface-water supplies, such as the Quabbin and Wachusett reservoirs serving ICC and NSPC communities and the Babson reservoir serving NSTF communities, are also important drinking water sources for the MAPC area. Annual surface-water availability from runoff in the northeast is projected to decrease by 3 to 12% by the end of the century due to increases in AET, despite projected increases in precipitation. Total available water (TAW) for the northeast region (including New England, New York, Pennsylvania, West Virginia, and Virginia), defined as the sum of internal surface water runoff (generated inside the region) and external water resources entering the region, is projected to increase up to 22% in the winter months from December through March, but decline up to 54% during the rest of the year, causing water supply concerns when demand is highest (Ehsani et al., 2017). Reservoir storage increases achieved during the winter months must be maintained to meet summer's high demand but may be lost when releases are made for flood control in the spring. The decrease in surface-water availability during the summer and fall emphasizes need for water conservation and increased groundwater storage capacity (Ehsani et al., 2017).

Overall annual precipitation and extreme precipitation events are projected to increase in this region with climate change (Easterling et al., 2017; Wake et al., 2019). The current extreme precipitation patterns consist of relatively infrequent and intense one-day events in late spring and early fall associated with extratropical storms (Nor'easters) or tropical storms (typically occurring in the fall) (Agel et al., 2015). The effect of extreme precipitation on aquifer recharge is unclear. Some studies suggest that an increase in extreme precipitation events may increase aquifer recharge. Tropical storm Irene, which brought extreme precipitation to the northeast in 2011, produced the wettest consecutive two-month (August and September) period in 123 years of record in western MA. Extraordinarily wet conditions preceding the event produced high antecedent soil moisture content. This event and the preceding extraordinarily wet conditions resulted in more aquifer recharge, water table increases, and long-term increases in streamflow (Boutt et al., 2019). Other studies suggest that more frequent high intensity storms can result in more direct runoff and flooding and less aquifer recharge (Price, 2011). High intensity rainfall events may or may not increase aquifer recharge in an area depending on the soil type and permeability, topography, percent impervious area, antecedent moisture content, and amount of ponding.

Streamflow and Flooding

Groundwater levels and antecedent soil moisture content can affect pluvial (runoff-related) flooding and stream baseflows. An investigation of the effect of antecedent moisture content on peak streamflow in Napa River Basin in California found that a 200-year precipitation event during a dry season (low antecedent soil moisture) generates a peak flow with only a 15-year recurrence interval, while a 7-year precipitation event with saturated soil conditions generates a peak flow with over a 100-year recurrence interval (Kim et al., 2019). In Australia, low or declining antecedent soil moisture substantially reduced the flood potential associated with extreme precipitation events (Wasko & Nathan, 2019). Projected increases in aquifer recharge during the late fall and early winter when AET is low may increase antecedent moisture content for winter and early spring precipitation events and early spring precipitation that will come more as liquid than frozen precipitation as temperatures warm (Ehsani et al., 2017). This may lead to more flooding in low lying inland areas as well as coastal areas faced with RSLR-induced groundwater rise. Flooding can also affect drinking water quality in areas where wellheads are in the floodplain (MAPC, 2017).

Stream baseflows also depend on groundwater levels and groundwater discharge. Trends in low flow statistics can inform water management and regulatory actions including water withdrawals, wastewater discharge, and water releases from dams (Dudley et al., 2020).

Maintaining adequate stream baseflows and groundwater/surface-water exchange is important for water quality, in-stream habitat, effluent dilution, stream biota, and benthic organisms (Rolls et al., 2012; Ficklin et al., 2016). In the northeast, both stream baseflows and stormflows have been increasing in the fall and winter months (Ficklin et al., 2016). Projections of more aquifer recharge in late fall and early winter suggest that baseflow increases during this time of year may continue. In contrast, during the late summer and early fall, a warming climate will increase AET in vegetated areas with shallow groundwater where AET is not limited by soil moisture (Condon et al., 2020; Ficklin et al., 2016). This may increase the frequency and severity of low streamflow events that can be exacerbated by increasing anthropogenic water demand and groundwater withdrawals that reduce natural groundwater levels and aquifer recharge rates (Dudley et al., 2020; Rolls et al., 2012).

4.2 CHANGING GROUNDWATER LEVELS IN COASTAL COMMUNITIES

In coastal areas, changing groundwater levels, either rising or falling, can have serious effects on both the natural and the built environment. Groundwater level declines have been periodically observed in the City of Boston caused by impervious area increases and/or drought or increases in groundwater discharge to surface water through leaks in underground infrastructure and sump pumps. Declining groundwater levels lead to the deterioration of wood pilings that support building foundations built in the City's filled areas, such as Back Bay, and result in building damage and possible destabilization if expensive repairs are not made (Thomas and Vogel, 2012; BGwT, 2022).

Similarly, groundwater rise caused by RSLR may result in premature failure of coastal infrastructure, wetland expansion, water quality degradation, and damage to historic structures (Habel et al., 2020; Jacobs et al., 2017; Knott, Daniel, et al., 2018; J. P. Masterson et al., 2013; Wake et al., 2019; Walter, et al., 2016). Many types of infrastructure can be impacted by rising groundwater levels in areas where the groundwater is already shallow. These include coastal roads, underground utilities, OWTSs, and hazardous waste disposal areas. Historic buildings are also vulnerable to damage from rising groundwater and increases in soil moisture content (Knott, Jacobs, et al., 2019; NYCDEC, 2019; Wake et al., 2019). Groundwater modeling of the MAPC area, which was beyond the scope of this research, is recommended to identify assets at risk from RSLR-induced groundwater rise. Below, we refer to a study conducted in the NH Seacoast that identified coastal roads at risk from RSLR-induced groundwater rise based on the current estimated groundwater depth and projected RSLR-induced groundwater rise within the GWRZ.

Coastal Roads Potentially at Risk from RSLR-Induced Groundwater Rise

As previously discussed, groundwater modeling work done in coastal NH showed that RSLR-induced groundwater rise will occur in areas within 5 km (3 mi) of the shoreline, i.e., the GWRZ (Knott, Jacobs, et al., 2018). Pavements are vulnerable to premature failure when groundwater weakens the underlying supporting base layers (gravel or crushed stone) and subgrade (underlying natural soils) beneath the asphalt. The pavement layers vary in thickness and materials, but typically vary from 0.4 m (1.3 ft) to 1.0 m (3.3 ft) thick (Elshaer, 2017; Knott et al., 2017). Road sections within the NH Seacoast study area were identified as potentially at risk for premature failure and/or increased maintenance as groundwater rises if the current groundwater is 1.5 m (4.9 ft) or less below the road surface and if the underlying surficial geology consists of unconsolidated material that may be weakened by saturation (Knott, Daniel, et al., 2018; Knott et al., 2017; Knott, Jacobs, et al., 2019). The vulnerable roads within the GWRZ are shown in Figure 4.1. Nearly one-quarter of the study-area roads, or 30% of the roads within the GWRZ, (235 km or 146 miles of roadway) are considered vulnerable to premature failure from SLR-induced groundwater rise. While all the functional classifications of roadways (interstate highways, statewide connectors, regional connectors, and local roads) are vulnerable, the local roads are the most vulnerable as they tend to be in low-lying areas closer to the coast (Knott et al., 2017).

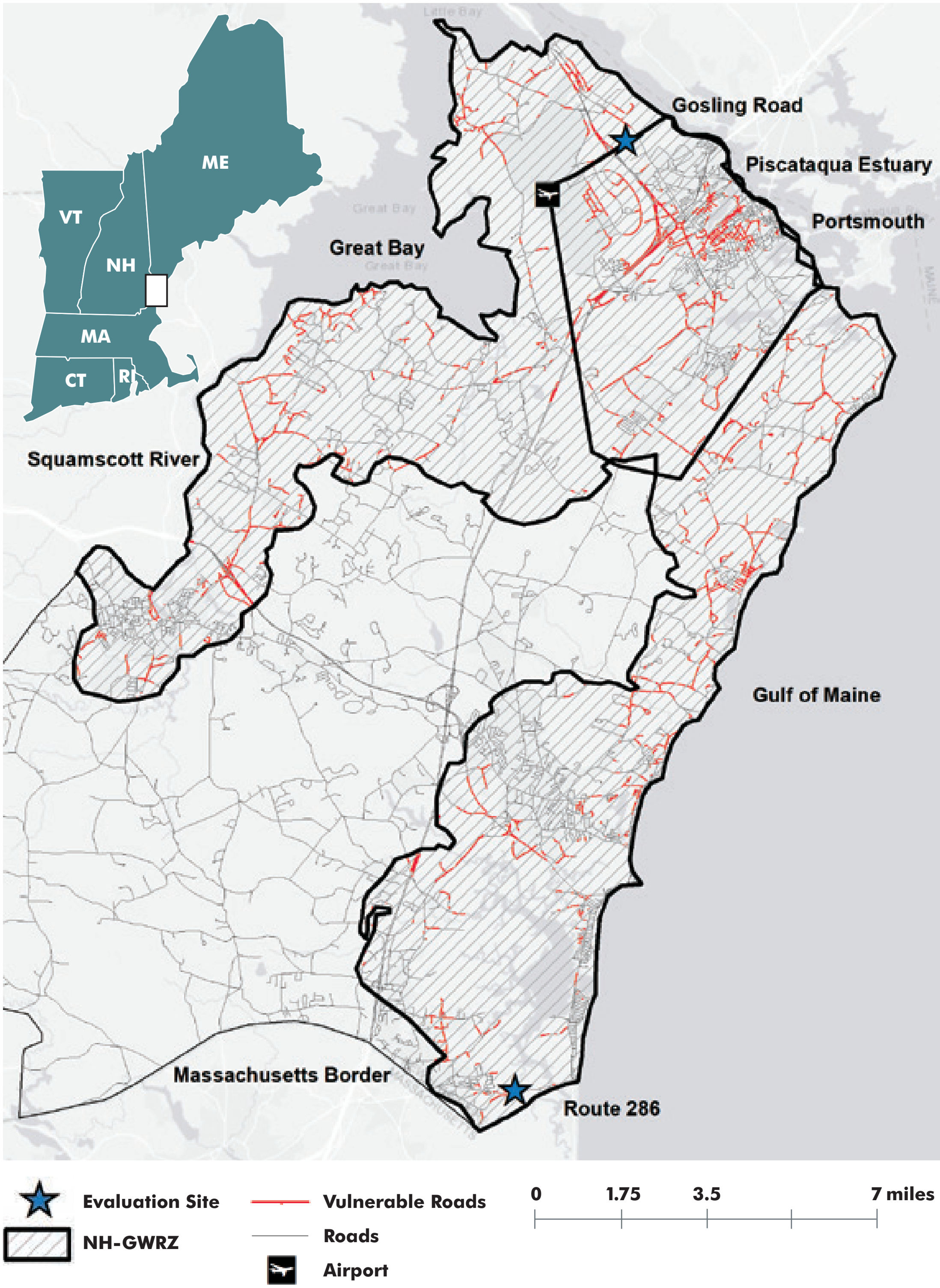
The impact of groundwater rise on pavement performance depends on the structure of the pavement and the degree of saturation in the underlying layers. In addition to being in more vulnerable areas, many of local roads and regional connectors do not have the substantial base layers and/or thick asphalt found in roads designed for higher traffic loads. For example, two pavement cross sections are projected to experience different pavement-life decreases caused by groundwater rise as shown in Figure 4.2.

Gosling Road has a more substantial pavement structure than Route 286 with three supporting base layers (crushed gravel, gravel, and sand) versus a single gravel base layer at Route 286; consequently, the projected reduction in pavement life with groundwater rise is less for Gosling Road than for Route 286. It has been shown that modifications to the pavement structure and maintenance practices can significantly increase pavement resiliency to avoid the high cost of pavement failure (Knott et al., 2017; Knott, et al., 2019).

The MAPC coastal road networks are similarly vulnerable to reduced pavement life and costly pavement failure due to RSLR-induced groundwater rise. This analysis can be done for MA coastal roads as well as vulnerable taxiways and runways at Logan Airport. Groundwater modeling is first used to identify the GWRZ, and the magnitude of groundwater rise in MA coastal communities, which will differ somewhat from the NH GWRZ depending on the geology, coastal geometry, surface-water drainage network, and land use in the MAPC coastal communities. Adaptation planning for these potential pavement impacts, as well as impacts to other critical infrastructure, requires simulation of future changes in the groundwater head and flow, as well as groundwater/surface-water interactions. Other facilities, assets, and natural resources vulnerable to groundwater rise in coastal communities include hazardous waste disposal areas, wastewater treatment plants, OWTSs, underground utilities, and wetlands.

Figure 4.1

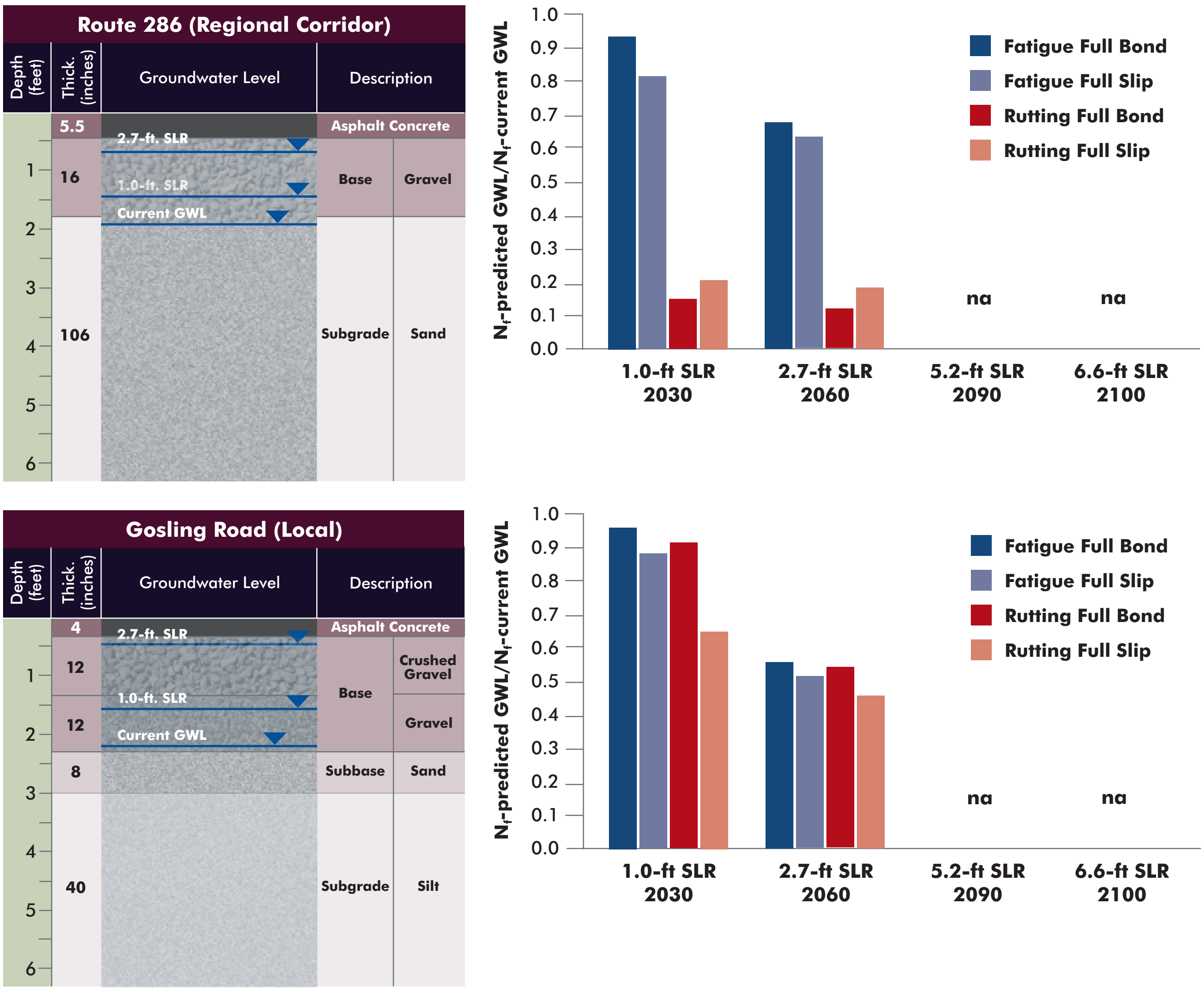
Vulnerable roads (highlighted in red) in coastal NH, i.e., roads in the GWRZ with current groundwater levels less than 1.5 m (5 ft) below the ground surface. The hatched area shows the GWRZ. The blue stars mark two pavement evaluation sites. (Modified from Knott, Daniel, et al., 2018).



Source: Esri, HERE, Garmin. © OpenStreetMap contributors and the GIS user community.

Figure 4.2

Projected pavement life reductions with groundwater rise caused by sea level rise at two pavement evaluation sites in the NH Seacoast. Blue bars and red bars represent the reduction in pavement life due to fatigue cracking and rutting respectively with 1.0 and 2.7 ft of sea level rise. The locations of these evaluation sites are shown in Figure 4.1. (Modified from Knott, Daniel, et al., 2018)



Flooding and Wetland Impacts

Coastal areas are projected to experience increased flooding, also called land inundation, caused by RSLR. This can be caused by direct marine water inundation (MI), the backflow of water through storm drains (DBF), and groundwater inundation (GWI), or two or more of these mechanisms (multi-mechanism) acting together (Habel et al., 2020). GWI is defined here as the flooding of underground infrastructure or the ground surface with rising groundwater tables. Affected infrastructure includes storm and sanitary sewer lines, underground electrical and communication lines, basements and building foundations, and OWTS's. In urban Honolulu, Habel et al. (2020) found that extreme high tides (king tides) currently

Table 4. 1.

Percentage of inundated area in Honolulu with four flooding thresholds at the Honolulu tidal gauge relative to MHHW. KT2017 is the 2017 King Tide elevation, GWI is Groundwater Inundation, MI is Marine Water Inundation, DBF is Drainage Backflow. (Modified from Habel et al., 2020)

		Single Mechanism			Multi-Mechanism			
Flood Type	Sea Level (m)	GWI Only	MI Only	DBF Only	GWI & MI	GWI & DBF	MI & DBF	GWI, MI, DBF
KT2017	0.35	26.33	2.39	0.40	41.98	10.10	0.65	18.15
Minor	0.52	23.32	2.66	0.84	36.65	16.14	0.95	19.43
Moderate	0.82	24.74	2.31	3.74	19.56	7.52	3.17	38.95
Major	1.19	15.19	3.61	0.55	17.55	2.35	6.05	54.71

flood low-lying densely populated areas. Of this flooding, more than a quarter is GWI and less than 3% is from MI acting alone. The flooding is projected to increase substantially with increasing sea levels. The percentage of total inundated area attributed to these mechanisms individually and in combination were determined for minor, moderate, and major flooding thresholds relative to mean higher high water (MHHW) in Honolulu. The future flooding thresholds were based on NOAA's intermediate SLR predictions (Sweet et al., 2017). These results are presented in Table 4.1.

GWI represents the largest single component of land inundation for all the flood types in this urban area. These results clearly show the importance of considering groundwater rise and GWI when planning for and adapting to RSLR in coastal areas. When considering these mechanisms acting together, GWI with MI make up the largest percentage of inundated area for the smaller floods. DBF, in combination with GWI and MI, becomes more important in the larger floods. The Greater Boston area is also vulnerable to impacts from rising groundwater. Like Honolulu, the Greater Boston coastal areas rely heavily on coastal barriers, reclaimed wetlands, and channelized gravity drainage, resulting in an increased vulnerability to GWI, MI, and DBF (Habel et al., 2020).

Wetlands are generally recognized for their value in flood control, nutrient attenuation, biodiversity, fisheries production, and recreation (Linhoss et al., 2015; Walters & Babbar-Sebens, 2016). Links between coastal wetlands and fisheries have been made globally and in the Northeast U.S. with saltwater commercial and sport fisheries depending on coastal estuaries and their wetlands for food sources, spawning grounds, nurseries for young, refuge, and clean water (Boesch & Turner, 1984; Graff & Middleton, 2001; Minello & Rozas, 2002). RSLR can result in the migration, transition, or drowning of coastal wetlands (salt marshes) depending on the nature of the adjacent land and the sedimentation rate relative to the rate of RSLR. Salt marshes with adequate sedimentation will naturally adapt to RSLR if marsh surface-elevation increases keep pace with RSLR (Costanza et al., 1990). In Rhode Island (RI), it was determined that the rate of RSLR over the period 1999 through 2015 was greater than elevation increases in most salt marshes (Raposa et al., 2017). In the San Francisco Bay Estuary, marsh transition from high marsh (Spartina patens) to low marsh (Spartina alterniflora) occurred with a RSLR rise rate of 1 to 1.6 m (3.3 to 5.2 ft) per century. The conversion of high marsh to low marsh and the loss of low marsh by drowning and light reduction (Kirshen et al., 2013; Smith, 2015) are likely to negatively impact coastal fisheries if proactive management policies are not implemented. Barriers designed to protect properties from flooding can inhibit or prevent marsh migration resulting in greater overall marsh loss and consequential impacts to fisheries (Passeri et al., 2015).

Both freshwater wetlands and salt marshes within the GWRZ are vulnerable to RSLR- induced groundwater rise (J. P. Masterson et al., 2014; Wilson et al., 2012). The main factors influencing the wetland habitat value for fisheries are water quality, water quantity, cover, substrate, interspersion, and salinity (Minello & Rozas, 2002; Moffett et al., 2012). Water depth and the duration of root-zone saturation are important in determining wetland plant health, plant distributions, and habitat (J. P. Masterson et al., 2014; Rheinhardt & Fraser, 2001). Many of these factors are directly related to long-term changes in groundwater levels and the interaction between fresh groundwater and tidal surface water (Bohlen et al., 2013; Hemond & Fifield, 1982; Linhoss et al., 2015; J. P. Masterson et al., 2014). In Portsmouth, NH, RSLR-induced groundwater rise is projected to increase the surface area of freshwater wetlands 3% by 2030, 10% by mid-century, and 19 to 25% by the end of the century with the RCP 8.5 scenario. In other areas, depending on the topography and the wetland plants' tolerance for hydrological alterations, wetland vegetation may drown or transition into other wetland species. This raises the question of whether the MA Wetlands Protection Act, which prescribes a horizontal 100-foot buffer zone from the wetlands' edge will be sufficiently protective of MAPC wetlands as groundwater rises with RSLR in MA coastal communities.

Water Quality Degradation

Saltwater Intrusion

Large volume water withdrawals from wells near the shoreline can cause saltwater intrusion into aquifers and drinking-water supply wells in some areas. Sea level rise can exacerbate saltwater intrusion by narrowing the freshwater lens in areas of increasing groundwater discharge at streams or wetlands (Langevin & Zygnerski, 2013; J. P. Masterson & Garabedian, 2007; Walter, eet al., 2016). In a study of a municipal water supply in coastal southeastern Florida, Langevin & Zygnerski (2013) found that the shallow aquifer was especially vulnerable to RSLR-induced saltwater intrusion for several reasons.

1. The aquifer is head controlled, i.e., it is an unconfined aquifer hydraulically connected to canals, the unsaturated zone is thin, and evapotranspiration is high. These factors keep the groundwater piezometric head relatively constant as sea level rises. Since the groundwater piezometric head cannot rise with RSLR, the saltwater/freshwater interface moves inland (saltwater intrusion).
2. Saline water travels through canals inland to or near the municipal well fields.
3. The shallow aquifer is very permeable, reducing the seaward hydraulic gradient.
4. Southeastern Florida has a high-water demand reducing the hydraulic head near the coast and increasing the potential for saltwater intrusion. (Langevin & Zygnerski, 2013). The MAPC area does not have these same risk factors as a whole; however, some communities with municipal water supplies in sand and gravel aquifers near the coast, with prevalent wetlands and streams, and relatively high evapotranspiration may be at risk for RSLR-induced saltwater intrusion. The Town of Duxbury recognizes this potential threat, stating that saltwater intrusion into its aquifer from "rising sea levels and increased pumping" is a "major area of concern" (Duxbury Master Plan, 2019). Public water supply wells in Cohasset, Scituate, and Marshfield may also face an increasing risk from saltwater intrusion with increasing water demand and RSLR.

Saltwater intrusion can also result in harm to freshwater ecosystems associated with barrier islands and barrier beaches (J. P. Masterson et al., 2014). As the freshwater/saltwater interface moves inland with RSLR, the freshwater lens above the saltwater shrinks and the vadose zone thins, increasing both the salinity and duration of root-zone saturation. These groundwater changes have consequences for ecosystem extent and function (Masterson et al., 2014). Freshwater wetlands, salt marshes, and barrier beaches on both the north shore and the south shore of MA are vulnerable to changing sea levels, groundwater levels, and salinity.

On-Site Wastewater Treatment Systems (OWTSs)

OWTSs, also known as septic systems, within the GWRZ where groundwater is shallow are vulnerable to premature failure with RSLR-induced groundwater rise. OWTSs receive wastewater that contains a mixture of contaminants, such as nitrogen (N), phosphorus (P), pathogens, a broad group of organic chemicals including pharmaceuticals, and PFAS compounds from individual homes or businesses. These contaminants move from the OWTSs into the groundwater and travel to sensitive receptors such as wells or surface-water bodies (Lusk et al., 2017; Schaider et al., 2016). While groundwater and surface-water contamination from OWTSs is already a concern, the increased contamination associated with rising groundwater in coastal communities has recently gained attention (Cooper et al., 2016; Iverson et al., 2015; Mihaly, 2018). In Honolulu, Habel et al. (2020) predict that the percentage of fully flooded OWTSs will increase from approximately 1.6% during king tide flooding in 2017 to 16 percent by the end of the century with an intermediate RSLR scenario (Sweet et al., 2017). On a North Carolina barrier island, where the groundwater table is shallow (from 5 to 10 feet deep), OWTSs will be compromised on 43 to 54% of the land serviced by these facilities by the end of the century (Manda et al., 2015).

Title 5, OWTS regulations in MA, states that the minimum vertical separation distance between the seasonal high-water table and the bottom of the soil absorption system (leaching field) must be 1.2 m (4 ft) for soils with a percolation rate more than 0.8 min/cm (2 mins/in) and 1.5 m (5 ft) for soils with a percolation rate less than 0.8 min/cm (2 mins/in) (Section 15.212—Depth to Groundwater, 310 Mass. Reg. 15.212). This allows for an adequate soil treatment zone consisting of unsaturated soils beneath the leaching field. In areas where the groundwater table is already shallow, RSLR-induced groundwater level increases will reduce the soil treatment zone to unacceptable levels and, in some locations, groundwater may inundate the soil absorption system resulting in OWTS failure. RSLR and future expansion of wetland areas caused by groundwater rise will also reduce horizontal separation (setbacks) between OWTSs and freshwater wetlands, coastal bank, or salt marshes (Section 15.211—Minimum Setback Distances, 310 Mass. Reg. 15.211).

The coastal communities in the MAPC area vary widely in sewage treatment capabilities. Many of the communities have MWRA or municipal sewer services and wastewater treatment plants. Beverly, on the north shore, provides sewer services to approximately 95% of all residences (PlanBeverly, 2021). In contrast, only 30% of all housing units in Scituate were sewered in 2004 and soils in some areas are unsuitable for traditional septic systems (Scituate Master Plan, 2004). The Town of Duxbury does not operate a wastewater treatment plant and all residences and businesses depend on either individual or shared OWTSs for sewage treatment (Kleinfelder, 2013). The Plymouth-Carver aquifer beneath the Town of Plymouth (just south of the SCC subregion) is a "sole-source aquifer" meaning that it is the only source of drinking water for the towns of Plymouth and Carver. In addition to drinking water dependency on groundwater, numerous freshwater ponds and wetlands are groundwater fed and are sensitive to groundwater levels and quality. The disposal of wastewater through OWTSs is prevalent and raising concerns about nitrate contamination of groundwater (Barry et al., 2017).

Sewer systems located within the GWRZ may also experience impacts as groundwater rises with RSLR. Some communities with ample sewerage are facing problems with aging infrastructure. For example, approximately 80% of the water and sewer pipes in Manchester-by-the-Sea are 50 years or older. Rising groundwater can infiltrate aging sewage infrastructure and overwhelm or damage sewage treatment plants with excessive water volumes and in some cases saline water entering the treatment systems (Flood & Cahoon, 2011).

5. Summary, Open Questions, and Data Gaps

Groundwater is important for human health and the environment but has often been overlooked in the development of climate change adaptation strategies. This is because groundwater is out of sight, it moves slowly and because changes in groundwater levels are not as dramatic as extreme flooding events, coastal storms, and storm surge. The importance of groundwater for drinking water, natural resources, and streamflow is well documented. Groundwater levels are also important considerations in the design of pavements, underground infrastructure, foundations, OWTSs, and in the remediation of hazardous waste disposal areas. Groundwater is especially important in the wet northeast, where groundwater levels tend to be shallow and impactful. It is typically assumed that on average groundwater levels are not changing. This is no longer true with climate change.

Climate change will affect groundwater levels due to changes in aquifer recharge associated with changing precipitation, temperature, and snow melt. Groundwater levels in the MAPC area have shown increasing trends in USGS long-term monitoring wells. This trend is projected to continue in the near term with relatively constant or slightly increasing annual average recharge. Later, in the century, however, annual groundwater recharge is projected to decline due to increases in evapotranspiration. A seasonal shift in groundwater recharge is projected, with recharge increases in the late fall and early winter and decreases during the rest of the year. Potential effects from declining aquifer recharge and groundwater levels in the spring and summer, coupled with increasing demand, may produce reductions in water supply and summer stream baseflows. These potential problems can be somewhat mitigated by proactive measures to increase aquifer recharge and groundwater storage.

In inland and coastal communities, the short-term projected increases in groundwater levels and the long-term projected increases in the fall and early winter may result in more pluvial flooding by increasing antecedent soil moisture before extreme precipitation events. This will be exacerbated by a shortening frozen period in which winter precipitation will occur more as rain than snow as the climate warms. Assets and natural resources in areas where the groundwater table is currently high are vulnerable to weakened soil conditions or groundwater inundation (flooding) as groundwater levels rise. Coastal MAPC communities will experience substantial groundwater level increases caused by RSLR that will extend farther inland than tidal inundation. In low-lying areas where groundwater tables are already high, potential impacts from rising groundwater include premature infrastructure failure, water quality degradation (including saltwater intrusion), wetland expansion and/or transition, and groundwater inundation (flooding). The MA coastline is complex with varying surficial geology, coastal geometry, wetlands, stream networks, and land use. Groundwater modeling to produce projections of RSLR-induced groundwater rise for the MA coastline is needed for adaptation planning.

A complete assessment of climate change effects and future land-use changes on the groundwater system in the MAPC area requires improved projections of aquifer recharge, soil moisture, and runoff. Three-dimensional groundwater modeling is a useful tool for simulating groundwater systems and groundwater/surface-water interactions (Harbaugh, 2005; Rumbaugh & Rumbaugh, 2011). Traditionally, groundwater models have been used to assess potential and existing drinking water supplies or to investigate groundwater contamination from hazardous waste disposal sites and to evaluate remediation alternatives. They are also well suited to investigate complex hydrologic interactions with a changing climate and changing land-use. Groundwater modeling can integrate aquifer recharge, evapotranspiration, water withdrawals, groundwater/surface water interactions, sea level, and contaminant transport (including saltwater intrusion). It can be used to simulate the effects of long-term changes in one or more of these parameters and can also predict solute transport from present or future contamination sources (Bjerklie et al., 2012; DeSimone et al., 2002; Habel et al., 2020; Kirshen, 2002; Knott, Jacobs, et al., 2018; Walter, et al., 2016). The modeling results can then be used with other tools, such as pavement performance or surface-water hydraulics models, to identify vulnerable assets and adaptation strategies in the MAPC area, like the analysis done for coastal roads in the NH Seacoast (Knott, Jacobs, et al., 2019; Knott, Jacobs, et al., 2018; Knott et al., 2017).

Changes in aquifer recharge caused by climate change are difficult to quantify. This is due to a limited number of studies integrating climate projections with aquifer recharge numerical modeling and estimation techniques. Accurate and useful models require data for model calibration and verification; consequently, more groundwater monitoring is necessary for modeling support as well as for the management of freshwater resources and flooding potential under climate change (Taylor et al., 2013). With increases in extreme precipitation projected for the northeast, more field studies are needed to quantify the effect of these events on aquifer recharge, groundwater levels, and flooding (Bjerklie & Sturtevant, 2018; Meixner et al., 2016).

More studies combining climate model output with rainfall-runoff models is essential. Antecedent soil moisture content has been shown to be important in predicting floods and aquifer recharge suggesting that a coupling of groundwater and surface water models that includes the unsaturated zone is needed. Boutt (2017) suggests that upland glacial till aquifers be included in rainfall-runoff models for investigating the impacts of climate change on groundwater. They represent 70% of the active groundwater storage, i.e., the annual or multi-annual water flux into and out of downgradient aquifers, in the glaciated northeast. The USGS PRMS model constructed for neighboring NH (Bjerklie & Sturtevant, 2018) can be readily expanded to the MAPC communities and MA at large to estimate changes in aquifer recharge, snow melt, and streamflow. Groundwater systems should also be integrated into land-surface models (LSMs) used in General Circulation Models (GCMs) as water-table depths from 2 to 7 m (7 to 23 ft) have been found to influence land-energy fluxes (Ferguson & Maxwell, 2010; Taylor et al., 2013).

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Climate Change Impacts on Groundwater in MAPC Communities

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Groundwater is important for human health and the environment but has often been overlooked in the development of climate change adaptation strategies. This is because groundwater is rarely visible, and because changes in groundwater levels are not as dramatic as extreme flooding events, coastal storms, and storm surge. The importance of groundwater for drinking water, natural resources, and streamflow is well documented. Groundwater levels are also important considerations in the design of pavements, underground infrastructure, foundations, on-site wastewater treatment systems, and in the remediation of hazardous waste disposal areas. Groundwater is especially important in the wet northeast, where groundwater levels tend to be shallow and impactful. It is typically assumed that on average groundwater levels are not changing. This is no longer true with climate change.

Groundwater is the world's largest distributed source of fresh water and is important for both ecosystems and human consumption. Groundwater levels are affected directly by recharge (water infiltrating the ground surface and moving into the groundwater system) and water losses through groundwater discharge to surface water bodies and groundwater withdrawals from aquifers. Many factors influence the amount of groundwater recharge that occurs. These include precipitation, temperature, evapotranspiration, land cover and land use, soil moisture, and topography. Climate change is affecting the global water cycle by increasing rates of ocean evaporation, terrestrial evapotranspiration, and precipitation.

Precipitation, temperatures, and sea levels are all projected to increase in the northeast due to climate change. These factors can result in long-term and seasonal changes in groundwater levels potentially impacting drinking water supplies, water quality, the useful life of pavements and underground infrastructure, and flooding.

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