Effects of Groundwater Withdrawals for Water Bottling and Municipal Use, Wards Brook Valley, Maine and New Hampshire

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Contents

Abstract

Hydrologic models for the Wards Brook valley near Fryeburg, Maine were developed for historical (2016 – 2021) and hypothetical future conditions (2046 – 2065 and 2080 – 2099) to understand the effects of groundwater withdrawals for bottled water and municipal use on hydrologic conditions (stream base flows and groundwater levels). Analyses showed that the simulated base flows in Wards Brook were reduced because of pumping for both municipal water supplies and for water bottling, and about half of the total pumping impact on the base flows in Wards Brook was from the bottled water extraction. Simulated flows were greater than the minimum recommended streamflow of 2,180 cubic meters per day (400 gallons per minute) throughout the historical period. Simulated groundwater levels at two of three nearby ponds (Round Pond and Davis Pond) were minimally affected by pumping conditions, and effects were primarily from the municipal well closest to the ponds.

Several estimates of future projected recharge were used to understand the potential effects of groundwater withdrawals on hydrologic conditions under multiple hypothetical climate conditions. Annual projected recharge rates in the mid- and late-21st century from two climate scenarios (stabilized greenhouse-gas emissions and high greenhouse-gas emissions) were similar to rates for 2016 – 2021. However, monthly recharge patterns for the future periods shifted toward more recharge in the winter months (December, January, and February) and less recharge in April, May, and October relative to 2016 – 2021.

The lowest mean monthly base flows from the future emission scenarios all remain larger than the minimum recommended streamflow and indicate no long-term declines in flow relative to historical conditions. However, simulated base flows during hypothetical 3-year drought

scenarios declined below minimum recommended streamflow during the summer months in the stabilized- and high-emission scenarios in the mid-21st century. Although water is generally plentiful in the Wards Brook valley, reduced pumping may be needed to maintain streamflows in Wards Brook under future climate conditions similar to modeled drought scenarios.

Introduction

As a result of the Nation's growing population and increased consumption of bottled water, the water bottling industry continues to grow. The volume of water bottled increased almost every year between 1977 and 2018, declining only during the years of the Great Recession (-1.1 percent in 2008 and -2.5 percent in 2009)(Rodwan, 2019). According to the International Bottled Water Association (2022), "In 2019, bottled water ranked as the largest beverage category by volume in the United States for the fourth consecutive year following a remarkable, more than decades-long streak of vigorous growth." In 2021, the average yearly per capita consumption of bottled water in the United States was approximately 47 gallons (Ridder, 2022). With over 330 million inhabitants in the United States in 2021 (U.S. Census Bureau, 2021) and with each person consuming approximately 47 gallons per year, about 15.5 billion gallons of bottled water would have been consumed in the United States that year.

To keep up with increasing demand, the water bottling industry continues to withdraw increasing amounts of water (U.S. Food and Drug Administration, 2022). The U.S. Food and Drug Administration (FDA) regulates water bottling by enforcing water-quality standards and overseeing of bottling plants to ensure sanitary conditions and safe levels of contaminants (Food and Drug Administration, 2018, 2022). However, FDA regulations do not evaluate the effects of water bottling facilities on water resources (Samek, 2004).

The amount of water used for bottling constitutes a small portion of the total water withdrawn in the United States. The estimated 15.5 billion gallons per year consumed in 2021 is approximately 0.013 percent of the 322 billion gallons per day total water withdrawals in the United States in 2015, and 0.11 percent of the 39 billion gallons per day water withdrawals for public supply in 2015 (Dieter and others, 2018). The growing population, rate of industry growth, and lack of federal regulations with respect to withdrawals create a need for better understanding of the hydrological, environmental, ecological, and societal impacts of water bottling facilities and their withdrawals.

In 2021, Congress directed the U.S. Geological Survey to initiate research to better understand the hydrologic impacts of extraction of water from springs and groundwater for bottling. One initial product of this research was an inventory of bottling facilities and compilation of water withdrawals for bottling (Buchwald and others, 2023). The inventory included facilities and water use for other beverages including soft drinks, wineries, and breweries. A second part of the research was to initiate studies to investigate the effects of withdrawal for water bottling on groundwater levels, concentrations of contaminants, and groundwater salinity. Three locations were initially chosen for study: the Saco River headwaters in New Hampshire and Maine (this study), Strawberry Creek in Southern California, and the Santa Fe River in north-central Florida. A regional aquifer study of the Great Lakes Basin was added in 2022. Wards Brook valley in the Saco River headwaters was chosen for study because extractions for bottled water have the potential to affect the availability of groundwater and streamflow for the community of Fryeburg, Maine.

Information on stream depletion from groundwater withdrawals, including the amount that water bottling contributes to those withdrawals, is often lacking. Groundwater-flow models can be used to simulate changes to stream base flows with and without the various types of withdrawals to determine their effects. Groundwater-flow models can also be used to examine the potential effects of withdrawals under future climate conditions. In New England, warmer and wetter future climate conditions (U.S. Bureau of Reclamation, 2013) could have opposing effects on groundwater recharge. A soil-water-balance (SWB) model incorporates landscape processes and meteorological data to estimate and constrain the rates and spatial distribution of potential recharge (Westenbroek and others, 2018). By including hypothetical climate signals in the recharge estimation, potential future scenarios can be explored.

Purpose and Scope

The purpose of this publication is to describe the findings of a study to evaluate the effects of groundwater withdrawals on hydrologic conditions (stream base flows and groundwater levels) in Wards Brook valley. The report documents the SWB model (SWB 2.0; Westenbroek and others, 2018) used to calculate potential recharge (as net infiltration) and the groundwater-flow model (MODFLOW 6; Langevin and others, 2017) used to simulate hydrologic conditions for (1) multiple hypothetical groundwater withdrawal volumes under current recharge conditions and (2) stable groundwater withdrawals under multiple recharge scenarios. Recharge scenarios were based on projected climate conditions resulting from a stabilized greenhouse-gas emission scenario (herein referred to as the stabilized-emission scenario) and a high greenhouse-gas emission scenario (herein referred to as the high-emission scenario). The groundwater-flow model was also used to simulate hydrologic conditions resulting from groundwater withdrawals combined with reduced recharge conditions that represented a 3-year drought. Effects of groundwater withdrawals were simulated for 2016 – 2021 and for future periods spanning the mid-21st century (2046 – 2065) and late-21st century

(2080 – 2099). The models are described in appendixes of this report. Model files are published in associated data releases (Barclay and others, 2026; Holland and Barclay, 2026).

Study Area Description

The Wards Brook valley, in Fryeburg, Maine, is within the headwaters of the Saco River watershed along the border between New Hampshire and Maine in an area of steep hills and valleys (figs. 1 and 2). Two aquifers provide most of the water for use in the Wards Brook valley: the crystalline bedrock and the glacial stratified deposits (Emery & Garrett Groundwater, Inc., 2005). The crystalline bedrock aquifer is used primarily for self-supplied domestic wells and some community wells. The glacial stratified deposits form the most productive aquifer in the study area and are the source from which municipal water supplies and bottled water facilities withdraw water. They include ice-contact deposits (ie., eskers and kames), alluvial fans, deltas, and lacustrine sediments deposited in glacial lakes. In Wards Brook valley, these deposits are up to about 30 meters (m) thick (Emery & Garrett Groundwater, Inc., 2005). Glacial till constitutes a minor aquifer in terms of water use but is widespread in areal extent. The generalized surficial lithology is shown in figure 3. Additional details about the geology of the study area are provided in appendix 1.

Under natural conditions, groundwater in Wards Brook valley and nearby uplands flows downgradient and discharges to the land surface at springs, seeps, and wetlands to eventually become either flow into Wards Brook or evapotranspiration (Emery & Garrett Groundwater, Inc., 2005). However, withdrawals can lower groundwater levels and capture flows that would have naturally discharged to the land surface and contributed to stream base flow in Wards Brook. Groundwater withdrawals for bottled water and municipal supply in Wards Brook valley were of the same order of magnitude as low flows in Wards Brook. The estimated 7-day 10-year low flow (7Q10) is 4,942 cubic meters per day (m^3/d) (Dudley, 2004; U.S. Geological Survey, 2023), whereas, the withdrawals from combined public-water supply (1,066 m^3/d) and bottled water extraction (1,308 m^3/d) averaged about 2,376 m^3/d from 2016 – 2021 (Fryeburg Water Company, 2016, 2017, 2018, 2019, 2020, 2021; Luetje Geological Services LLC, 2016, 2017a,b, 2018a,b, 2019a,b, 2020; Luetje Geological Services LLC and McDonald Morrissey Associates, 2021, 2022).

Previous Investigations

The hydrogeology in the New Hampshire part of the study area has been described by Moore and Medalie (1995), Medalie and Moore (1995), Tepper and others (1990), and Johnson and others (1987). Hydrogeology in Maine is detailed in aquifer maps (Maine Geological Survey, 2023a) and surficial geology maps for the region (Davis and Holland, 1997a, b; LePage, 1997; Newton, 1997; Newton and Holland, 1997; Gosse and Thompson, 1999; Thompson and Holland, 1999; Thompson, 2014; Maine Geological Survey, 2023b).

A groundwater-flow model was used in the Wards Brook valley to determine sustainable pumping rates for the aquifer and delineate wellhead protection areas for public-supply wells (Emery & Garrett Groundwater, Inc., 2005; Emery & Garrett Groundwater Investigations, 2018). The model indicated that about 2,300 m^3/d of water could be sustainably removed from the aquifer in addition to the water required for municipal use. Model documentation from those investigations included descriptions of the local geology and hydrologic system that were used to inform development of the groundwater-flow model used for this study.

Previous studies that explored the effects of climate change on streamflows in the Saco River have associated seasonal changes to streamflows with a warmer climate. Hodgkins and others (2003) documented earlier arrival of the spring center of volume of flow in the Saco River related to earlier snowmelt. Hodgkins and Dudley (2011) found increased stream base flow in the Saco River near Conway, New Hampshire (USGS streamgage 01164500, fig. 1) that was attributed to increasing summer precipitation.

Potential future changes to recharge have been estimated for the State of New Hampshire. The hydrologic response to projected future climate conditions was evaluated by Bjerklie and Sturtevant (2018) using a Precipitation Runoff Modeling System (PRMS) model that evaluated streamflow and groundwater recharge for historical conditions (1981 – 2000) and hypothetical climate conditions for two future periods (2046 – 2065 and 2081 – 2100). The future conditions were simulated using five downscaled General Circulation Models (GCMs; U.S. Geological Survey, undated; U.S. Bureau of Reclamation, 2013) and Representative Concentration Pathways (RCPs; Stocker and others, 2013) 4.5 and 8.5 scenarios (representing stabilized- and high-emission scenarios). Mean outputs for the Saco River Basin in New Hampshire indicated increases in simulated groundwater recharge for the period 2081 – 2100, relative to the 1981 – 2000 period (7.3 percent increase for the stabilized-emission scenario and 8.3 percent increase for the high-emission scenario). The outputs also indicated higher streamflows in winter months (January through March) and lower streamflows in summer months (May through September) for the future emission scenarios relative to 1981 – 2000 conditions (Bjerklie and Sturtevant, 2018, table 5). Those outcomes were related to earlier groundwater recharge from greater rainfall in the winter and less snowfall. Those conditions would also coincide with less snowpack available later in the season to provide water for recharge and stream base flow. The outcomes may also be caused by increasing evapotranspiration that outpaces the projected increases in precipitation for the future periods (Bjerklie and Sturtevant, 2018).

Methods

A 3-dimensional groundwater-flow model for the Wards Brook valley aquifers and surrounding area (figs. 1 and 2) was used to simulate effects of groundwater withdrawals on stream base flow and groundwater levels for historical (2016 – 2021) conditions, hypothetical reduced pumping scenarios, and for stable groundwater withdrawals combined with potential recharge based on two emission scenarios (stabilized- and high-emission scenarios). The emission scenarios were each simulated with five downscaled General Circulation Models (GCMs). Future conditions were evaluated across two 20-year periods spanning the mid-21st century (2046 – 2065) and late 21st century (2080 – 2099). Groundwater withdrawals for historical and future periods were also evaluated under reduced recharge conditions that represented a 3-year drought. An overview of the historical and future hypothetical scenarios is provided in this section. Details of the model development and scenarios are in appendixes 1 and 2. The model files are published in associated data releases (Barclay and others, 2026; Holland and Barclay, 2026).

Effects of groundwater withdrawals were evaluated by comparing simulated hydrologic conditions (stream base flows and groundwater levels) from multiple pumping and recharge scenarios to a historical baseline simulation. Changes to simulated base flows were evaluated at the measurement site SG-3, near the downstream end of Wards Brook and upstream from Lovewell Pond (fig. 2). Simulated base flows were compared to a recommended minimum flow of 2,180 m^3/d (400 gallons per minute; Emery & Garrett Groundwater Investigations, 2018) for

Wards Brook that was based on flow volume measured during a “dry August.” Minimum flows needed to protect aquatic life were not evaluated when determining that threshold. The mean simulated groundwater levels of model cells within Wards Brook valley were used to evaluate changes in groundwater levels. Simulated groundwater levels in layer 1 of the groundwater-flow model were also assessed at Black Pond, Davis Pond, and Round Pond (fig. 2). Groundwater withdrawal scenarios were evaluated by comparing the simulated outputs for each month during the 2016 – 2021 period. Recharge scenarios were evaluated on a seasonal basis by comparing mean monthly simulated outputs during 2016 – 2021, 2046 – 2065, and 2080 – 2099. A detailed description of methods for developing the mean monthly data is provided in appendix 2.

Groundwater Withdrawal Scenarios

Groundwater withdrawals for the historical baseline simulation were from measured withdrawals at two wells used for municipal supply and two wells used for water bottling (Fryeburg Water Company wells 1-4; figs. 2, 4, and 5) (Fryeburg Water Company, 2016, 2017, 2018, 2019, 2020, 2021; Luetje Geological Services LLC, 2016, 2017a,b, 2018a,b, 2019a,b, 2020; Luetje Geological Services LLC and McDonald Morrissey Associates, 2021, 2022). Hypothetical pumping scenarios which represented no pumping and partial pumping (pumping only for municipal water use) were simulated for 2016 – 2021, and outputs were compared to the historical baseline simulation to understand the effects of groundwater withdrawals. This approach provided an indication of the effects on hydrologic conditions from bottled water withdrawals compared to the combined effects from municipal and bottled water pumping in Wards Brook valley. Self-supplied domestic wells and additional commercial and institutional wells are used in the study area, but the small withdrawals from those wells along with their associated return flows from onsite wastewater disposal were not simulated in the groundwater-flow model.

Recharge Scenarios

Recharge used for the historical baseline and hypothetical future emission scenarios was obtained from the SWB model (Westenbroek and others, 2018). The SWB model simulates net infiltration and therefore represents potential recharge. The SWB model required input datasets including: (1) daily meteorological data, (2) hydrologic soil groups, (3) available water capacity, (4) land cover, and (5) surface-water flow direction. Development of the SWB model followed procedures used by Nielsen and Westenbroek (2019) for an SWB model developed for the State of Maine. Daily precipitation and maximum and minimum daily air temperatures for the historical baseline simulation were obtained from Daymet Version 4 data (Thornton and others, 2022). Future meteorological data for the two hypothetical emission scenarios (stabilized- and high-emission scenarios) were from the output of five GCMs (U.S. Bureau of Reclamation, 2013). To verify that outputs from the emission scenarios represented reasonable conditions, the projected climate data were compared to historical meteorological data that were collected after the climate projections were published. Model outputs from the recharge scenarios are reported as the median of the simulation results from the five GCMs for each of the emission scenarios. Additional details about the SWB model construction and emission scenarios are provided in appendix 2.

Hypothetical scenarios were also created to represent recharge during drought conditions for the historical period (2016 – 2021) and for the mid-21st and late-21st century (2046 – 2065 and 2081 – 2099). The drought scenarios were used to illustrate hydrologic effects from a period of reduced recharge. The year with the lowest total recharge in each respective simulation period was selected to represent the drought scenario for that period, and the monthly recharge values from the selected year were applied to a 3-year simulation period.

Groundwater Recharge

Groundwater recharge rates estimated with the SWB model include (1) mean annual recharge rates based on meteorological data for historical conditions, emission scenarios, and 3-year drought; (2) monthly recharge rates for 2016 – 2021 from historical meteorological data; and (3) mean monthly recharge rates based on meteorological data for historical conditions, emission scenarios, and 3-year drought. Mean annual recharge rates from the historical baseline simulation and the emission scenarios were generally similar (fig. 6A). The mean annual recharge based on historical period meteorological data was 0.47 m. The median of the mean annual recharge rates obtained from the five GCMs for the stabilized-emission scenario for 2016 – 2021 (0.45 m) was slightly lower than historical baseline conditions, and the median of the annual recharge rates for the high-emission scenario for that period was the same as the historical baseline conditions, indicating that the median of the high-emission scenario for future periods may provide a better estimation of future recharge conditions if emissions continue along the same trajectory. Mean annual recharge rates for the mid-21st century were slightly lower for the stabilized-emission scenario and slightly higher for the high-emission scenario when compared to the historical baseline value. These differences also were apparent for the annual recharge rates in the late-21st century. Mean annual recharge representing drought conditions during the historical period was about 13 percent lower than the historical baseline simulation. The medians of the mean annual recharge rates for drought conditions in the future periods were substantially lower than the historical baseline and emission scenarios, varying from about 50 to 75 percent of the mean annual recharge used in the historical baseline simulation (fig. 6B).

Estimated recharge in Wards Brook valley is largest in the non-growing season, primarily during March, April, October, and November, with limited potential recharge during the summer months (fig. 7A). Several differences were observed when mean monthly potential recharge results from the historical baseline simulation were compared with results from the emission scenarios for 2016 – 2021 to verify that the projected climate data had reasonably represented that period. The historical baseline simulation had higher recharge in April, May, and October and lower recharge in January, March, and December than recharge based on climate data from the emission scenarios. In general, the magnitude of these differences was still relatively small (within 0.05 m per month), and the emission scenarios were generally able to replicate the historical baseline conditions (fig. 7A).

The differences in mean monthly recharge between the historical baseline simulation and the future emission scenarios include larger recharge in December, January, and February in the mid-21st century and late-21st century periods and lower recharge for the months of April, May, and October from the emission scenarios relative to the historical baseline conditions (fig. 7A). However, the differences for October were about the same as they were for the 2016 – 2021 validation period; therefore, projected future recharge conditions for that month may be less reliable. The range of values from the five GCMs, shown as shading in figure 7A, was greater in the winter months than in the summer months, but the ranges often overlapped. Simulated mean monthly recharge rates were highly variable among the drought scenarios in the winter months, and predicted recharge values were close to zero from May through September (fig. 7B).

Effects of Groundwater Withdrawals

Reduced Groundwater Withdrawals

Effects of groundwater withdrawals on simulated stream base flows in Wards Brook were evaluated using measured groundwater pumping data for 2016 – 2021, a non-pumping scenario, and a partial pumping scenario where there was only pumping for municipal supply (no pumping for bottled water extraction) (fig. 8). The simulated stream base flows for each month at site SG-3 (fig. 2) during 2016 – 2021 under current pumping rates ranged from about 3,250 m^3/d in August 2016 to 15,840 m^3/d in April 2019 (fig. 8). Under a non-pumping scenario, the simulated base flows ranged from about 6,350 m^3/d in September 2016 to 17,680 m^3/d in April 2019. The differences in flow highlight the effects of all the groundwater withdrawals which are likely to reduce the flow in Wards Brook substantially under base-flow conditions. A third scenario simulating the base flows of Wards Brook under pumping conditions without bottled water extraction produced simulated base flows that were about halfway between the two conditions described above. The lowest simulated flows under the pumping conditions (3,250 m^3/d) approached the recommended minimum flows of 2,180 m^3/d (400 gallons per minute; Emery & Garrett Groundwater Investigations, 2018) in August 2016, which was during a drought period. The simulated base flows represent the mean base flow for each month, so there could be substantial daily variation within the month. In general, about half of the total pumping impact on the base flows in Wards Brook is from the bottled water extraction (fig. 8).

Simulated groundwater elevations at Black Pond, Davis Pond, and Round Pond (fig. 2) were also compared under the pumping conditions described above (fig. 9). The simulated groundwater levels at Round Pond and Davis Pond showed a slight difference between the full pumping and the non-pumping scenario. However, differences between the pumping scenarios were much smaller than seasonal variability. In the partial pumping simulation with no withdrawals for bottled water, the groundwater elevations were about the same as in the full pumping simulation. Most bottled water pumping takes place at a distance from these ponds, whereas withdrawals for the Town of Fryeburg supply well FWC3 (fig. 2) are much closer to the ponds. The simulations show that most of the groundwater-level fluctuations at these ponds were primarily caused by differences in recharge rather than from groundwater withdrawals. There was no noticeable difference in simulated groundwater levels at Black Pond under the different pumping conditions because the till-covered upland likely prevents a strong hydraulic connection between the pumping well locations and that pond. These results should be used with consideration that the groundwater-flow model was primarily designed to assess stream base flow, and understanding the effects of pumping on pond levels would require a more targeted analysis of the hydrologic conditions at the pond locations.

Hypothetical Future Recharge Conditions

Effects of groundwater withdrawals on seasonal hydrologic conditions under variable recharge conditions were evaluated by comparing results from the historical baseline simulation that used recharge estimated from measured meteorological data for 2016 – 2021 with results from future recharge scenarios for mid- and late-21st century that were estimated using projected meteorological data from hypothetical climate conditions. The hydrologic conditions compared were stream base flows for each month in Wards Brook (fig. 10A), mean simulated groundwater levels across Wards Brook valley (fig. 11A), and differences in groundwater levels at selected ponds (fig. 12A). Drought simulations were similarly compared (figs. 10B, 11B, and 12B).

In general, the simulated base flows for the historical and future conditions followed the same seasonal pattern with higher base flows in winter and spring and lower base flows in summer and fall (fig. 10A). The lowest mean monthly base flows from the future emission scenarios all remain larger than the minimum recommended streamflow (2,180 m^3/d) and indicate no long-term declines in flow relative to historical conditions. The base flows from the historical baseline simulation were generally higher than the base flows simulated for 2016 – 2021 that were based on data from the emission scenarios, though the historical baseline values usually were within the range of values from the five GCMs. Future simulated stream base flows from the emission scenarios were lower from April to November and higher in December through March relative to the baseline simulation, indicating changes in future seasonal patterns of recharge. The changes in mean monthly base flows simulated in the future emission scenarios likely represent a shift to a longer growing season and a shorter winter season, possibly with reduced snowfall and increased rainfall. Under those conditions, groundwater storage would begin to decline earlier in the year and cause reduced summer base flows.

The differences noted in simulated stream base flows were also observed in plots of the mean simulated groundwater levels of model layer 1 in Wards Brook valley under the two emission scenarios (fig. 11A). The seasonal variability in groundwater levels is about 1.4 m for the historical baseline period and did not change substantially for the future emission scenario conditions. The simulated groundwater levels at Round Pond in the high-emission scenario for both the mid-21st and late-21st century were somewhat higher than in the historical baseline simulation from January through April and lower than the baseline in September through November (fig. 12A). For the stabilized-emission scenario, simulated groundwater levels were generally lower from April through December for the mid-21st century and lower in October to December in the late-21st century.

Analysis of the drought scenarios showed potential reductions to stream base flows and groundwater levels (figs. 10B, 11B, and 12B). Drought conditions applied to future emission scenarios caused base flows to approach or fall below minimum recommended streamflow (2,180 m^3/d, Emery & Garrett Groundwater Investigations, 2018) during the summer months (fig. 10B). In the mid-21st century, simulated base flows were below the recommended streamflow in August and September. In the stabilized-emission scenario, base flow was 1,972 m^3/d in August and 2,078 m^3/d in September. Base flows in the high-emission scenario were slightly larger, with 2,101 m^3/d in August and 2,144 m^3/d in September. In the late-21st century, base flows approached the recommended streamflow. The lowest base flow in the stabilized-emission scenario was 2,229 m^3/d (September), and the lowest base flow in the high-emission scenario was 2,903 m^3/d (September). Although these drought scenarios were hypothetical, they indicate that under extended (multi-year) periods of low groundwater recharge, base flows could remain low for extended periods in the summer months, potentially requiring reductions in groundwater withdrawals to maintain minimum recommended streamflow.

The mean groundwater levels in layer 1 of the groundwater-flow model under simulated drought (fig. 11B) showed a similar pattern to base flows over time, with the stabilized-emission scenario producing the lowest groundwater levels in both the mid-21st century and late-21st century. Lower groundwater levels in the stabilized-emission scenario compared to the high-emission scenario were likely related to differences in precipitation between the two scenarios (appendix 2, fig. 2-7). The simulations indicate that under drought conditions, groundwater levels could decline by approximately 1 m overall. Simulated groundwater levels at Round Pond under the drought scenarios had a similar pattern (fig. 12B), indicating that the groundwater levels at this pond could drop substantially and shrink the pond size under a drought of magnitude simulated in the mid-21st and late-21st century. Hydrologic changes under the simulated scenarios may also have effects on water levels within wetlands, and these effects may be exacerbated in areas closest to groundwater withdrawals. However, the model is not sufficient for detailed analysis of ponds or wetlands.

Summary

To keep up with demand, the water bottling industry continues to withdraw increasing amounts of water from multiple freshwater sources. Effects of groundwater withdrawals for bottled water and municipal supply were assessed for multiple hypothetical groundwater withdrawal and recharge scenarios within Wards Brook valley in Maine and New Hampshire. The study was part of a national effort to better understand the impact of extracting water for bottling on hydrologic conditions and water availability.

A groundwater-flow model was used to simulate stream base flow and groundwater levels for historical (2016 – 2021) conditions, hypothetical reduced pumping scenarios, and future recharge conditions based on the output of five downscaled General Circulation Models for two greenhouse-gas emission scenarios (stabilized- and high-emission scenarios). Historical and future periods were also evaluated using reduced recharge conditions that represented a 3-year drought for each period. Effects of groundwater withdrawals under those scenarios were evaluated by comparing simulated hydrologic conditions (stream base flows and groundwater levels) from the different pumping and recharge scenarios with a historical baseline simulation that represented conditions for 2016 – 2021.

Analyses showed that the simulated base flows in Wards Brook were reduced because of pumping for both municipal water supplies and for water bottling, and about half of the total pumping effect on the base flows in Wards Brook was from bottled water extraction. Simulated base flows approached the minimum recommended streamflow of 2,180 cubic meters per day (400 gallons per minute) for the month of August 2016. Simulated groundwater levels at two of three nearby ponds (Round Pond and Davis Pond) were minimally affected by pumping conditions, and effects were primarily from the municipal well closest to the ponds.

Several scenarios of future projected recharge were used to understand the potential effects of groundwater withdrawals on hydrologic conditions under multiple hypothetical climate scenarios. Annual projected recharge rates in the mid- and late-21st century from the stabilized and high-emission scenarios were similar to rates for 2016 – 2021. However, monthly recharge patterns for the future periods shifted toward more recharge in the winter months (December, January, and February) and less recharge in April, May, and October in relation to 2016 – 2021.

The lowest mean monthly base flows from the future emission scenarios all remain larger than the minimum recommended streamflow and indicate no long-term declines in flow relative to historical conditions. Compared to historical conditions, simulated stream base flows during the future periods were lower from April through November and higher in December through March. The seasonal variability in the simulated groundwater levels was about 1.4 meters and does not change substantially for the future periods. The simulated groundwater levels at Round Pond in the high-emission scenario for both the mid- and late-21st century were higher than in the historical period from January through April but were lower in September through November. For the stabilized-emission scenario, simulated groundwater levels at ponds were generally lower from April through December during the mid-century period and lower in October to December in the late-century period.

Simulated base flows during hypothetical 3-year drought scenarios declined below minimum recommended streamflow during the summer months in the stabilized- and high-emission scenarios in the mid-21st century. Future climate conditions similar to these drought scenarios might require reduced pumping in Wards Brook valley to maintain recommended streamflow. The simulations demonstrate that under drought conditions, groundwater levels could decline by about 1 meter overall.

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Appendix 1. Groundwater-Flow Model Development

The groundwater-flow model for this investigation was constructed using MODFLOW 6 with the Newton-Raphson solver (Langevin and others, 2017). The groundwater-flow model input and output files are published in an associated data release (Barclay and others, 2026). The model uses a finite-difference grid with 5 layers of 590 rows and 524 columns that are 30.48 by 30.48 meters (m)(100 by 100 feet). The active area in this model grid covers about 98 square kilometers (fig. 1-1). In areas underlain by glacial stratified deposits, the top two layers represent surficial sediments, and the bottom three layers represent the underlying crystalline bedrock. The top two layers have thicknesses interpreted from global gridded depth to bedrock data with a 250-m resolution (Shangguan and others, 2017) and observations of depth to bedrock at selected monitoring wells (Emery & Garrett Groundwater, Inc., 2005). The maximum thickness of the glacial stratified deposits was 32.4 m in the model. In areas underlying glacial till or surface bedrock, the top two layers of the model were simulated as a composite of till-covered bedrock, with thickness ranging from 5.3 m to 32.5 m, and the bottom three layers represent crystalline bedrock. In all areas, each of the three crystalline bedrock layers have a uniform thickness of 30 m.

The simulation included a steady-state stress period with initial groundwater levels set to the land surface that was defined by a 1-m digital elevation model (U.S. Geological Survey, 2024). The steady-state period was followed by a transient period consisting of two sets of 72 monthly stress periods from 2016 – 2021, the period with data available for calibration and verification. Starting groundwater levels for the transient period were from the steady-state period. The second set of 72 stress periods was included to allow the model to adjust to values more typical of the early data (January to March 2016). This was done because the change from a steady-state simulation with average conditions to January 2016 conditions was large for the early months of the transient simulation. The second set of stress periods allowed the transition to January 2016 to be more representative of January conditions, and groundwater levels matched more closely. Outputs from the final 72 stress periods were extracted as the final model outputs to represent historical baseline conditions.

The effects on hydrologic conditions from groundwater withdrawals under future hypothetical climate and drought conditions were evaluated seasonally. To achieve this, a dynamic equilibrium model was developed using the mean monthly recharge and mean monthly groundwater withdrawals from 2016 – 2021 to simulate average conditions for each month within that time period. This dynamic equilibrium model was the historical baseline simulation used for evaluating the seasonal effects of groundwater withdrawals by comparing its outputs with those that were obtained using hypothetical scenarios of recharge from the soil-water-balance (SWB) model that were based on the projected greenhouse-gas emissions described in appendix 2.

Boundary Conditions

Natural hydrologic boundaries were used as boundaries in the groundwater-flow model (fig. 1-1). The lateral boundary of the groundwater-flow system includes basin drainage divides in areas of glacial till or bedrock and the Saco River, which defines some of the model boundary with a meandering course. The lateral boundaries of the model in the uplands and on drainage basin boundaries (southeast and southwest edges of the model) were considered no-flow boundaries. The upper boundary of the model was land surface based on a 1-m digital elevation model (U.S. Geological Survey, 2024), which received spatially variable recharge from precipitation as estimated by the SWB model described in appendix 2. The bottom of the model (layer 5) was set as a no-flow boundary.

The Saco River was considered a head-dependent boundary simulated in this model using the Streamflow Routing (SFR1) package, which is described in more detail below. The other rivers or streams within the active model area also were simulated with the SFR1 package. In addition to the streams, five wetland areas were simulated as head-dependent boundaries using the Drain (DRN) package. Simulated drains were used in areas where there was likely surface flow from wetlands, but these areas were not connected to the stream network in the dataset. The drains allowed water to leave the model and simulate a reasonable water-table elevation. Additional information on simulated wetland drains is provided in more detail below.

Aquifer Properties

The modeled area consists of three primary geologic settings: glacial stratified deposits, Saco River floodplain, and uplands. The glacial stratified deposits are dominated by a series of sediments deposited by glacial Lake Pigwacket (Thompson, 2014). One of the important deposits in the model area includes the ice-contact deposits known as the Oak Hill stage deposits of Lake Pigwacket. As the head of outwash, this unit contains coarse-grained materials that, based on interpretation of well logs and test holes in this area, continue for some distance to the west in the subsurface and are likely the primary deposits from which water for municipal use and bottling is extracted. Overlying these ice-contact deposits and much of the immediate area around the major pumping wells are a series of finer deposits which are the Fryeburg stage deposits. These deposits are likely deltaic and get finer with depth. Overlying this unit in much of the area are eolian deposits, which are present as dunes or a mantle over the glacial till or stratified deposits where they conceal the contact between these formations. In the southern part of the model area, there are additional deltaic sand and gravel deposits of Glacial Lake Marston (Davis and Holland, 1997a). These deposits envelop some earlier esker deposits in the southern part of the study that were formed when glacial ice covered the area.

The stream alluvium of the modern Saco River floodplain is present primarily where the glacial deposits have been incised by the post glaciation meandering of the river. The texture of these deposits varies depending on the local deposition characteristics of different areas along the Saco River.

The uplands are composed of a layer of glacial till that overlies bedrock. Areas of glacial till in the modeled area include those with the units till, hummocky moraine, or ribbed moraine as described by Thompson (1999). The underlying bedrock, which may be exposed in places, particularly on some of the hilltops, is primarily a muscovite granite of Carboniferous age from the Sebago Batholith (Osberg and others, 1985; Lyons and others, 1997).

The simulated aquifer properties, particularly for horizontal hydraulic conductivity, were classified using lithology information from Maine surficial geology maps for the Fryeburg and

Brownfield quadrangles (Davis and Holland, 1997a,b; Thompson, 2014). Additional detailed interpretations in Emery & Garrett Groundwater, Inc. (2005) were used to help understand and interpret the geology for assigning hydraulic properties to aquifer materials. The following general assignments of texture, grain size, or horizontal hydraulic conductivity were made (figs. 1-2, 1-3). The Oak Hill stage deposits of glacial Lake Pigwacket and the Glacial Lake Marston and associated ice-contact deposits were assigned as coarse-grained deposits; these deposits were simulated as the same horizontal hydraulic conductivity in layers 1 and 2 (50 meters per day [m/d]). The Saco River floodplain alluvium was assigned as medium grained, indicating that the horizontal hydraulic conductivity of these deposits (35 m/d) is generally less than the coarse ice-contact and deltaic deposits. In layer 2, underlying this alluvium, it was assumed that the deposits were very fine-grained glaciolacustrine deposits and assigned a low horizontal hydraulic conductivity (0.5 m/d in layer 2). The upper Fryeburg stage deposits were assumed to be finer textured on average than the alluvial deposits in layer 1. In layer 2, these deposits were assumed to be the same texture as the deposits in layer 2 underlying the Saco River floodplain alluvial deposits. It was assumed that eolian deposits, where present, were primarily above the water table and likely do not substantially influence groundwater flow. However, they can be permeable and had an influence on recharge estimates because soil type was a factor in the recharge determinations from the SWB model.

In areas with glacial till and bedrock, the layers were considered to become less permeable with depth. It was assumed that for most places in the uplands, layer 1 had the highest horizontal hydraulic conductivity (0.6 m/d). Layer 2 was assumed to be less permeable than layer 1 (0.3 m/d). The top two layers were at the depths where much of the groundwater flow was likely to occur. The bottom three layers of bedrock were set with a low hydraulic conductivity

(0.025 m/d), indicating limited groundwater flow through these layers. There was limited information on the lithology beneath Lovewell Pond. According to Emery & Garrett Groundwater, Inc. (2005), Lovewell Pond was the location of a stagnant ice block during deglaciation. It was assumed that the material beneath Lovewell Pond was either glacial till or bedrock, and that relative horizontal hydraulic conductivity was assigned to be the same as layer 2 in other areas with till and bedrock (0.3 m/d). A summary of the final values used is given below in the model calibration section because these values were adjusted as part of the calibration process. In the groundwater simulations, it was assumed that there was no horizontal anisotropy, and the ratio of horizontal to vertical hydraulic conductivity was set to 10:1 for glacial stratified deposits, whereas for the till and/or bedrock it was set to 1:1.

The transient simulations required values of specific yield (Sy) and specific storage (SS). For the stratified glacial drift deposits, Sy was assumed to be 0.2 (unitless) and the SS was assumed to be 0.0002 meter^{-1} (m^{-1}). For areas of glacial till and bedrock, the Sy was 0.1 and the SS was 1×10^{-5} m^{-1} in layer 1. In layers 2 through 5, they were set to 0.05 and 1×10^{-5} m^{-1} respectively. The values selected were within the ranges reported in the literature (Morris and Johnson, 1967; Domenico and Schwartz, 1997; Lyford and others, 2007; Starn and Brown, 2007; Masterson and Granato, 2013).

Internal Sources and Sinks of Water

Streamflow was simulated using the Streamflow Routing (SFR1) package in MODFLOW 6 (Langevin and others, 2017) to allow routing of water downstream. This allowed for determination of streamflow at the downstream end of Wards Brook. Streamflow in this model was considered base flow and does not include surface runoff. The following parameters were used in the streamflow package and were generalized. Elevation of the streambed was based on maximum and minimum elevation attribute values from the value-added attribute tables (VAA) associated with each stream segment found in the high resolution NHDPlus flow line dataset (Moore and others, 2019). A linear interpolation using those values resulted in streambed elevation decreasing in the downstream direction along each stream segment. The elevation of the bottom of layer 1 was compared with the streambed elevations, and the bottom of layer 1 was reduced in one cell where it was higher than the streambed. The stream widths were determined by upstream drainage areas and ranged from 1 to 140 m. In areas with large ponds, wider stream widths were used (up to 1,200 m) but were limited such that the streambed area (streambed width multiplied by the stream length in the model grid cell) did not exceed the model grid cell area of 929.03 square meters (m^2). The vertical hydraulic conductivity was assigned as 1 m/d to all stream reaches, and the bed thickness was assumed to be 0.46 m (about 1.5 feet). Streamflow was routed through ponds in the study area, except for kettle ponds with no surface inflow or outflow. All pond areas were simulated as high hydraulic conductivity zones to allow for adjustment of the water surface. Flow input to the furthest upstream SFR cell was determined using a monthly base-flow separation of streamflow data at USGS streamgage 01064500 (fig. 1 in main body of preprint), using the streamflow partitioning algorithm, PART (Rutledge, 1998). In the dynamic equilibrium simulations of groundwater flow, this input flow to the Saco River was estimated using the monthly recharge from the SWB models for the basin upstream of the Saco River near Conway streamgage (USGS streamgage 01064500). The inputs to this reach were likely not relevant to the water budget of the area in Wards Brook valley, but it was determined that the water table varied in the immediate area of the Saco River. Therefore, a reasonable approximation of seasonal base flow was required.

Drains were used in several areas (fig. 1-1) to route some water out of wetland areas that were not connected to the stream network and at the location where the Saco River leaves the modeled area at the downstream end of the model. The drain located at the downstream end of the Saco River allowed water that would have been underflow to leave the model and not to build up in the simulation and cause an excessively high water table. In the areas simulated as drains, the conductivity of the drain was set to a large value (1,000 m/d) so that water could move freely through the drain and out of the model. Drain elevations were determined by the mean altitude of each cell from elevation data found in the NHDPlus dataset (Moore and others, 2019). Drain elevations remained unchanged through all simulations.

Water withdrawals were **s**imulated at four large-capacity wells that pump water from the glacial stratified deposits (figs. 4, 5 in main body of preprint). Monthly withdrawal data were from a series of reports from the Town of Fryeburg Maine for the period of interest (Fryeburg Water Company, 2016, 2017, 2018, 2019, 2020, 2021; Luetje Geological Services LLC, 2016, 2017a,b, 2018a,b, 2019a,b, 2020; Luetje Geological Services LLC and McDonald Morrissey Associates, 2021, 2022). These monthly data (fig. 4) were used in the transient simulation of conditions from 2016 – 2021. The mean withdrawals for each month (mean monthly) for that period (fig. 5) also were used as fixed values in the dynamic equilibrium simulations for the 2016 – 2021 period and for the simulations of each of the ten General Circulation Models (GCM) and emission scenario combinations used for future periods. The simulated wells draw water from layer 2 at each well location (fig. 2 in main body of preprint). The well designations are Fryeburg Water Company (FWC) 1 – 4, where wells 1 and 4 have been used for bottled water extraction and wells 2 and 3 have been used for municipal supplies. During the period of study (2016 – 2021), the FWC4 well was only used during one month, although it has been used more extensively in the past (Emery & Garrett Groundwater Investigations, 2021). Withdrawals and associated return flows from septic systems were not included for private domestic wells or for small wells used for commercial purposes.

Recharge used for the historical baseline simulation and hypothetical future emission scenarios was obtained from the SWB model (Westenbroek and others, 2018) described in appendix 2. Monthly data from the SWB model were used in the transient simulation of conditions from 2016 – 2021. Recharge used for the future hypothetical climate and drought conditions is described in the Simulating Hydrologic Conditions for Future Projected Recharge section of this appendix. The SWB model simulates net infiltration and therefore represents potential recharge. Cells designated “Open Water” in the National Land Cover Database (NLCD) 2019 input land cover layer (Dewitz and U.S. Geological Survey, 2021) were excluded from the SWB calculations because SWB does not have a mechanism to simulate recharge in ponded areas. Instead, “pond recharge” was calculated separately for the groundwater-flow model to fill in missing values in the “Open Water” cells. Pond recharge was specified to be the difference between monthly domain-averaged precipitation for the corresponding period and an estimated monthly rate of free water surface evaporation (2.54 centimeters per month from October to April and 9.40 centimeters per month from May to September; Farnsworth and others, 1982, map 2 and 3). This pond recharge filling approach was adapted from Barlow and Dickerman (2001). In the steady-state periods of each simulation, recharge was estimated as the mean recharge across all days and years in the period.

Model Calibration

Although limited data were available for calibration, groundwater levels in Wards Brook valley were compared to observed groundwater level data from 21 wells to assess model results. The model was calibrated initially by comparing simulated groundwater levels to mean observations of groundwater levels for 2016 – 2021 in the steady-state stress period. The model was further refined by comparing individual monthly groundwater-level measurements to groundwater levels simulated in the transient model. Data were provided by Daniel Tinkham (Emery & Garrett Groundwater Investigations, written commun., August 30, 2022), Fryeburg Water Company (2016, 2017, 2018, 2019, 2020, 2021), Luetje Geological Services LLC (2016, 2017a,b, 2018a,b, 2019a,b, 2020), and Luetje Geological Services LLC and McDonald Morrissey Associates (2021, 2022).

The calibration process was primarily a manual trial-and-error process which involved adjusting the horizontal and vertical hydraulic conductivities of the units described in the Aquifer Properties section. The hydraulic conductivity values were adjusted for areas with the same general material classifications. The initial changes to hydraulic conductivity values were for areas in the till-covered uplands to ensure that groundwater levels for the steady state stress period were similar to the elevations of perennial streams in these areas. Next, values were adjusted in the other units until the smallest overall groundwater level residuals were observed within the glacial stratified drift deposits. Hydraulic conductivity values used in final transient and dynamic equilibrium simulations are in table 1-1.

Table 1-1. Hydraulic conductivity values used in final transient and dynamic equilibrium simulations. General lithology is also shown on figures 1-2 and 1-3.

General lithology	Model layer	Horizontal hydraulic conductivity (meter/day)	Vertical hydraulic conductivity (meter/day)
Till and bedrock	1	0.6	0.6
Till and bedrock	2	0.3	0.3
Bedrock	3, 4, 5	0.025	0.025
Swamp deposits	1	6.1	0.61
Swamp deposits	2	4.6	0.46
Fines	1	7	0.7
Fines	2	0.5	0.05
Saco River floodplain alluvium	1	35	3.5
Coarse grained	1,2	50	5

Groundwater-level observations were compared to simulated groundwater levels (heads) in stress periods 73 to 145 (figs. 1-4 and 1-5; table 1-2), which represented a second simulation of all the stress periods from January 2016 – December 2021. The second set of stress periods was needed because the change in recharge from steady-state conditions to January 2016 conditions was abrupt, and early-time simulation values did not match observations as well in stress periods representing the early months. The second set of stress periods allowed the transition to January 2016 to be more representative of January conditions, and groundwater levels matched more closely. During the final 72 stress periods, the mean discrepancy in the simulated values for 13 of the 21 wells used in the calibration was less than 0.5 m, and the largest mean discrepancy overall was just over 2 m (table 1-2). Groundwater levels were also compared to stream elevations and land surface to qualitatively ensure that water levels were not below streambeds in areas with perennial streams.

Table 1-2. Mean water-level residuals and root-mean squared error of residuals at individual monitoring wells, Wards Brook model area, 2016 – 2021. Residuals calculated as simulated minus observed groundwater level. [*well depth unknown and layer assumed]

Location identifier on figure 1-4	Site name	Model layer	Mean difference, in meters	Root mean squared error
1	TW-2-03	2	-0.06	0.45
2	TW-09-03	2	0.29	0.51
3	MW-101	1	1.44	1.50
4	MW-103	2	0.05	0.26
5	MW-105	2	1.36	1.37
6	MW-107	1	-0.77	0.94
7	MW-108	2	-0.45	0.54
8	MW-109	2	-0.73	0.77
9	MW-110	2	0.60	0.76
10	MW-113	2	-0.19	0.55
11	MW-114	2	-1.45	1.48
12	FWC-MW1	1*	-2.01	2.07
13	FWC-MW2	1*	0.02	0.29
14	FWC-MW3	1*	0.04	0.52
15	FWC-MW4	1*	0.56	0.81
16	Round Pond SG-1	1	0.08	0.67
17	Rainmaker MW-1	2	-0.01	0.27
18	Rainmaker MW-2	2	-0.01	0.27
19	Rainmaker MW-3	2	-0.01	0.27
20	Rainmaker MW-4	2	-0.04	0.26
21	Rainmaker MW-5	2*	0.04	0.24

Simulated base flows were also compared with estimated base flows in Wards Brook to confirm that the volume of water leaving Wards Brook valley was within a reasonable range. Streamflow data were collected in Wards Brook at site SG-3 (fig. 1-1) on a monthly basis from June 2020 – December 2021 (Luetje Geological Services LLC and McDonald Morrissey Associates, 2021, 2022). A formal base-flow analysis requires continuous data and therefore could not be performed on the available streamflow data for Wards Brook. Instead, roughly estimated base flows were obtained from observed streamflow using the daily calculated baseflow fraction from three nearby streamgages: (1) Saco River at River Street, at Bartlett, NH (USGS streamgage 010642505; U.S. Geological Survey, 2022), (2) Saco River near Conway, NH (USGS streamgage 01064500), and (3) Bearcamp River at South Tamworth, NH (USGS streamgage 01064801). The baseflow fractions at the streamgages were calculated using the PART (Rutledge, 1998), Base-Flow Index (BFI) (Gustard and others, 1992), and HYSEP (Sloto and Crouse, 1996) algorithms. The mean fraction for each day across all three streamgages and algorithms was multiplied by the measured streamflow to estimate baseflow. This comparison may be limited because base flow in Wards Brook may not be adequately represented using base-flow separation data from the nearby streamgages. In addition, base-flow estimates represented a point in time, and the model outputs represented the mean base flow for the entire month. However, monthly simulated base flows were usually within 25 percent of the roughly estimated base flows for Wards Brook, providing confidence that the simulated volume of water discharging to Wards Brook was within a reasonable range.

Simulating Hydrologic Conditions for Future Recharge Scenarios

Effects of groundwater withdrawals on hydrologic conditions (stream base flows and groundwater levels) were assessed under several scenarios of future projected recharge. Projected recharge for two emission scenarios that were simulated with five GCMs (table 2-2) was obtained from the SWB model as described in appendix 2. Net infiltration obtained from each of the 10 SWB simulations was extracted for the Wards Brook groundwater-flow model boundary and averaged for each month across the 6-year historical period (2016 – 2021) and for two future twenty-year periods (2046 – 2065 and 2080 – 2099) and then passed to the groundwater-flow model as a set of 30 input files representing mean monthly recharge for each unique combination of emission scenario, GCM model, and period.

A set of dynamic equilibrium models were developed using the groundwater-flow model and the recharge estimates obtained from the SWB model to simulate the hydrologic system on a monthly basis because seasonal changes are important when evaluating hydrologic conditions for this area. The concept of the dynamic equilibrium model is to capture the central tendency of the projections without bias toward a particular year or reliance on outputs from individual years. A dynamic equilibrium model with monthly stress periods was first created as a historical baseline simulation for comparison with future projected climate scenarios. The historical baseline simulation averaged the recharge calculated by SWB using meteorological data from Daymet Version 4 (Thornton and others, 2022) and measured groundwater withdrawals across the 2016 – 2021 period for each month to obtain mean monthly model inputs (fig. 6 in main body of preprint). The historical baseline simulation was run using 97 stress periods, which included a steady-state stress period, followed by 8 sets of 12 monthly periods, to allow the models to come to a dynamic equilibrium for the months simulated. The initial groundwater levels for the steady-state stress period were set to land surface. Simulated groundwater levels from the steady-state stress period were then used as starting conditions for the subsequent periods. The final 12 stress periods were extracted from the model results to be used to represent the simulated effects of groundwater withdrawals for the monthly historical baseline conditions from 2016 to 2021.

A dynamic equilibrium model was then created for each of the 30 recharge files that were created by the SWB model using the climate projection data from the emission scenarios. Monthly pumping rates for the models were maintained at the mean monthly 2016 – 2021 rates (fig. 5). The simulated hydrologic conditions (stream base flows and groundwater levels) from those models were summarized such that the medians of the five GCMs used for each emission scenario and period were used for analyses.

Hypothetical drought scenarios were created for the historical period and for the two future periods. To determine plausible drought scenarios, the year with the lowest annual recharge from each SWB model in each period was used in a drought simulation that lasted for a 3-year period. For the future periods, lowest annual recharge was selected for each of the 10 simulations (two emission scenarios and five GCMs) for each 20-year period. The 3-year period was added as a series of 36 additional monthly stress periods that were appended to each dynamic equilibrium model. The final 12 stress periods were analyzed to determine the simulated effects of these drought conditions on hydrologic conditions.

Groundwater-Flow Model Assumptions and Limitations

All hydrologic models are a simplification of the hydrologic system and therefore include assumptions and limitations. Some primary assumptions important to this study include:

1. All potential recharge obtained from the SWB model instantly reaches the water table as groundwater recharge.
2. The model boundaries used for the simulations coincide with groundwater system flow boundaries.
3. Pumping rates for the future periods would be maintained at the same monthly volumes as the 2016 – 2021 period.

Limitations of the groundwater-flow model include:

1. Simulated streamflows in MODFLOW 6 represent only the base flow, or the component of streamflow from groundwater discharge, and cannot be used to predict depletions to total streamflow.
2. Reliable measurements of stream base flow were not available as volumetric calibration targets. The only streamflow measurements collected in Wards Brook were monthly measurements during 2020 and 2021 and could not be used to determine the portion of streamflow that was from base flow. Instead, approximate estimates of stream base flow in Wards Brook were used to confirm that simulated flows were within reason.
3. Small withdrawals for self-supplied domestic, commercial, and institutional uses along with their associated return flows from areas with onsite wastewater disposal were not simulated in the groundwater-flow model.
4. Most of the water-level calibration targets (17 of 21) were located in the Wards Brook valley in areas with fine or coarse-grained materials in model layers 1 and 2 (figs. 1-3 and 1-5). Therefore, calibration results are better able to simulate results in those areas than in upland areas underlain by till and bedrock or outside the Wards Brook valley.
5. The model cannot simulate localized groundwater level or stream base flows near pumping wells but reasonably simulated the overall water budget.
6. The model is not sufficient for detailed analysis of pond levels, even if simulated groundwater levels at Round Pond were simulated relatively accurately.

7. Aquifer properties were assumed to be spatially uniform within geologic units. Localized variation of aquifer properties within the geologic units likely has little effect on the hydrologic conditions at the scale of Wards Brook valley but may affect simulated hydrologic conditions at particular locations.

8. Although simulated groundwater levels were in good agreement with measured values within the Wards Brook valley (mostly within 0.5 m), simulated groundwater levels were less reliable in areas outside of the valley and were above land surface in some areas.

Despite these potential limitations, calibration results provided evidence that this preliminary groundwater-flow model could provide reasonable estimates of groundwater levels and volume of water resources within Wards Brook valley for purposes of this study.

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Appendix 2. Soil-Water-Balance Model Development and Estimated Potential Recharge

Groundwater recharge inputs for the groundwater-flow model were obtained from a soil-water-balance version 2.0 (SWB) (Westenbroek and others, 2018) model built for the Saco River headwaters. The SWB model simulates “net infiltration”, or potential recharge, which represents water in the soil column that infiltrates below the plant root zone. For the purposes of the groundwater-flow model, simulated potential recharge was assumed to represent groundwater recharge. The SWB model used static gridded data layers and daily meteorological data to produce spatially variable estimates of groundwater recharge for the Saco River headwaters drainage basin upgradient of the Saco River at Cornish, Maine streamgage (U.S. Geological Survey [USGS] streamgage 01106600, fig. 1 main body of report), which has a drainage area of about 3,351 square kilometers (km^2). Recharge estimates were developed for a grid with cells that were 122 by 122 meters (400 feet), resulting in 647 rows, 557 columns, and 225,557 active model cells. The model was used to produce a historical simulation (2000 – 2021) and a series of future simulations that were used to calculate future projected recharge based on hypothetical climate conditions for 2016 – 2099. Development of the SWB model followed procedures in Nielsen and Westenbroek (2019). Data sets used in the development of the SWB model along with model outputs are available in a USGS data release by Holland and Barclay (2026).

The Saco River headwaters area surrounding Wards Brook valley was chosen for the SWB model study area so that simulated results could be compared with previously published recharge models (Bjerklie and Sturtevant, 2018; Nielsen and Westenbroek, 2019). Modeling the Saco River headwaters also allowed for estimating the amount of base flow that enters the Saco River along the upgradient side of the groundwater-flow model (refer to the Internal Sources and Sinks of Water section of appendix 1 for details).

The elevation of the Saco River headwaters ranges from about 80 meters (m) where the Saco River leaves the study area to 1,917 m above North American Vertical Datum of 1988 (NAVD 88) on the summit of Mount Washington, the highest point in the northeastern U.S. The White Mountain National Forest covers 1,162 km^2, or about 35 percent of the study area. The population of the Saco River headwaters area at the time of the 2020 Census was about 45,790 (Esri, 2023). Annual precipitation at the Fryeburg Eastern Slope Regional Airport in the Saco River valley averaged 118 centimeters/year (cm/yr) for the period 2000 – 2021 (Menne and others, 2012). However, the highest annual precipitation (about 203 cm/yr) was on the high elevation areas surrounding Mount Washington (Thornton and others, 2022). The mean annual discharge exiting the study area was estimated as 79 centimeters (cm) for water years 2000 – 2021, based on an analysis of the daily mean streamflow data at USGS streamgage 01066000 (fig. 1). Base flow in the Saco River headwaters represents about 60 to 82 percent of total streamflow based on an analysis of base flows at the four USGS streamgages (fig. 1) in the watershed using the streamflow partitioning algorithm, PART (table 2-1; Rutledge, 1998).

Table 2-1. Runoff and base-flow characteristics in the Saco River headwaters, 2000 – 2021.

Runoff calculated with data from U.S. Geological Survey (2025). Base flow calculated using the streamflow partitioning algorithm, PART (Rutledge, 1998). [cm, centimeters].

U.S. Geological Survey site number	Site name	Mean annual discharge (cm)	Mean annual base flow (cm)	Time period
010642505	Saco R. at River Street, Bartlett, NH	106.9	68.4	2010 - 2021
01064500	Saco R. near Conway, NH	95.7	64.4	2000 - 2021
01064801	Bearcamp R. at S. Tamworth, NH	83.2	53	2000 - 2021
01066000	Saco R. at Cornish, ME[1]	79.3	65.3	2000 - 2021

[1] Site affected by upstream flow regulation for power generation

Model Design

The SWB model code (Westenbroek and others, 2018) uses a modified Thornthwaite-Mather soil-moisture accounting method (Thornthwaite, 1948; Thornthwaite and Mather, 1957) to calculate potential recharge based on meteorological, soil, and land cover data. Individual components of the water budget were calculated according to user-activated modules for each grid cell in the model domain on a daily time step. In this model, precipitation was partitioned into runoff and an initial abstraction term using the Natural Resources Conservation Service (NRCS) curve number method (Cronshey and others, 1986), potential evapotranspiration (ET) was estimated by the Hargreaves-Samani method (Hargreaves and Samani, 1985), actual evapotranspiration and soil moisture were estimated by the Thornthwaite-Mather method (Thornthwaite and Mather, 1957), and snowmelt was estimated using a temperature-index method (Dripps and Bradbury, 2007). Soil moisture was updated at a daily time step as the difference between each grid cell's sources (precipitation, runon, snowmelt) and sinks (interception, runoff, and ET), as in equation 2-1:

$$\theta_t = \theta_{(t-1)} + precipitation + runon + snowmelt - interception - runoff - ET \quad (2\text{-}1)$$

where

$\theta_{(t)}$ *is the soil moisture on the current similation day,*

$\theta_{(t-1)}$ *is the soil moisture on the previous similation day, and*

ET *is the actual evapotranspiration.*

When soil moisture exceeded a specified available water capacity (AWC) for a grid cell, net infiltration was assumed to take place within a daily limit specified by a maximum net infiltration parameter. This limit prevented the model from calculating unreasonably high recharge values. Flow routing was enabled in this model, so any amount of infiltration greater than the specified maximum net infiltration for each grid cell was routed to the next downslope cell as runoff. This approach allowed runoff from upland areas to infiltrate on the edges of the sand and gravel deposits in the valley.

Input Datasets

The SWB model required input datasets including: (1) daily meteorological data, (2) hydrologic soil groups, (3) available water capacity, (4) land cover, and (5) surface water flow direction. Maps showing the gridded input datasets are provided in the companion data release (Holland and Barclay, 2026). The time-varying meteorological data inputs included daily precipitation, daily maximum temperature, and daily minimum temperature. For the historical simulation, these data were obtained from Oak Ridge National Laboratory's (ORNL) Daymet

Version 4 data (Thornton and others, 2022) for January 1999 – December 2021. The Daymet data provided gridded estimates of daily weather patterns across North America at a 1-kilometer spatial resolution by interpolating ground-based meteorological observations with an inverse-distance weighting technique. Daily climate data were required for 1999 to properly initialize the state of soil moisture for the first year of the desired simulation (2000). Simulations representing future periods (2046-2065 and 2080-2099) used an ensemble of meteorological data obtained from five downscaled General Circulation Models (GCMs) and two greenhouse-gas emission scenarios (U.S. Bureau of Reclamation, 2013) that are described in the Future Climate Scenarios for the SWB Model section of this appendix.

Hydrologic soil group (HSG) and AWC data were obtained through the NRCS Soil Survey Geographic Database (SSURGO; Soil Survey Staff, 2022) using the NRCS Soil Data Development Toolbox for ArcGIS. The HSG can take on a value of A, A/D, B, B/D, C, C/D or D based on the infiltration capacity of the soil (Cronshey and others, 1986). For soils assigned to a dual hydrologic group (A/D, B/D, and C/D), the first letter corresponds to the drained condition, and the second letter corresponds to the undrained condition. The AWC is the maximum amount of plant-available water that is contained in the soil column, and in this dataset, AWC was averaged across the top 100 cm of soil. The AWC data were provided to the SWB model as a dimensionless fraction of water depth per soil thickness, and conversion to inches of water per foot of soil thickness occurred within the model code. The SWB model ultimately calculated AWC for each model grid cell by multiplying the AWC fraction by the specified root-zone depth for each grid cell.

Land cover data were obtained from the 2019 National Land Cover Database (NLCD) (Dewitz and U.S. Geological Survey, 2021) available from the Multi-Resolution Land

Characteristics Consortium. The NLCD data were used as a static land cover layer for the entire duration of the simulated periods (1999 to 2099) under the assumption that land cover does not change during this time. While land cover would likely change over this 100-year period, the effects of land cover change were not a focus of this study and therefore remained static in all simulations. The land cover layer consisted of 15 land cover classes across the study area at a 30-m resolution. Land cover data were resampled to the larger SWB model grid cells using a majority resampling technique, where each SWB model grid cell was assigned the dominant land cover class within the grid cell.

The “D8 flow direction” method was selected for the SWB model which allows routing runoff from one or more cells to downslope cells. In this method, each cell is assigned a flow direction code that defines the flow in one of eight directions to an adjacent cell based on the direction of the steepest slope. The D8 flow direction was generated from the National Elevation Dataset (Gesch and others, 2002).

Parameter Values

The SWB model required input parameters to control how inputs to the hydrologic system were partitioned, including maximum net infiltration, runoff curve number, root-zone depth, growing season start and end date, and precipitation interception storage. These parameters were supplied by a lookup table where parameter values were a function of land cover and hydrologic soil group. The maximum net infiltration parameter indicated the maximum allowable daily infiltration rate. Calculated net infiltration greater than the maximum net infiltration rate was considered “rejected net infiltration” and was routed to the next downslope grid cell to prevent SWB from calculating unreasonably high recharge values. The runoff curve number values were used in the curve number method (Cronshey and others, 1986) to partition precipitation into runoff and an initial abstraction term. Root-zone depth was used to calculate total AWC. In total, there were 375 parameter values supplied by the lookup table. Parameter values were adapted from a published SWB model for Maine (Nielsen and Westenbroek, 2019) and a published SWB model for southern New England (Holland and Barclay, 2024). The final parameters are provided in a USGS data release (Holland and Barclay, 2026).

Cells designated “Open Water” in the NLCD 2019 input land cover layer were excluded from the SWB calculations because SWB does not have a mechanism to simulate recharge in ponded areas. Instead, “pond recharge” was calculated separately for the groundwater-flow model to fill in missing values in the “Open Water” cells as described in appendix 1.

Calculated Potential Recharge

The simulated mean annual potential recharge rates from 2000 – 2021 are shown in figures 2-1 and 2-2. The recharge rates for individual cells in the SWB model varied with factors such as soil characteristics, land use, and the orographic effect on precipitation, with higher precipitation rates at higher elevations. The monthly recharge rates in Wards Brook valley were somewhat smaller than for the Saco River headwaters due to its lower elevation (fig. 2-3). Recharge for 2016 – 2021 is largest in the non-growing season, primarily January through April and October through December, with limited recharge during the summer months.

Model Verification

Verification of the simulated recharge rates was necessary because limited streamflow data were available to constrain volumetric flow in the groundwater-flow model. Outputs from the SWB model were compared to values calculated using the RORA recession-curve displacement method (Rutledge, 1998) and published recharge values from an existing SWB model for Maine (Nielsen and Westenbroek, 2019) and a Precipitation-Runoff Modeling System (PRMS) for New Hampshire (Bjerklie and Sturtevant, 2018).

RORA Model Comparison

Recharge outputs summarized by Saco River Basin streamgage watersheds (fig. 1) were compared with an analysis of recharge rates from a RORA model, which estimates average groundwater recharge at the quarterly to annual time scale from daily streamflow data in a watershed (Rutledge, 1998). The RORA model was implemented with the DVstats package in the statistical software R, which includes functions to determine the recession index and estimated annual recharge (Lorenz, 2017; R Core Team, 2021). For this analysis, available streamflow data from three streamgages in the Saco River headwaters from 1980 – 2021 (U.S. Geological Survey, 2022) were used to determine recession index and recharge. The two streamgages for the upgradient watersheds (USGS streamgages 01064801 and 010642505; U.S. Geological Survey, 2022) had a shorter period of record than the downstream streamgage (USGS streamgage 01064500).

The mean recharge rates from the RORA model were determined for each streamgage site and compared with the mean recharge determined from SWB for each year. The plots of these data with comparison to a 1:1 line demonstrate the range of annual recharge rates for parts of the Saco Rivers headwaters and confirm that the SWB estimates were similar in magnitude (fig. 2-4). However, mean annual recharge calculated by SWB averaged across the gaged areas tended to be lower than recharge estimated with the RORA method (table 2-2). The prevalence of complex topography in these three small subbasins may limit the accuracy of SWB-simulated recharge due to complexities of rainfall runoff processes combined with soil data and meteorological data limitations in these areas.

Table 2-2. Comparison of Saco River headwaters soil-water-balance (SWB) model annual net infiltration with RORA-derived recharge for watersheds draining to three U.S. Geological Survey (USGS) streamgages, 2000 – 2021 (U.S. Geological Survey, 2022).

USGS streamgage station number	USGS streamgage station name	Mean annual recharge, SWB model (meters)	Mean annual recharge, RORA method (meters)	R-squared
01064500	Saco River near Conway, NH	0.69	0.74	0.84
01064801	Bearcamp River at South Tamworth, NH	0.59	0.62	0.74
010642505	Saco River at River St at Bartlett, NH	0.73	0.77	0.70

Maine Soil-Water-Balance Model Comparison

A published SWB model for Maine (herein referred to as the Maine SWB model) provided grids of minimum, maximum, and mean annual recharge for 1991-2015 (Nielsen and Westenbroek, 2019). To facilitate comparison of Saco River headwaters SWB recharge results with that study, both Saco River headwaters SWB output and Maine SWB model published data were summarized across only grid cells that overlapped between the two models. The Saco River headwaters SWB model had a 2000-2021 mean recharge rate of 0.51 meters/year (m/yr), while the Maine SWB model had a 1991-2015 mean recharge rate of 0.55 m/yr (fig. 2-5). This comparison confirms that outputs from the Saco River headwaters SWB model were within a range of reasonable values for this region.

Precipitation-Runoff Modeling System Comparison

A PRMS model for New Hampshire (Bjerklie and Sturtevant, 2018) reported monthly simulated recharge by Hydrologic Response Unit (HRU). The PRMS recharge was compared to the Saco River headwaters SWB model recharge for the overlapping period of 2000 – 2005 (fig. 2-6). Overall, there is good agreement between PRMS output for New Hampshire HRUs that overlapped areas of the Saco River headwaters SWB model (Pearson's correlation coefficient [r] = 0.72). Disagreement in monthly recharge values may be explained by key differences between the SWB and PRMS models such as underlying modeling schemes, parameterizations, and scale. In addition, the PRMS model was driven by daily input climate datasets derived by Maurer and others (2002) while the Saco River headwaters SWB model is driven by climate data from Daymet Version 4 (Thornton and others, 2022).

Future Climate Scenarios for the Soil-Water-Balance Model

Potential recharge estimates for hypothetical climate scenarios were developed by creating SWB simulations that used meteorological data inputs from a stabilized greenhouse-gas emission scenario (Representative Concentration Pathway [RCP] 4.5) and a high greenhouse-gas emission scenario (RCP 8.5) for the 2016 – 2099 period (Stocker and others, 2013). The two greenhouse-gas emission scenarios (hereafter referred to as emission scenarios) were each driven by five GCMs (table 2-2) that were downloaded from the USGS Geo Data Portal, Bias Corrected

Constructed Analogs V2 Daily Future Coupled Model Intercomparison Project (CMIP5) Climate Projections (U.S. Geological Survey, undated; U.S. Bureau of Reclamation, 2013) to obtain a set of 10 SWB simulations. The downscaled climate models provided daily precipitation, daily minimum near-surface air temperature, and daily maximum near-surface air temperature for each cell of a 0.125-degree by 0.125-degree (about 12 kilometers) grid. An earlier study of the effects of climate change on hydrology in New Hampshire using a PRMS model (Bjerklie and Sturtevant, 2018) used an ensemble of five GCMs to provide a distribution of future outcomes under each climate scenario. To facilitate comparison between the New Hampshire PRMS model and this study, the same five GCMs were used (table 2-2). For each GCM, the r1i1p1 (realization=1, initialization method=1, and physics version=1) ensemble member was selected.

Table 2-2. General Circulation Models used for emission scenarios, 2016 – 2099 (U.S. Bureau of Reclamation, 2013).

General Circulation Model	Abbreviation
Centre National de Recherches Meteorologiques / Centre Europeen de Recherche et Formation Avancees en Calcul Scientifique	CNRM-CM5
Commonwealth Scientific and Industrial Research Organisation in collaboration with the Queensland Climate Change Centre of Excellence	CSIRO-Mk3-6-0
Geophysical Fluid Dynamics Laboratory	GFDL-ESM2G
Max Planck Institute for Meteorology (MPI-M)	MPI-ESM-MR
Meteorological Research Institute	MRI-CGCM3

RCP 4.5 represents a stabilized-emission scenario where greenhouse-gas emissions peak around the year 2040 and then decline. RCP 4.5 is generally associated with lower population growth, more technological innovation, and lower carbon intensity of the global energy mix (Reidmiller and others, 2018). According to Thomson and others (2011) “RCP 4.5 is a

stabilization scenario and assumes that climate policies, in this instance the introduction of a set of global greenhouse gas emissions prices, are invoked to achieve the goal of limiting emissions and radiative forcing." RCP 8.5 is a high-emission scenario that has no climate mitigation target and assumes emissions increasing over time (Riahi and others, 2011). It represents higher population growth, less technological innovation, and higher carbon intensity of the global energy mix than RCP 4.5 (Reidmiller and others, 2018). Annual mean temperature and precipitation from the five GCMs and 2 RCPs demonstrate the general trend and differences between RCP 4.5 and 8.5 (fig. 2-7).

Recharge results from each of the 10 SWB simulations were extracted for the groundwater-flow model area, and mean monthly values were calculated for the 6-year historical period (2016 – 2021) and for two 20-year periods (2046 – 2065 and 2080 – 2099). The simulated potential recharge was then used as the recharge dataset for the groundwater-flow model. Potential recharge from the climate models was simulated for 2016 – 2021 to determine whether projected climate outputs reasonably replicated measured data that were collected after the climate projections were published in 2012 and 2013 (U.S. Bureau of Reclamation, 2013). The 20-year simulation periods were deemed a sufficient length of time to provide representative mean monthly values and also enabled comparison with results from the New Hampshire PRMS model that were simulated for those periods.

Soil-Water-Balance Model Limitations

The SWB model is a physically based method for estimating spatially variable recharge for monthly and yearly periods but is a simplification of the hydrologic system and has limitations. Descriptions of SWB model limitations and assumptions are provided in

Westenbroek and others (2018) and Nielsen and Westenbroek (2019). Limitations most important for this study include:

1. The SWB model simulates net infiltration of water and therefore represents the amount of water that would become recharge if all of it reached the water table. Under this assumption, the model more accurately simulates net infiltration where soils extend to the water table, but not where bedrock surfaces are above the water table. Soil and unsaturated zone conditions in parts of the Saco River headwaters may not yield accurate potential recharge; however, conditions in Wards Brook valley facilitated more reliable recharge estimates.

2. The SWB model is not equipped to handle conditions where the water table is near the land surface, and wetland areas within Wards Brook valley may not be simulated accurately with respect to losses of shallow groundwater to evapotranspiration. To compensate for this limitation, the groundwater-flow model was constructed with drains in wetland areas to simulate shallow groundwater losses to evapotranspiration.

3. The reliability of outputs from a water-balance model depends on the accuracy of the model inputs, particularly when the magnitude of the model output is much smaller than the magnitude of the inputs. Precipitation rates were much greater than recharge, and the meteorological data for the historical and future periods were obtained from sources that estimated the spatial distribution of daily precipitation and temperature data at a coarser resolution than the SWB model. Generalized meteorological conditions could cause errors in estimated recharge, particularly in mountainous areas, like the Saco River headwaters, where meteorological conditions can be highly heterogenous.

4. Estimated recharge based on projected future climate conditions from the emission scenarios cannot account for potential changes to the intensity of rainfall events that might result from climate change.

Despite these potential limitations, comparison with other methods used for estimating recharge in the Saco River headwaters provided evidence that the SWB model simulates reasonably reliable recharge estimates for the purposes of this study.

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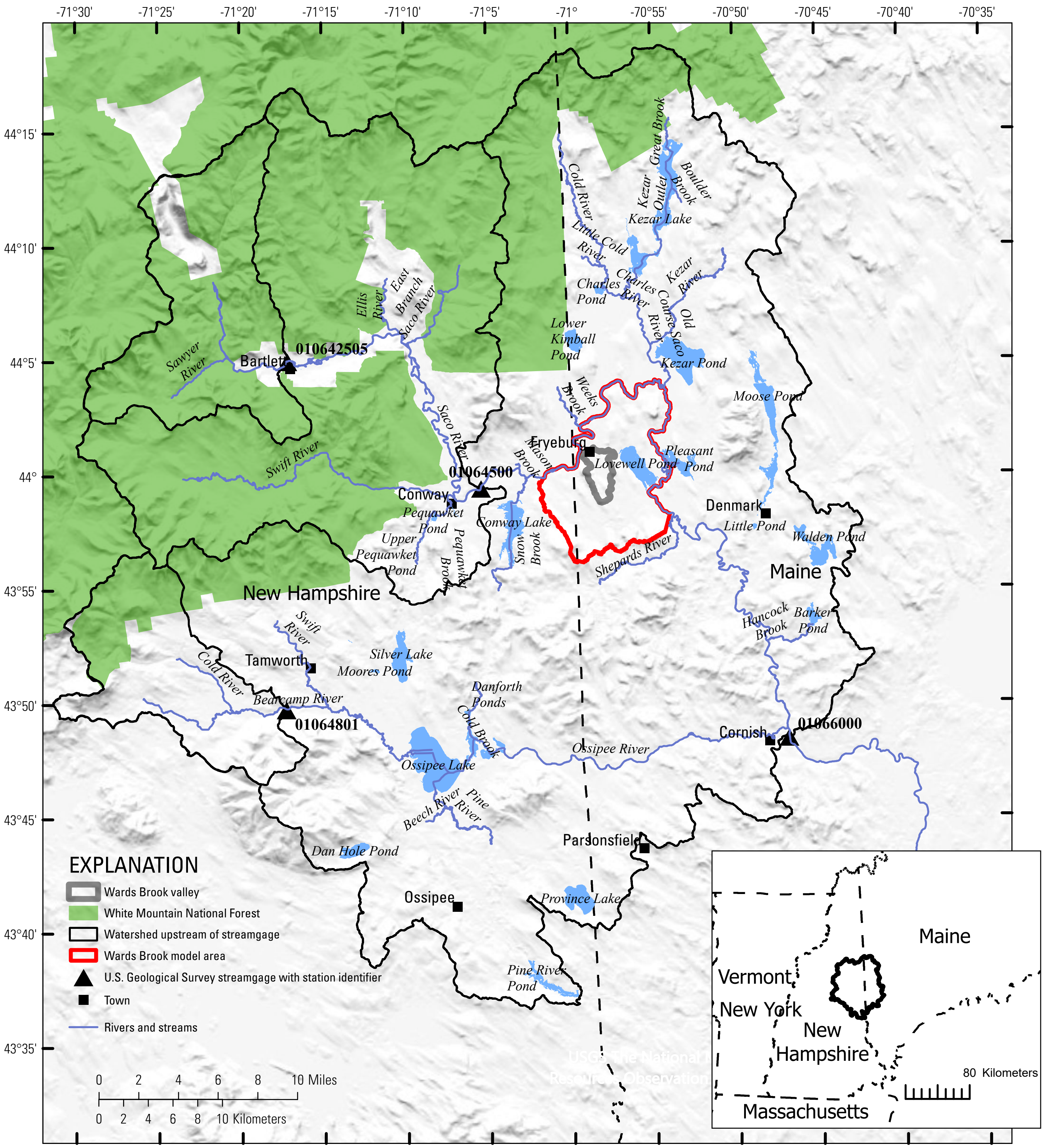
FIGURES



Base from U.S. Geological Survey, The National Map.
New Hampshire State Plane FIPS 2800 projection
North American Datum of 1983

Stream and waterbody information from the NHDplus high resolutiondataset, Moore and others, 2019. Streamgage watersheds from Streamstats, U.S. Geological Survey, 2023.

Figure 1. Wards Brook valley and Saco River headwaters, New Hampshire and Maine.

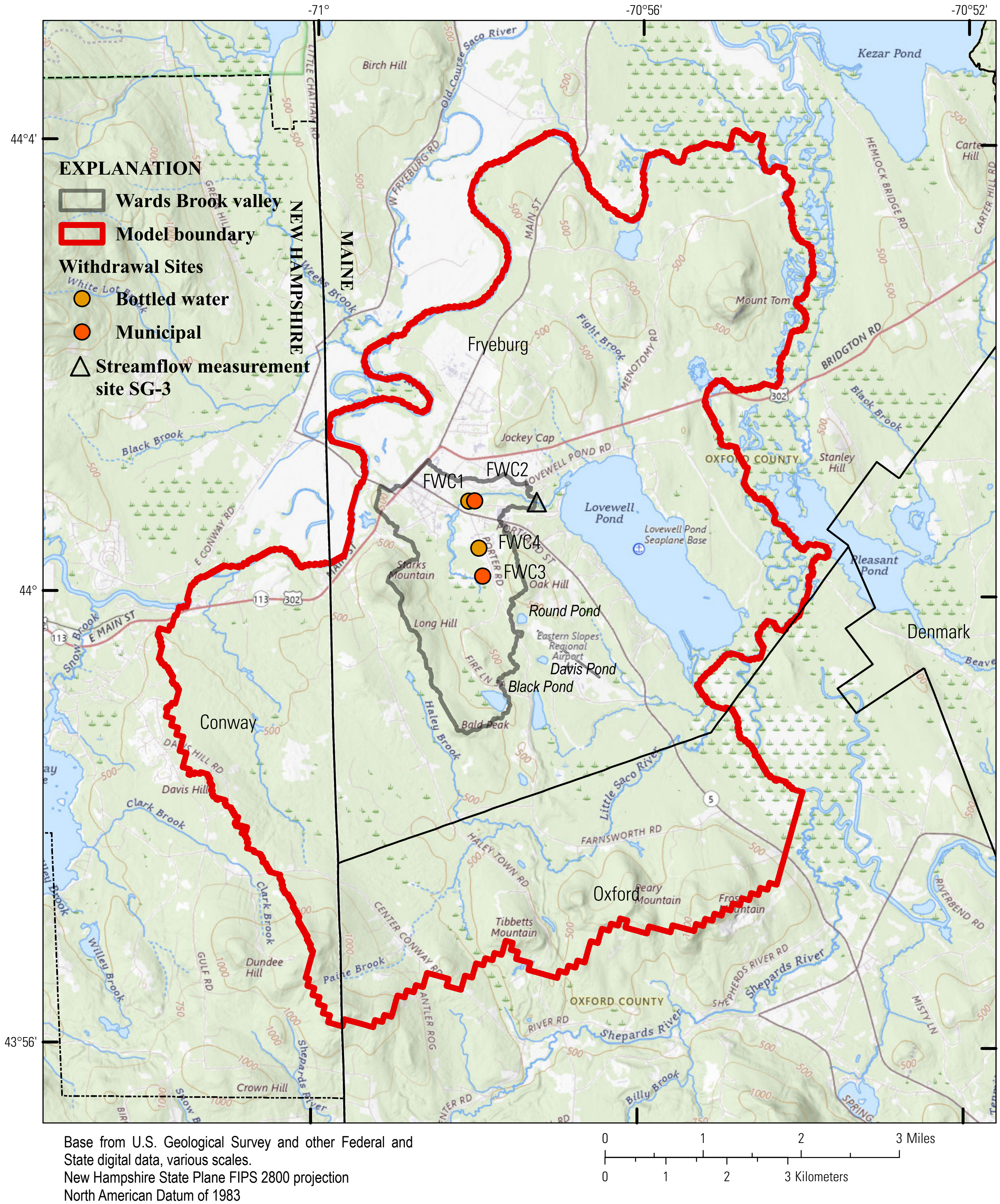


Figure 2. Wards Brook valley, groundwater-flow model boundary, municipal and bottled water well sites, and streamflow measurement site SG-3, Maine and New Hampshire. Streamflow site SG-3 measured by Luetje Geological Services LLC and McDonald Morrissey Associates (2021, 2022) [FWC, Fryeburg Water Company. Wells 1 and 4 are used for bottled water extraction, and wells 2 and 3 are used for municipal water supplies.]

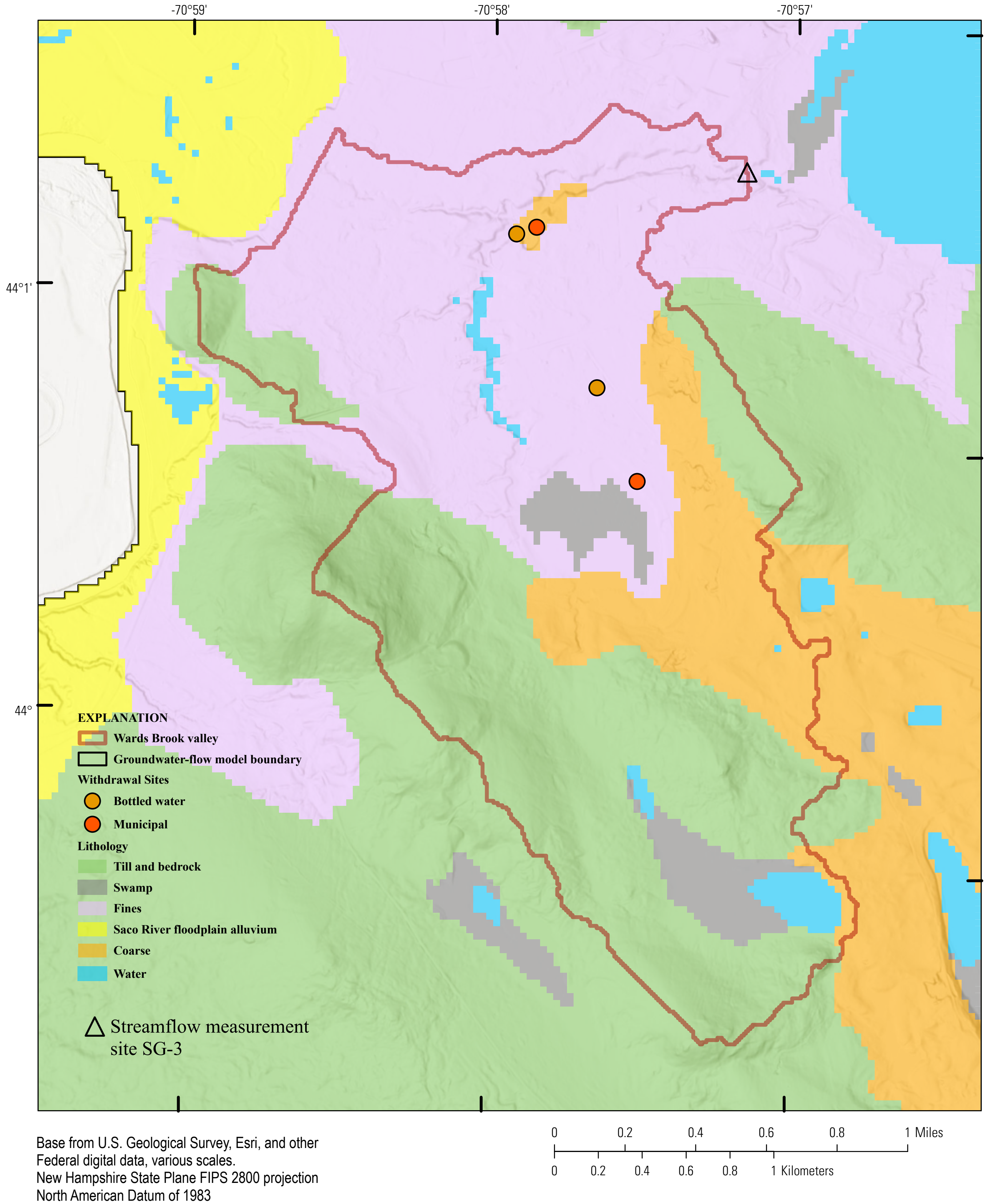


Figure 3. Generalized lithology of Wards Brook valley, New Hampshire and Maine. Geology modified from Davis and Holland (1997a,b), Emery & Garrett Groundwater, Inc. (2005), and Thompson (2014).

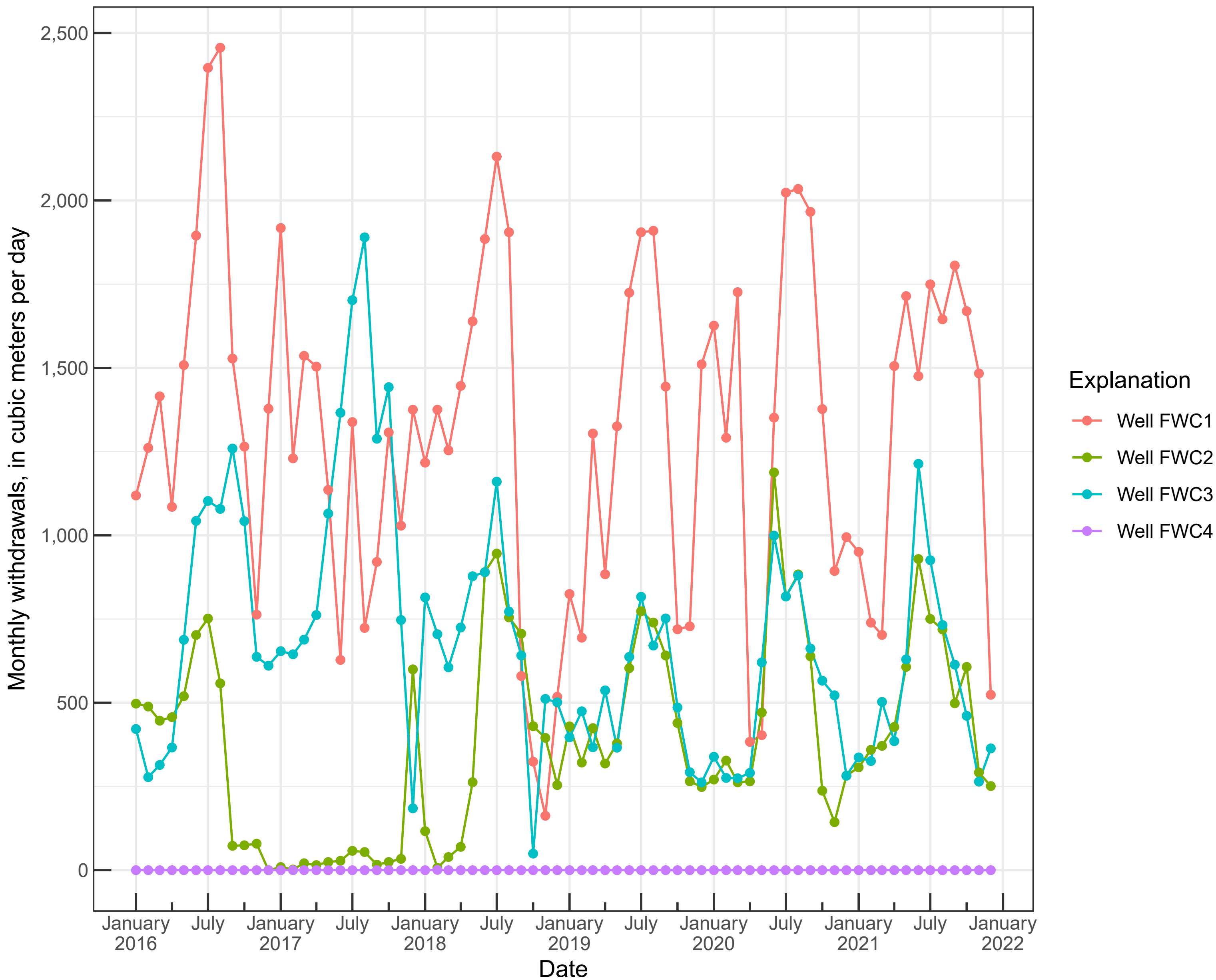


Figure 4. Monthly water withdrawals in Wards Brook valley for municipal and water bottling uses, 2016 to 2021 (Fryeburg Water Company, 2016, 2017, 2018, 2019, 2020, 2021; Luetje Geological Services LLC, 2016, 2017a,b, 2018a,b, 2019a,b, 2020; Luetje Geological Services LLC and McDonald Morrissey Associates, 2021, 2022).
[FWC, Fryeburg Water Company. Wells 1 and 4 are used for bottled water extraction, and wells 2 and 3 are used for municipal water supplies. Note that the daily withdrawal rates shown need to be applied to the days in each month to determine total monthly pumpage].

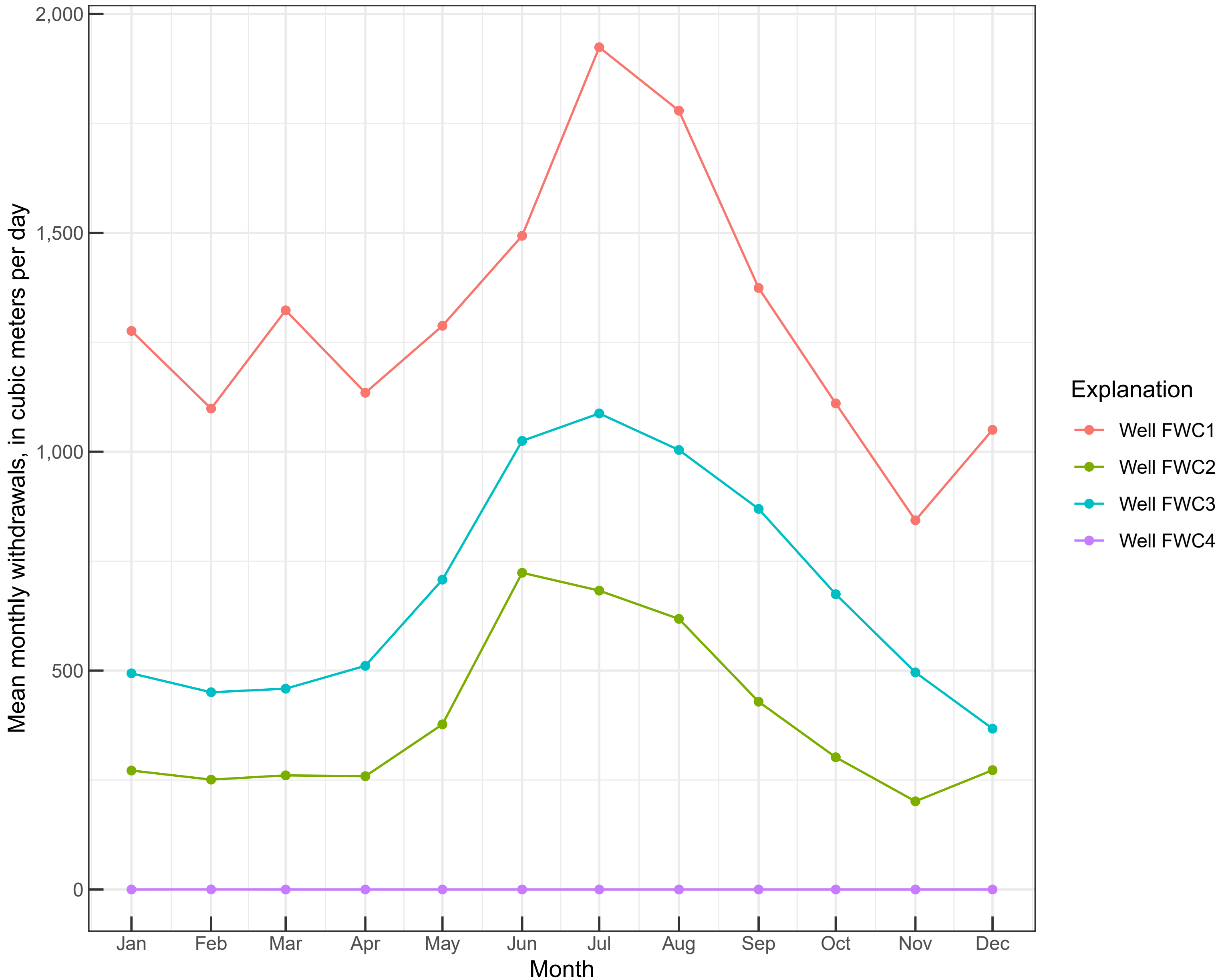


Figure 5. Mean monthly water withdrawals in Wards Brook valley for municipal and water bottling uses, 2016 to 2021 (Fryeburg Water Company, 2016, 2017, 2018, 2019, 2020, 2021; Luetje Geological Services LLC, 2016, 2017a,b, 2018a,b, 2019a,b, 2020; Luetje Geological Services LLC and McDonald Morrissey Associates, 2021, 2022). [FWC, Fryeburg Water Company. Wells 1 and 4 are used for bottled water extraction, and wells 2 and 3 are used for municipal water supplies. Note that the daily withdrawal rates shown need to be applied to the days in each month to determine total monthly pumpage].

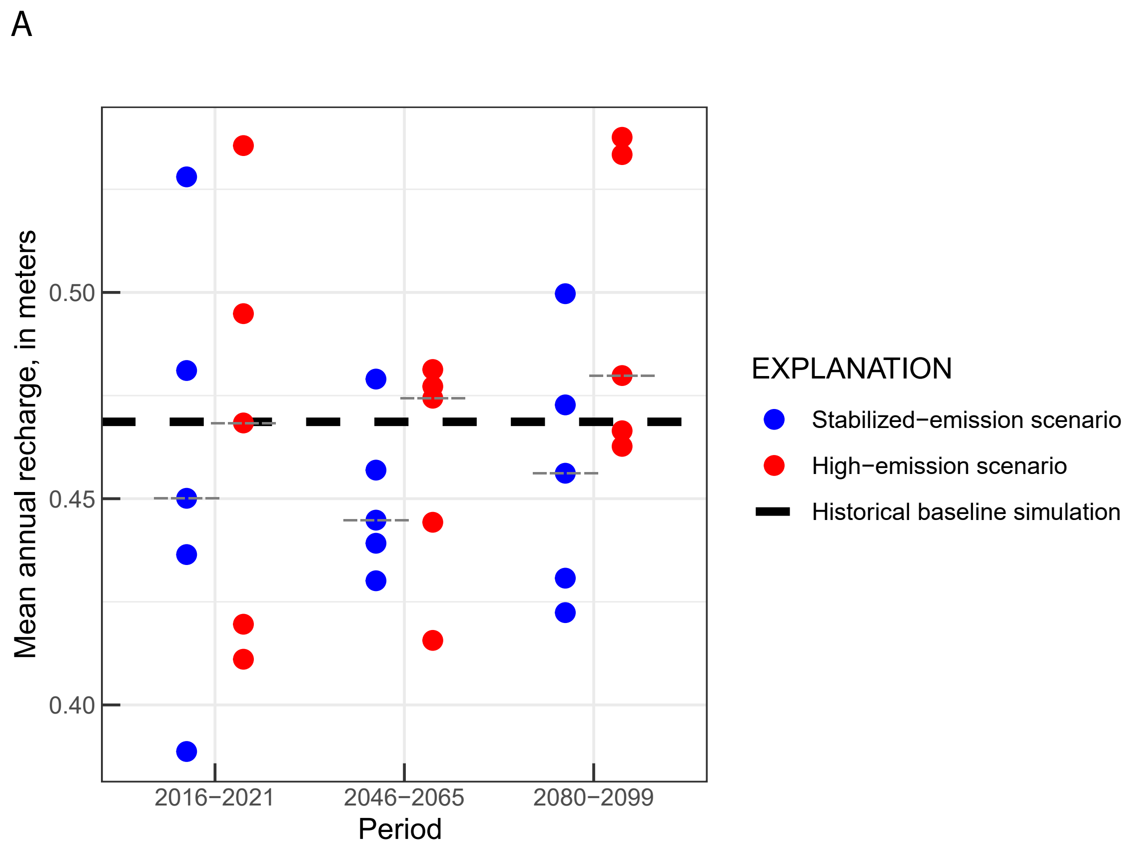
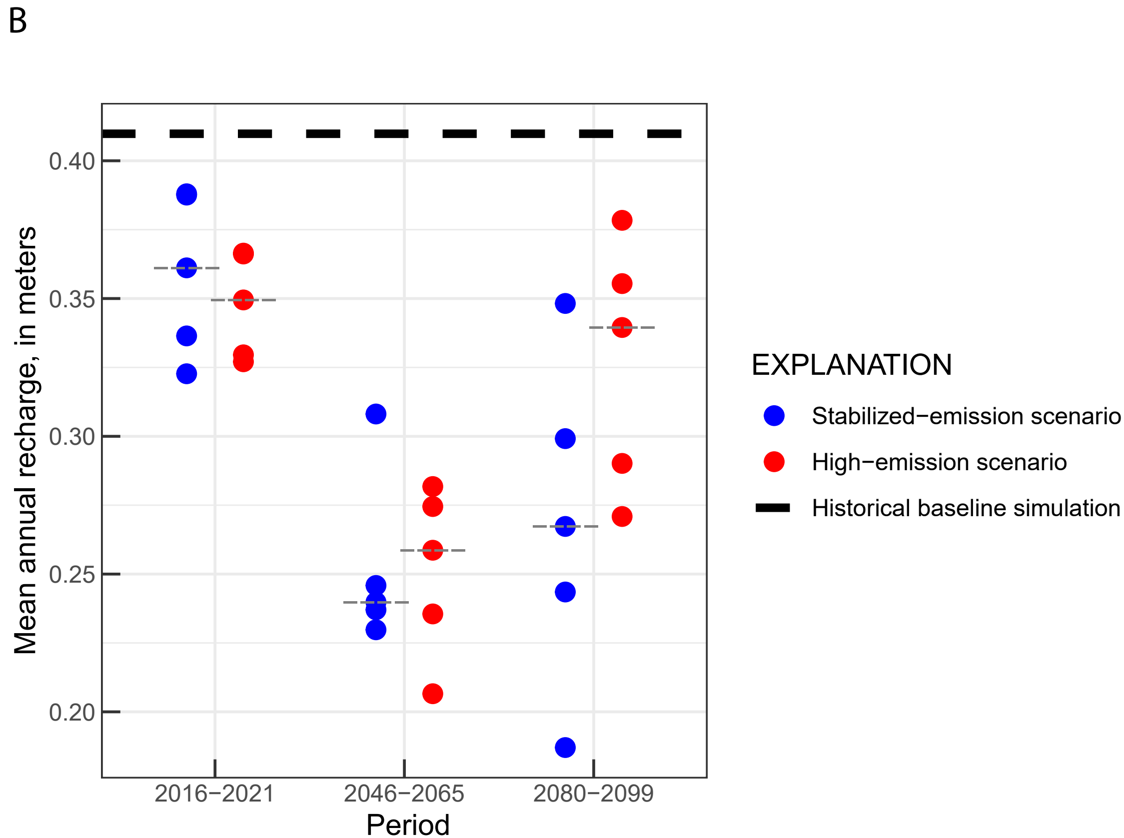


Figure 6. Mean annual recharge values for Wards Brook valley based on A. projected future emission scenarios for the mean of time periods 2016 – 2021, 2046 – 2065, and 2080 – 2099, and the baseline condition from historical data (2016 – 2021), and B. drought simulations for a 3-year drought based on the lowest annual recharge projected for emission scenarios (2016 – 2021, 2046 – 2065, and 2080 – 2099) and baseline conditions from historical data (2016 – 2021). Results from the emission scenarios are shown as results from the five General Circulation Models (median shown as a gray dashed line).



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A

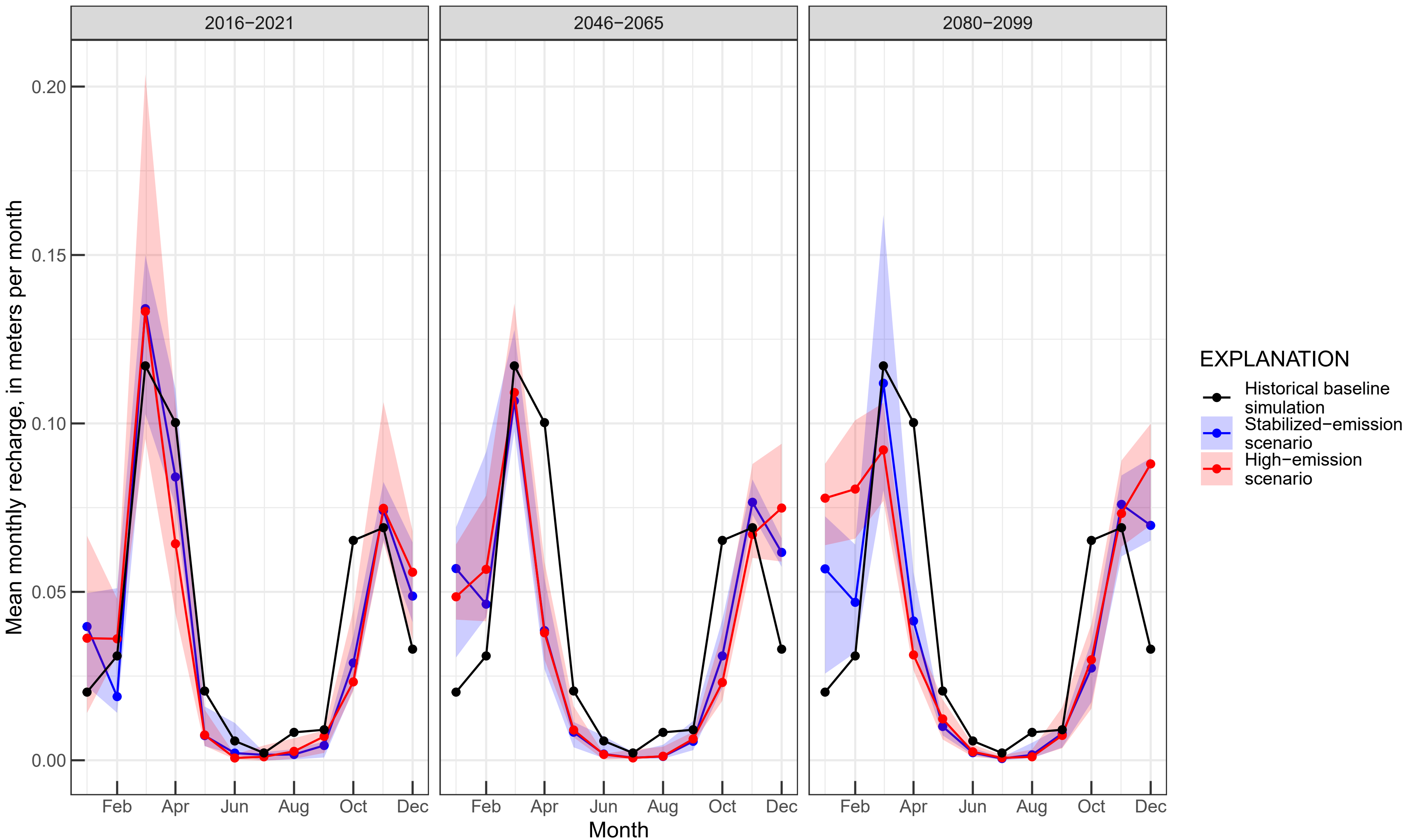
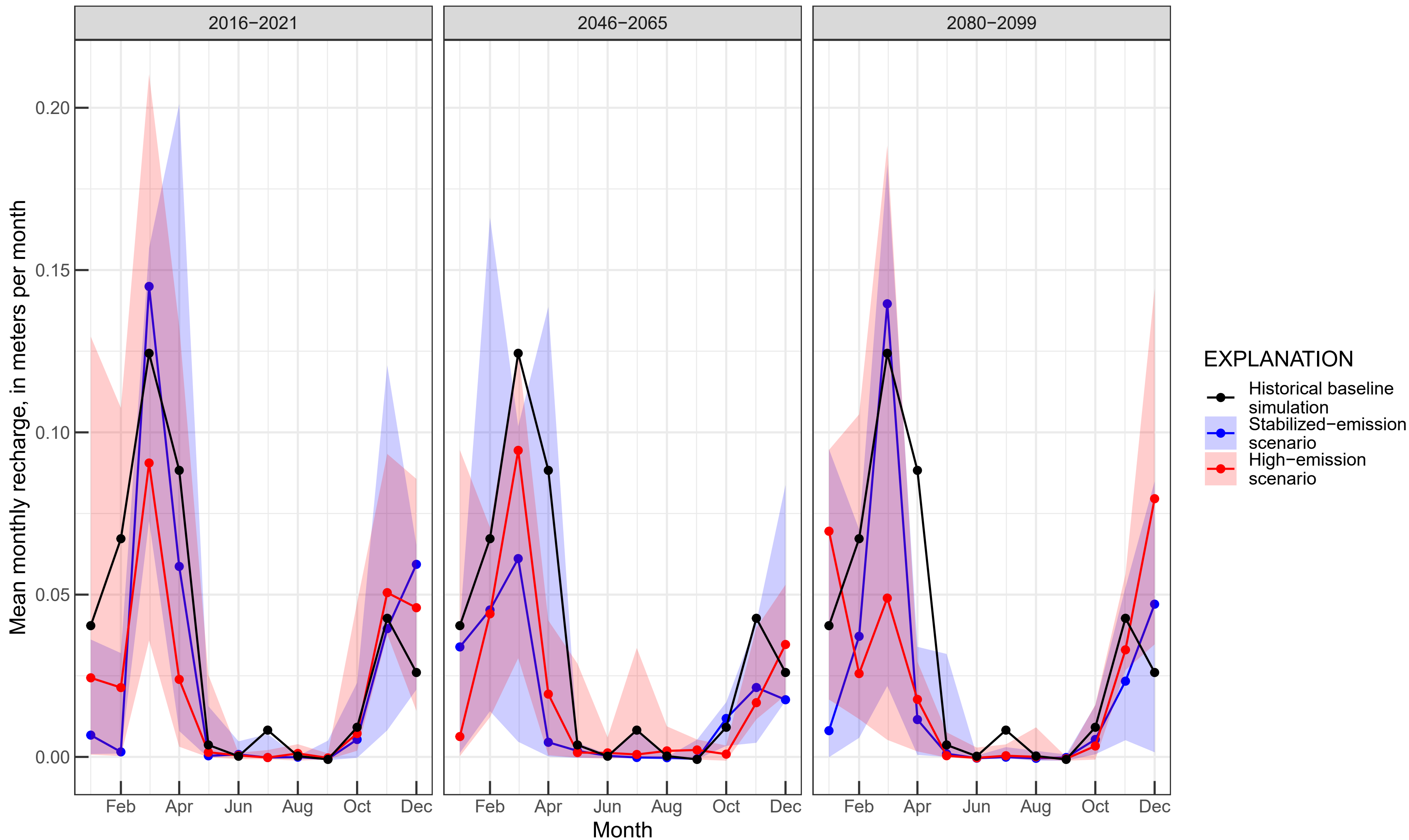


Figure 7. A. Simulated mean monthly recharge for the historical baseline simulation and two emission scenarios from five General Circulation Models, and B. simulated monthly recharge of the year with lowest annual recharge of each General Circulation Model and emission scenario, and the historical baseline simulation, 2016 – 2021, 2046 – 2065, and 2080 – 2099. Values shown for each month are the mean of all cells in Wards Brook valley. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded extent) and the median values (line) of the five General Circulation Models.

B



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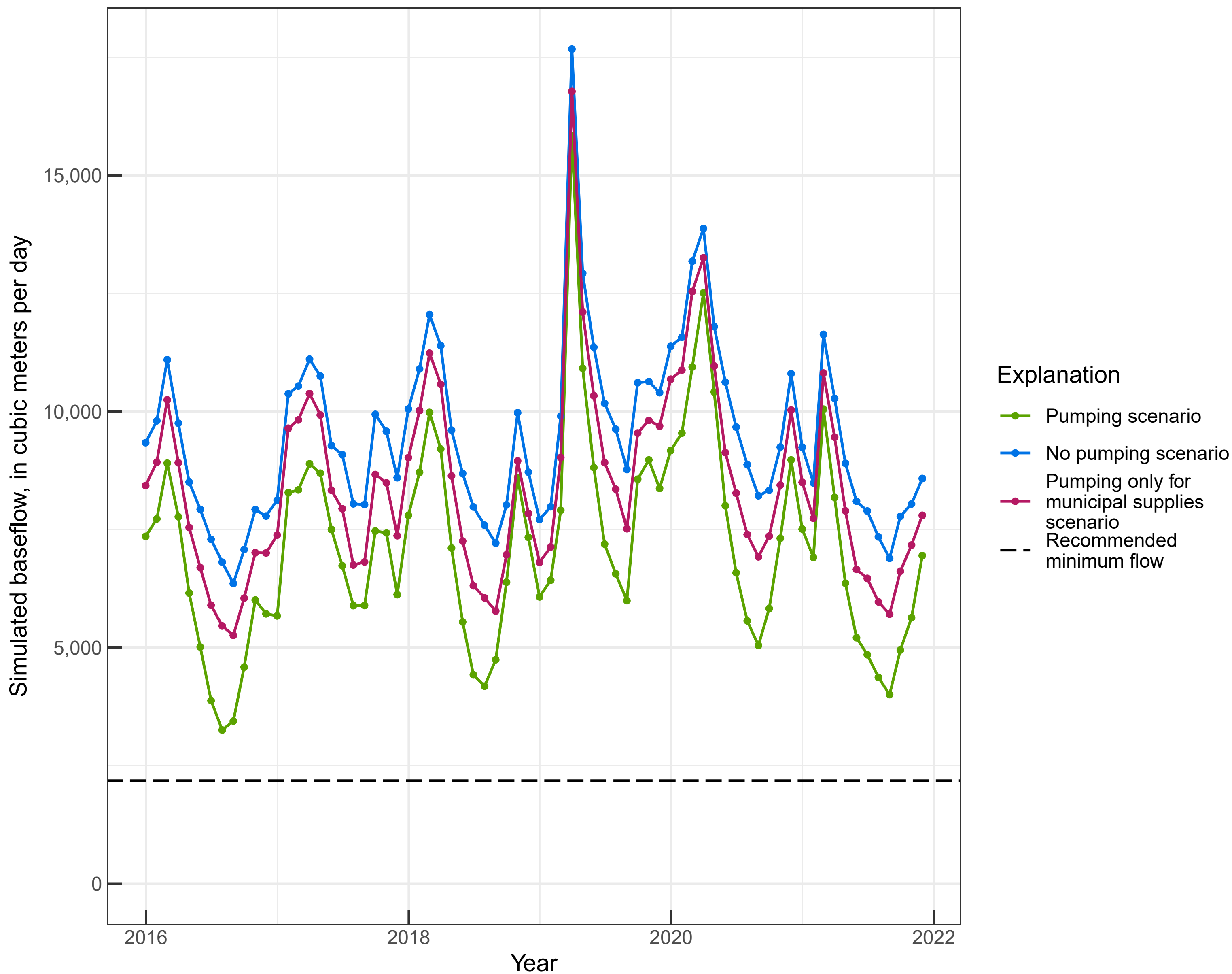


Figure 8. Simulated monthly stream base flows in Wards Brook upstream from Lovewell pond (site SG-3), 2016 – 2021, with pumping, no pumping, and pumping only for municipal supplies.

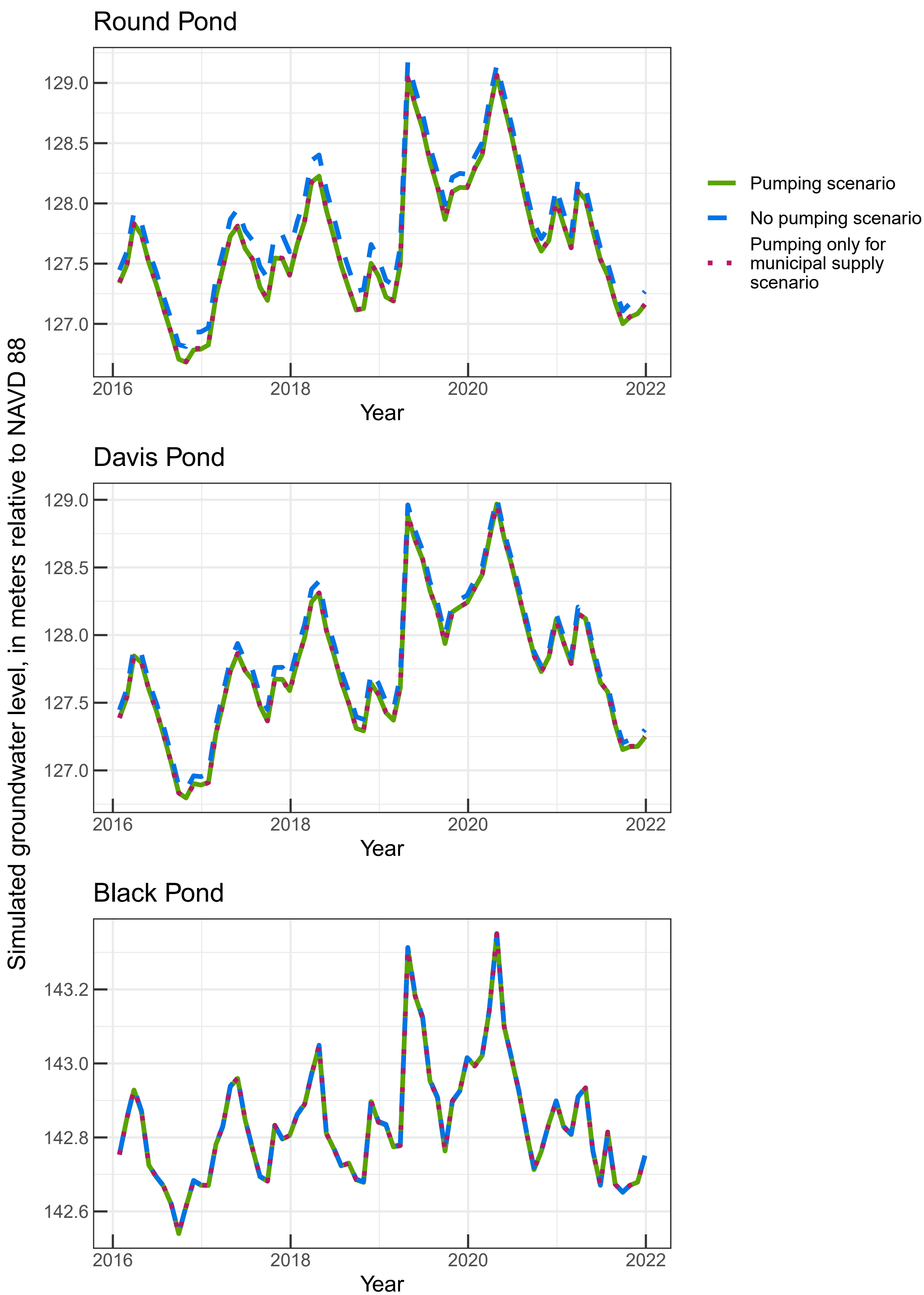


Figure 9. Simulated monthly groundwater levels at selected ponds with pumping, no pumping, and pumping only for municipal supplies, 2016 – 2021. [m, meter; NAVD 88, North American Vertical Datum of 1988]

A

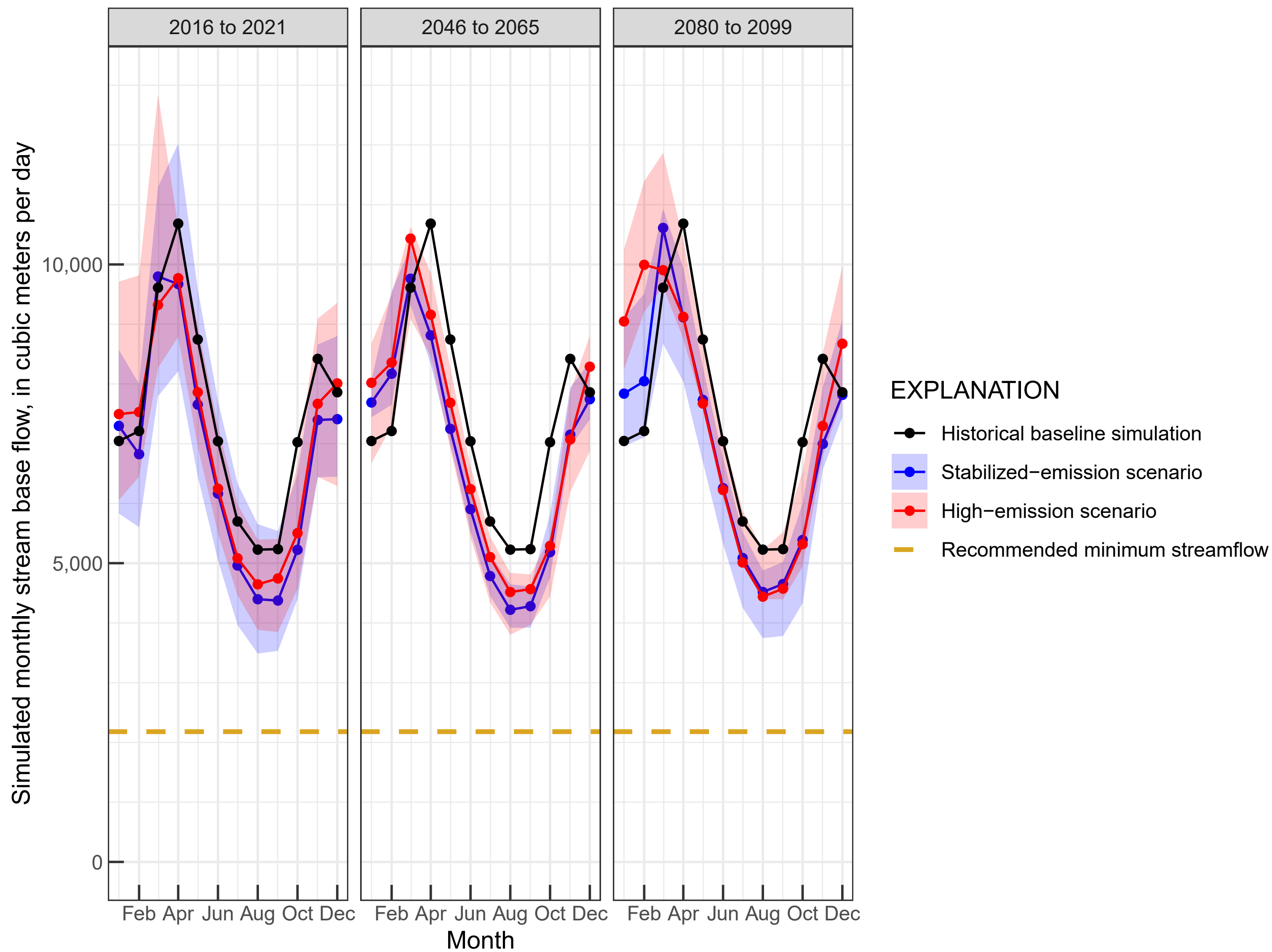


Figure 10. Monthly simulated stream base flows for historical baseline conditions (2016 – 2021), and the stabilized- and high-emission scenarios (2016 – 2021, 2046 – 2085, 2080 – 2099) for A. mean monthly of period conditions, and B. last year of a 3-year drought based on the lowest annual recharge in each General Circulation Model, emission scenario, and period. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models.

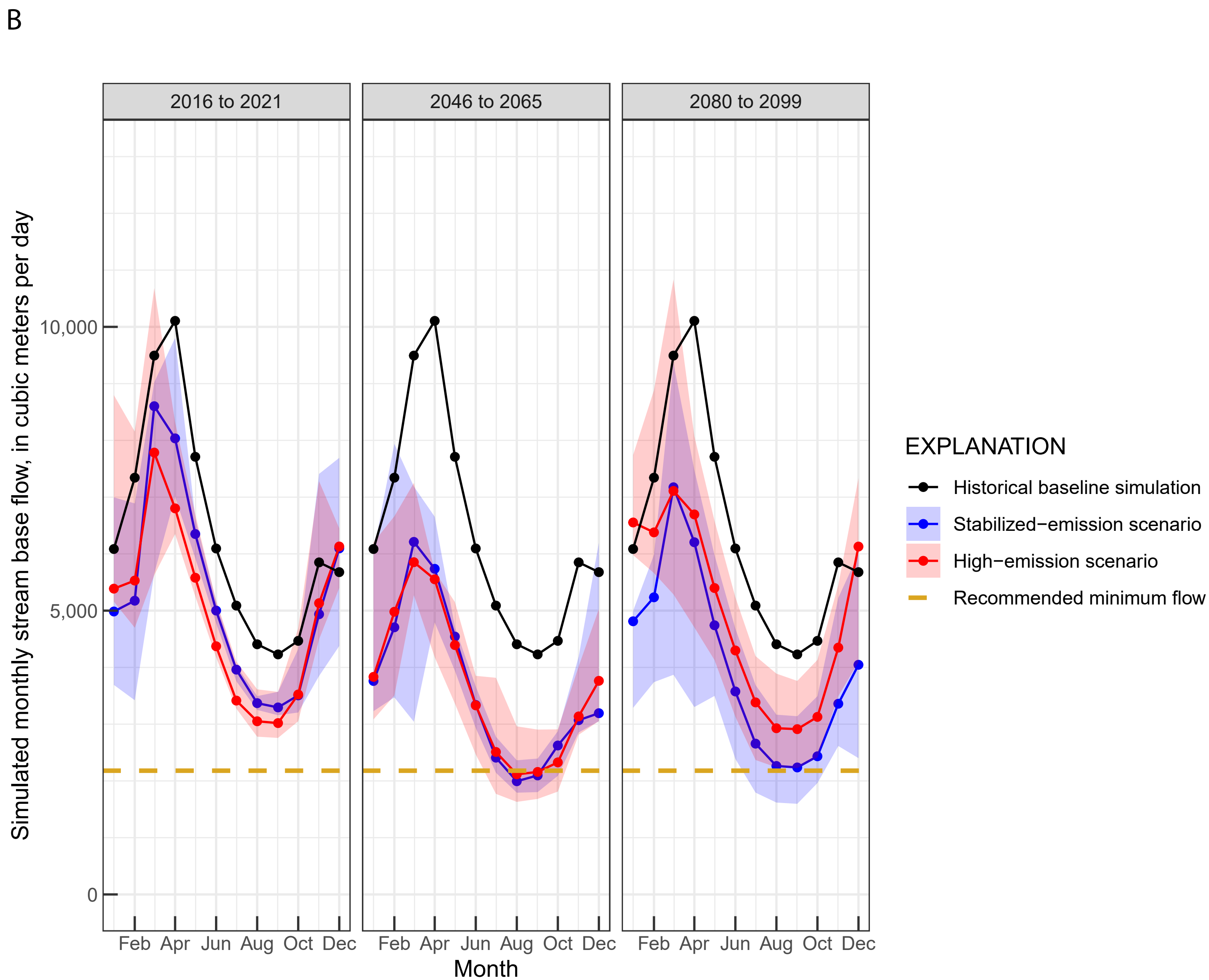


Figure 10. Monthly simulated stream base flows for historical baseline conditions (2016 – 2021), and the stabilized- and high-emission scenarios (2016 – 2021, 2046 – 2085, 2080 – 2099) for A. mean monthly of period conditions, and B. last year of a 3-year drought based on the lowest annual recharge in each General Circulation Model, emission scenario, and period. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models.

A

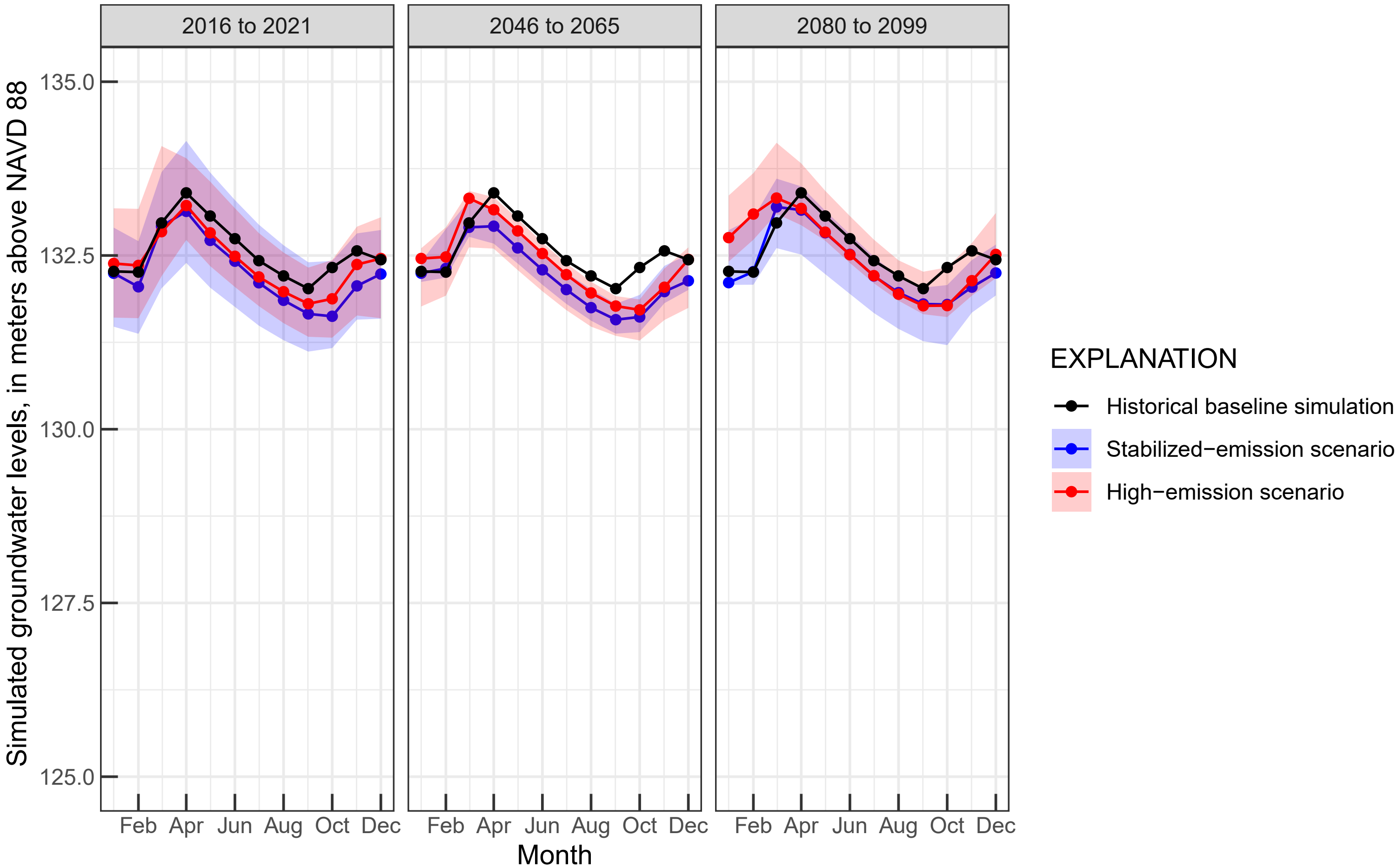


Figure 11. Simulated average groundwater-level elevations of layer 1 in the area of Wards Brook valley under historical baseline conditions (2016 – 2021), and for stabilized and high-emission scenarios (2016 – 2021, 2046 – 2085, 2080 – 2099) for A. mean monthly of period conditions, and B. last year of a 3-year drought based on the lowest annual recharge in each General Circulation Model, emission scenario, and period. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models. [NAVD 88, North American Datum of 1988]

B

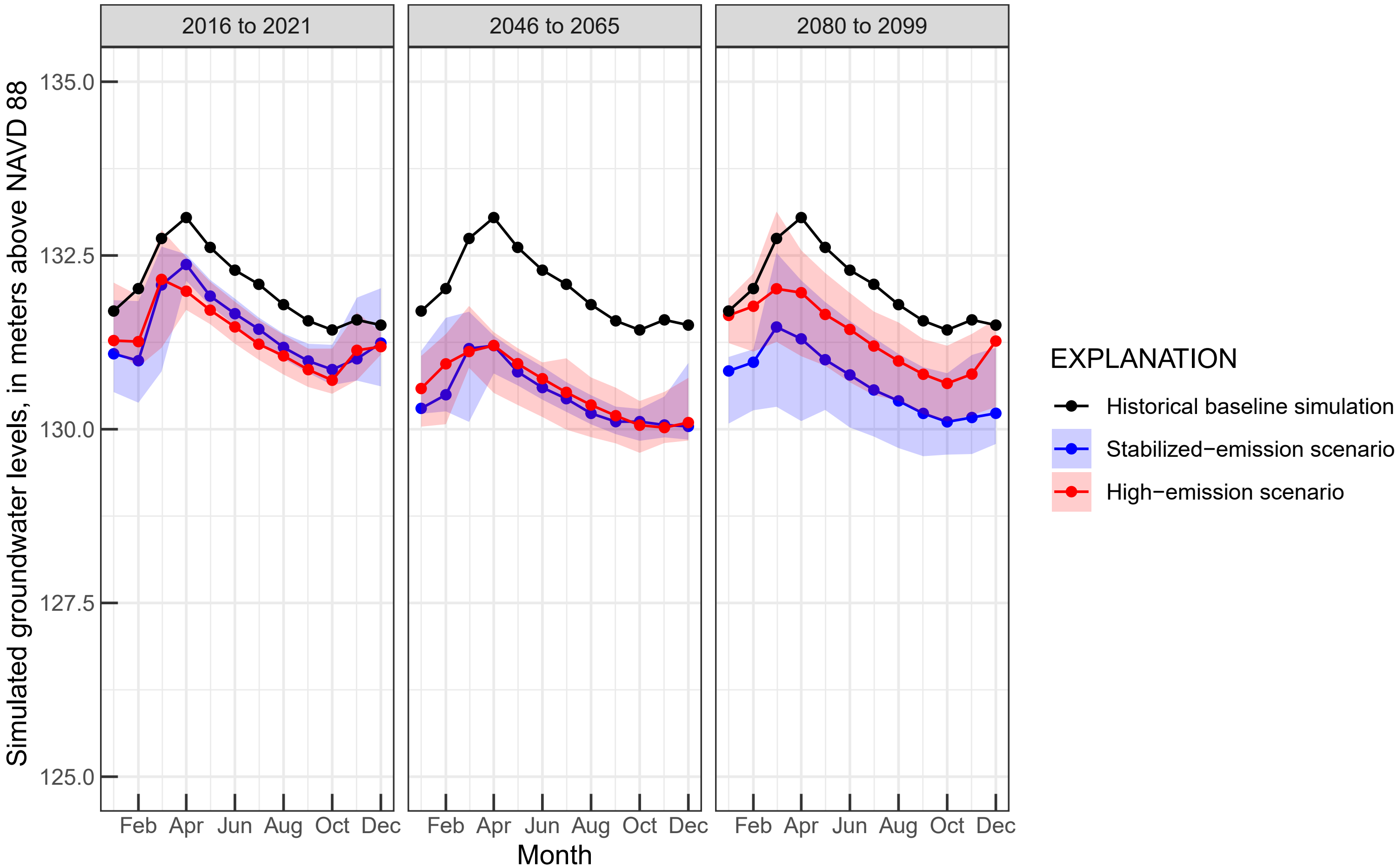


Figure 11. Simulated average groundwater-level elevations of layer 1 in the area of Wards Brook valley under historical baseline conditions (2016 – 2021), and for stabilized and high-emission scenarios (2016 – 2021, 2046 – 2085, 2080 – 2099) for A. mean monthly of period conditions, and B. last year of a 3-year drought based on the lowest annual recharge in each General Circulation Model, emission scenario, and period. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models. [NAVD 88, North American Datum of 1988]

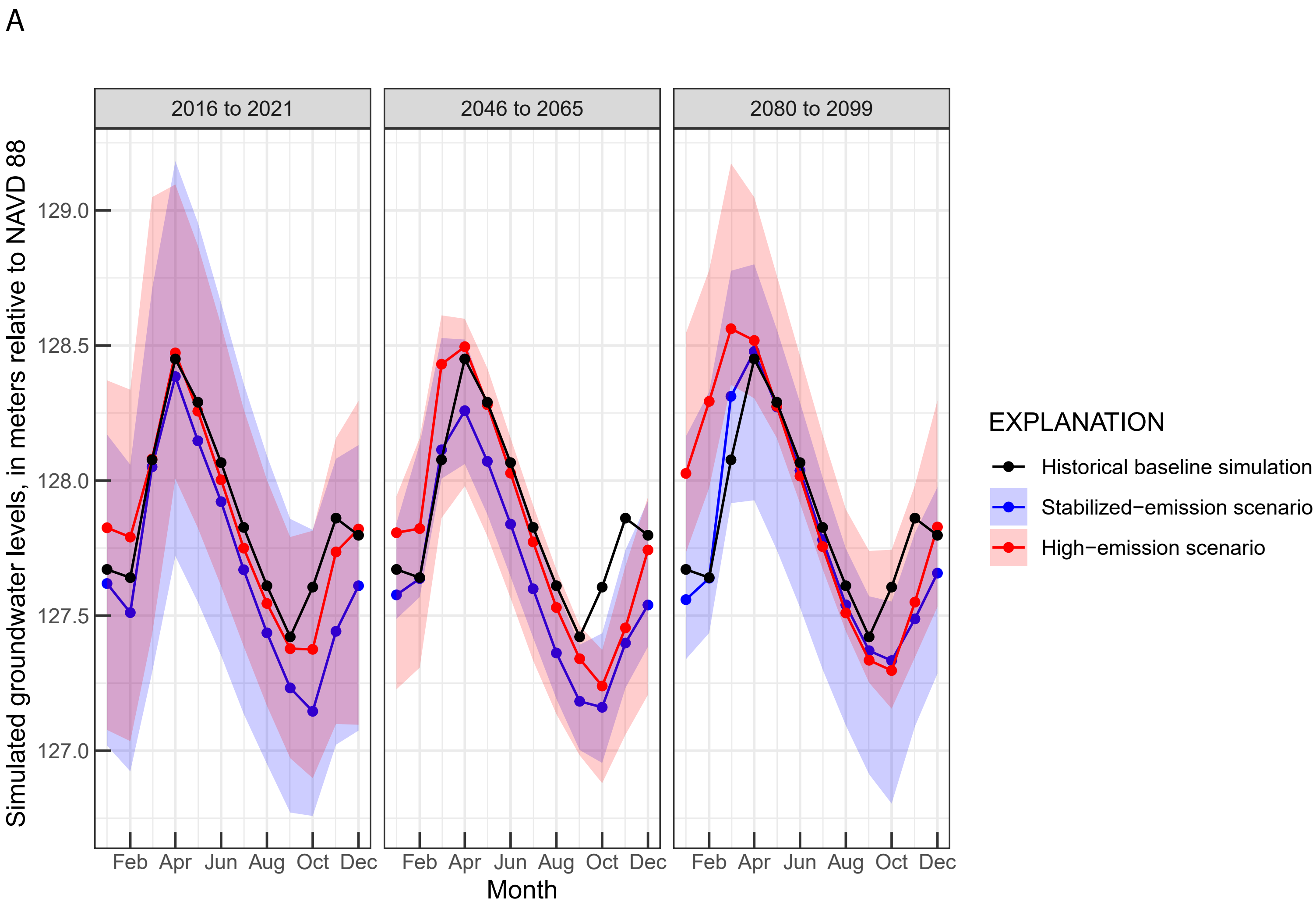


Figure 12. Simulated groundwater level elevations for each month at Round Pond (model layer 1) for the historical baseline conditions (2016 – 2021), and the median of five GCMs for stabilized and high-emission scenarios (2016 – 2021, 2046 – 2085, 2080 – 2099) for A. mean monthly of period conditions, and B. last year of a 3-year drought based on the lowest annual recharge in each General Circulation Model, emission scenario, and period. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models. [NAVD 88, North American Datum of 1988]

B

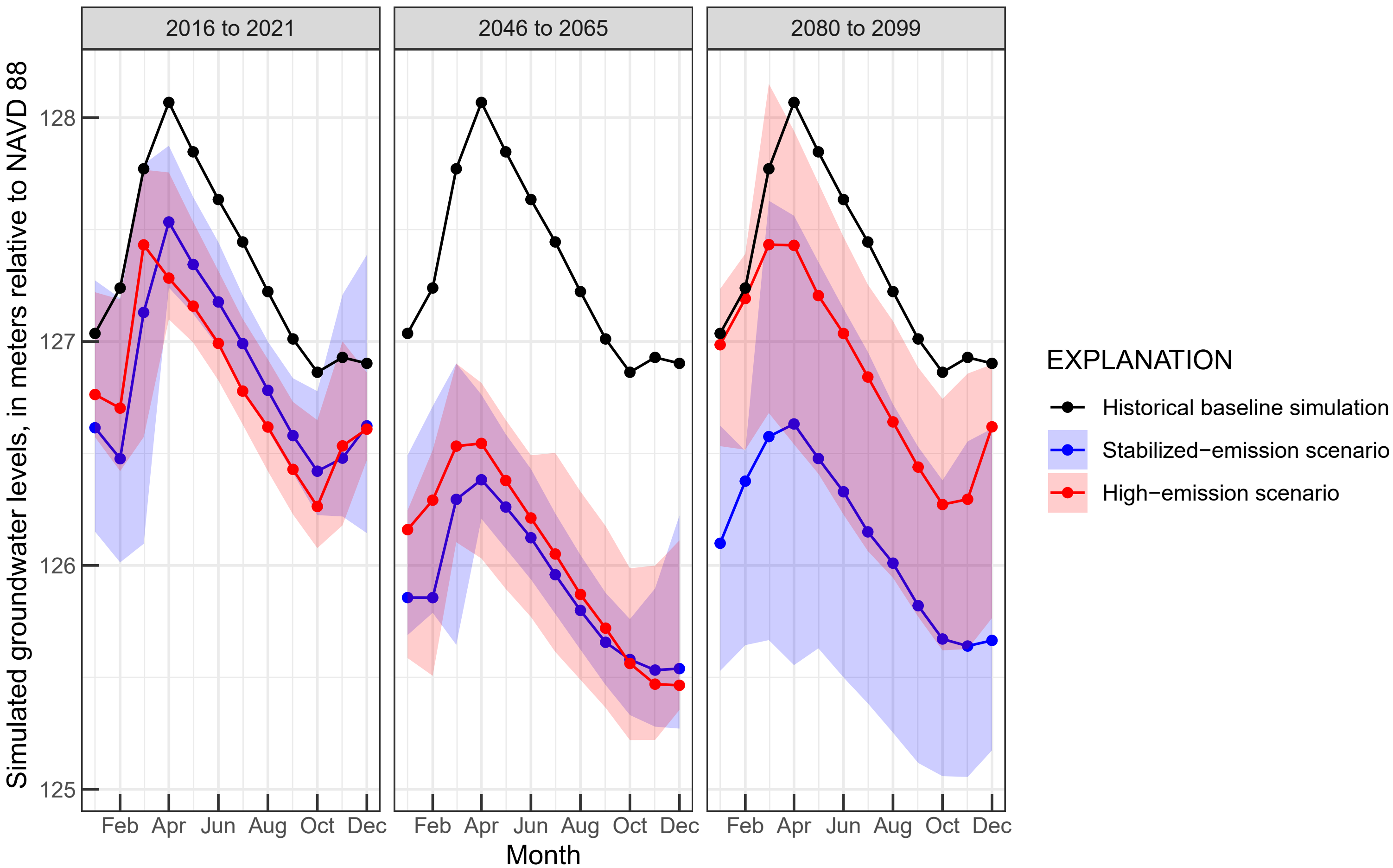
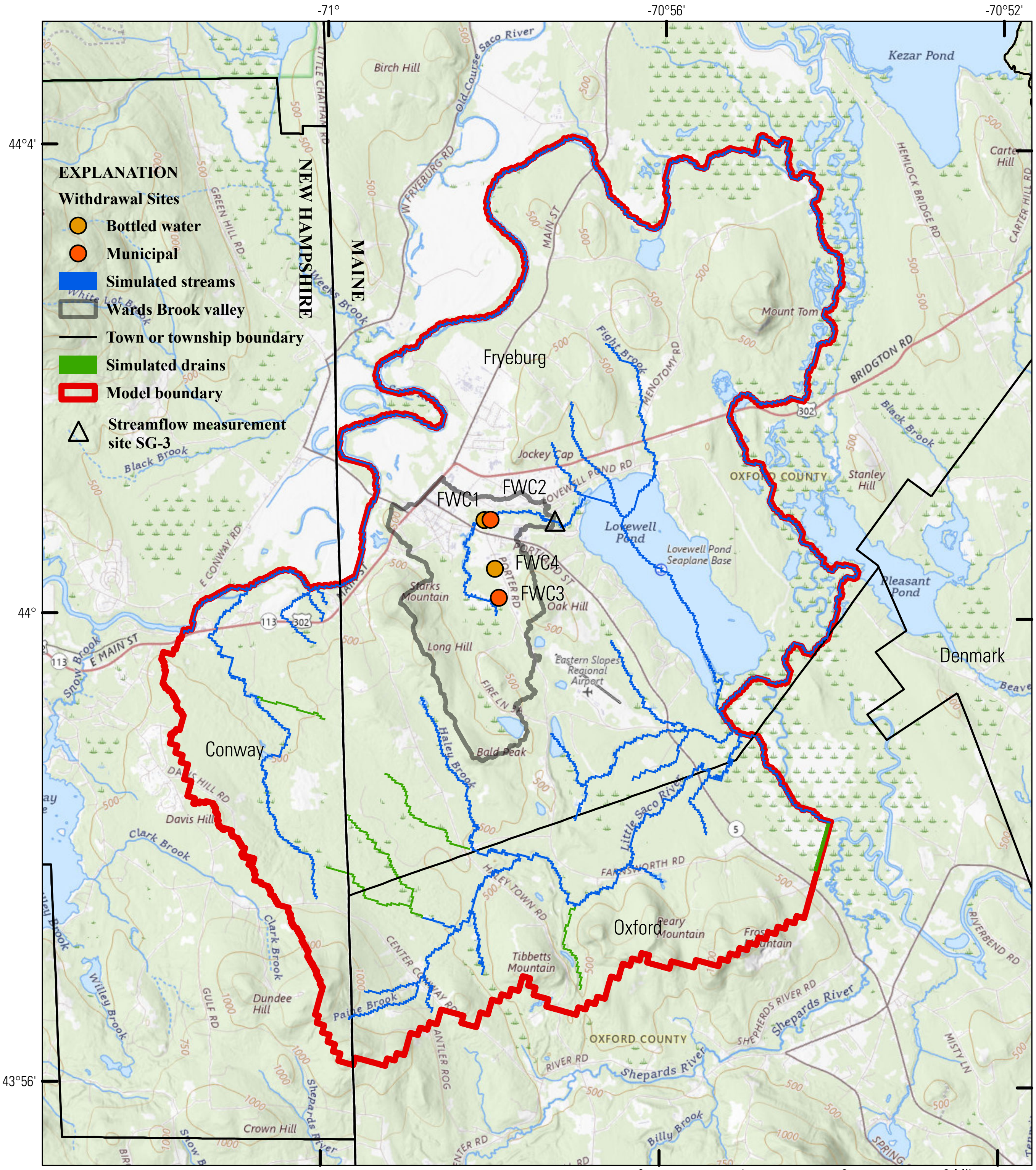


Figure 12. Simulated groundwater level elevations for each month at Round Pond (model layer 1) for the historical baseline conditions (2016 – 2021), and the median of five GCMs for stabilized and high-emission scenarios (2016 – 2021, 2046 – 2085, 2080 – 2099) for A. mean monthly of period conditions, and B. last year of a 3-year drought based on the lowest annual recharge in each General Circulation Model, emission scenario, and period. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models. [NAVD 88, North American Datum of 1988]



Base from U.S. Geological Survey and other Federal and State digital data, various scales.
New Hampshire State Plane FIPS 2800 projection
North American Datum of 1983

0 1 2 3 Miles
0 1 2 3 Kilometers

Figure 1-1. Groundwater-flow model boundary cells, Wards Brook valley and surrounding areas, New Hampshire and Maine. Streamflow site SG-3 measured by Luetje Geological Services LLC and McDonald Morrissey Associates (2021, 2022).

A

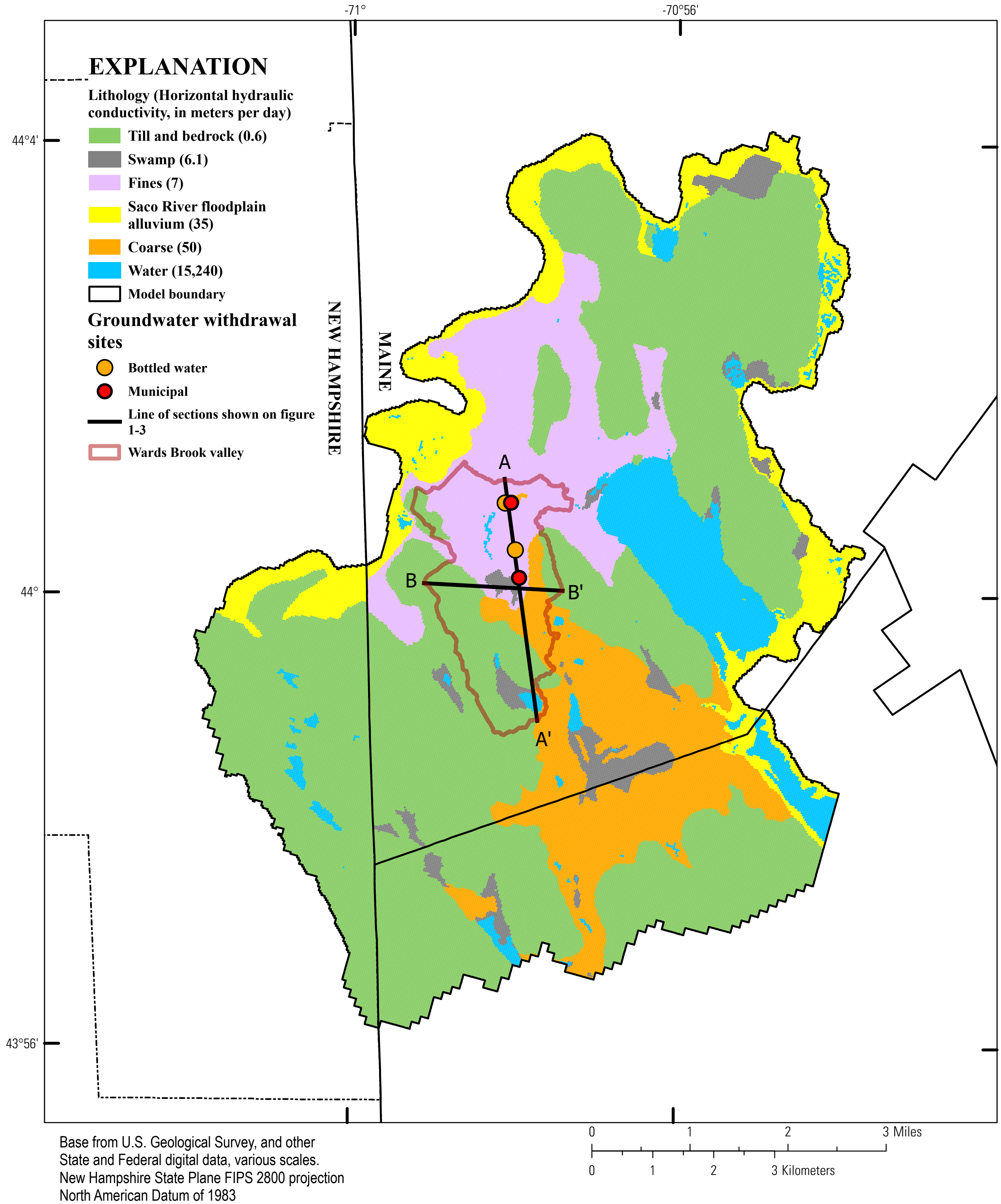
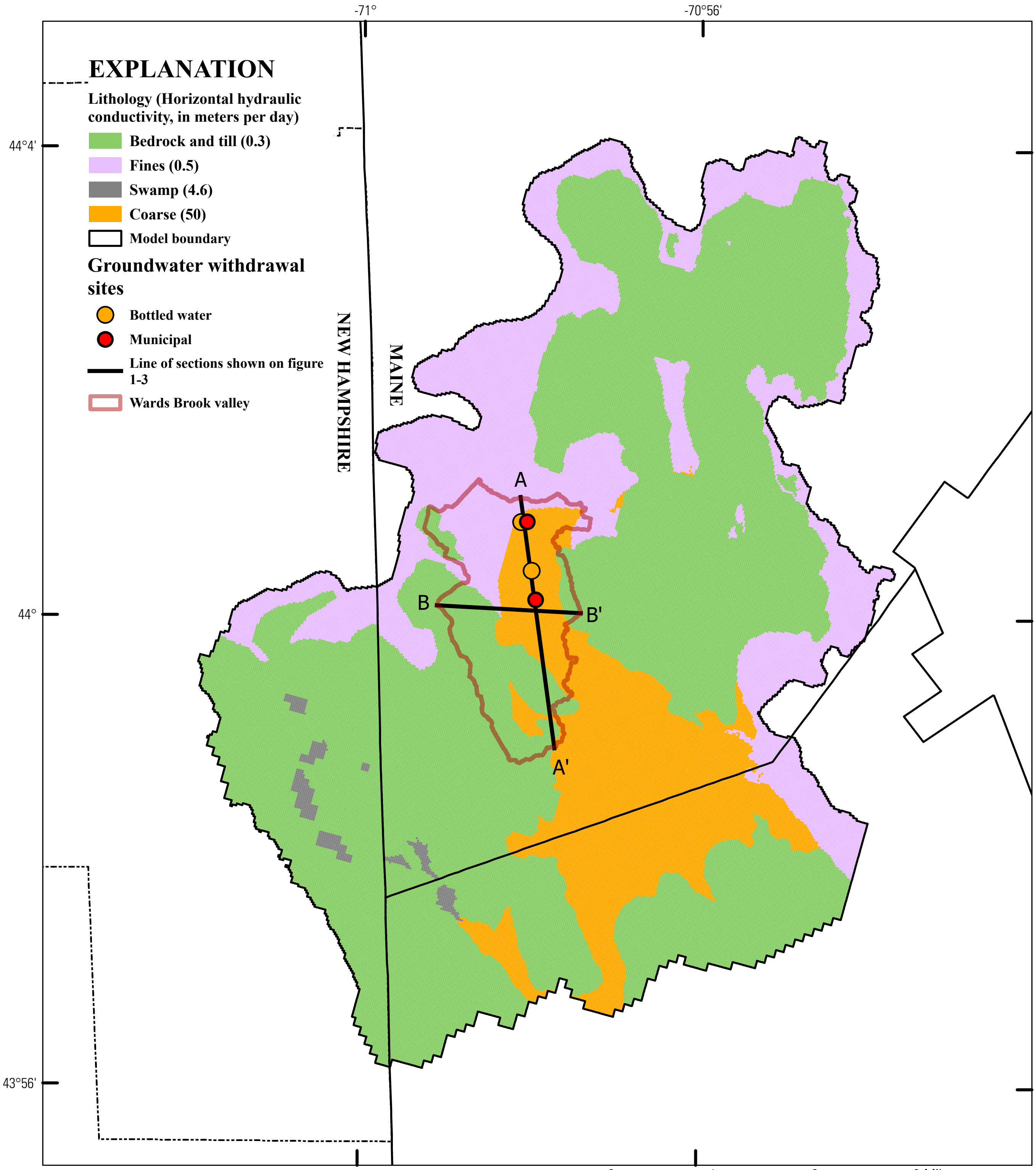


Figure 1-2. General lithology and horizontal hydraulic conductivity values used in final simulation in A. layer 1, and B. layer 2. Geology modified from Davis and Holland (1997a,b), Emery & Garrett Groundwater, Inc. (2005), and Thompson (2014).

B



Base from U.S. Geological Survey, and other State and Federal digital data, various scales.
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A

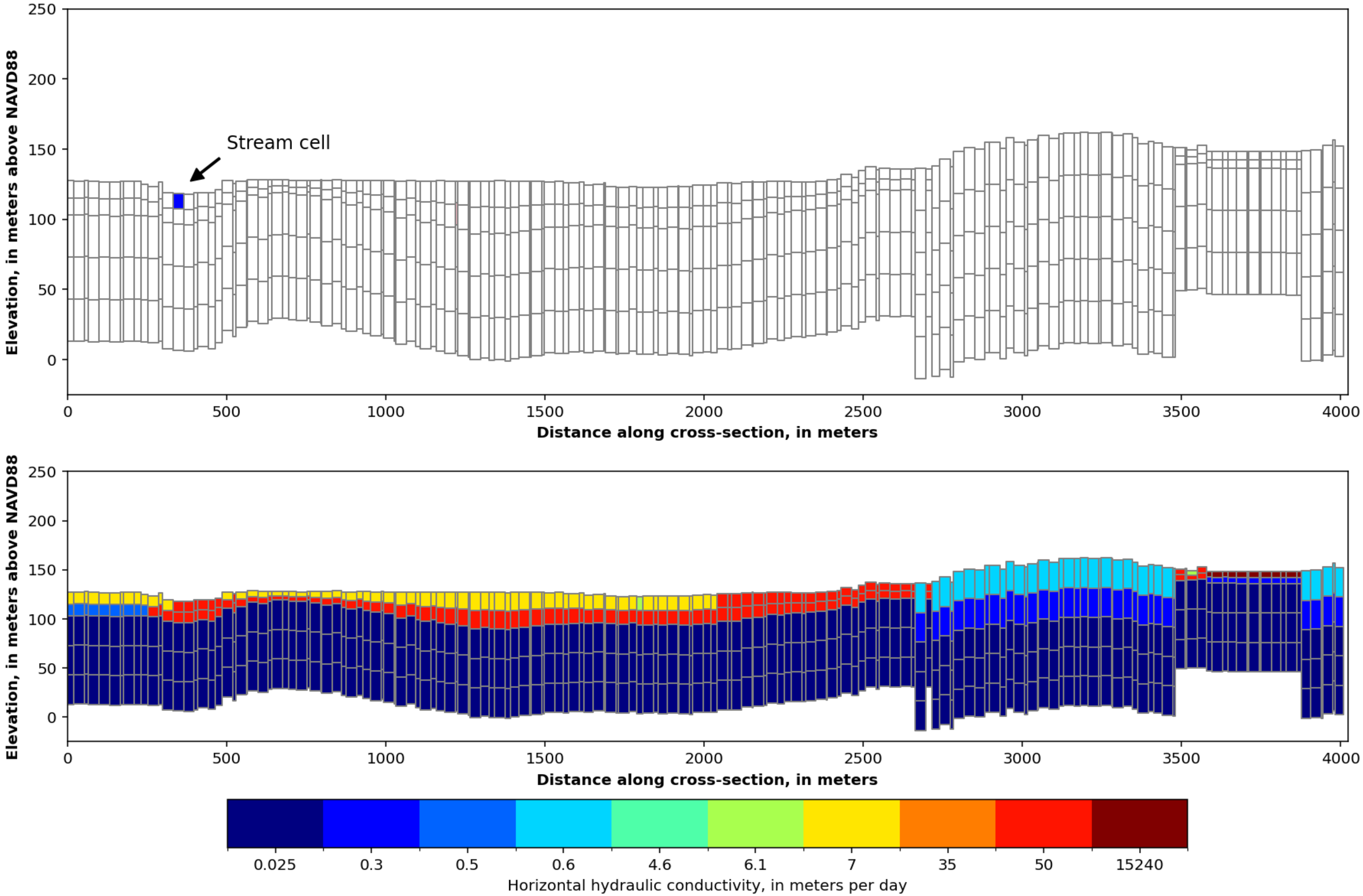
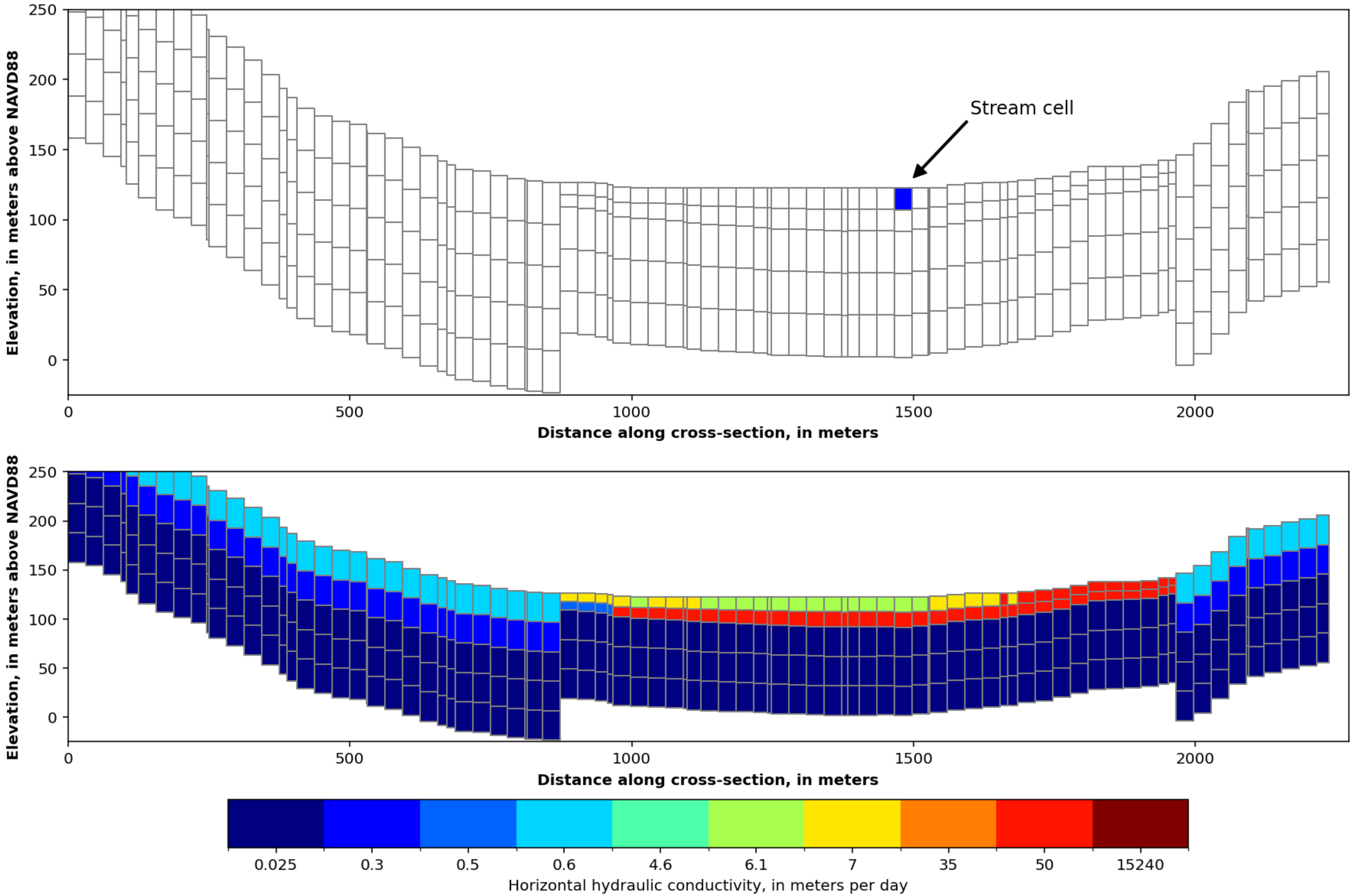


Figure 1-3. Vertical distribution of model grid cells, stream cells, and horizontal hydraulic conductivity for A. section line A-A′, and B. section line B-B′. Lines of cross sections are shown on figure 1-2.

B



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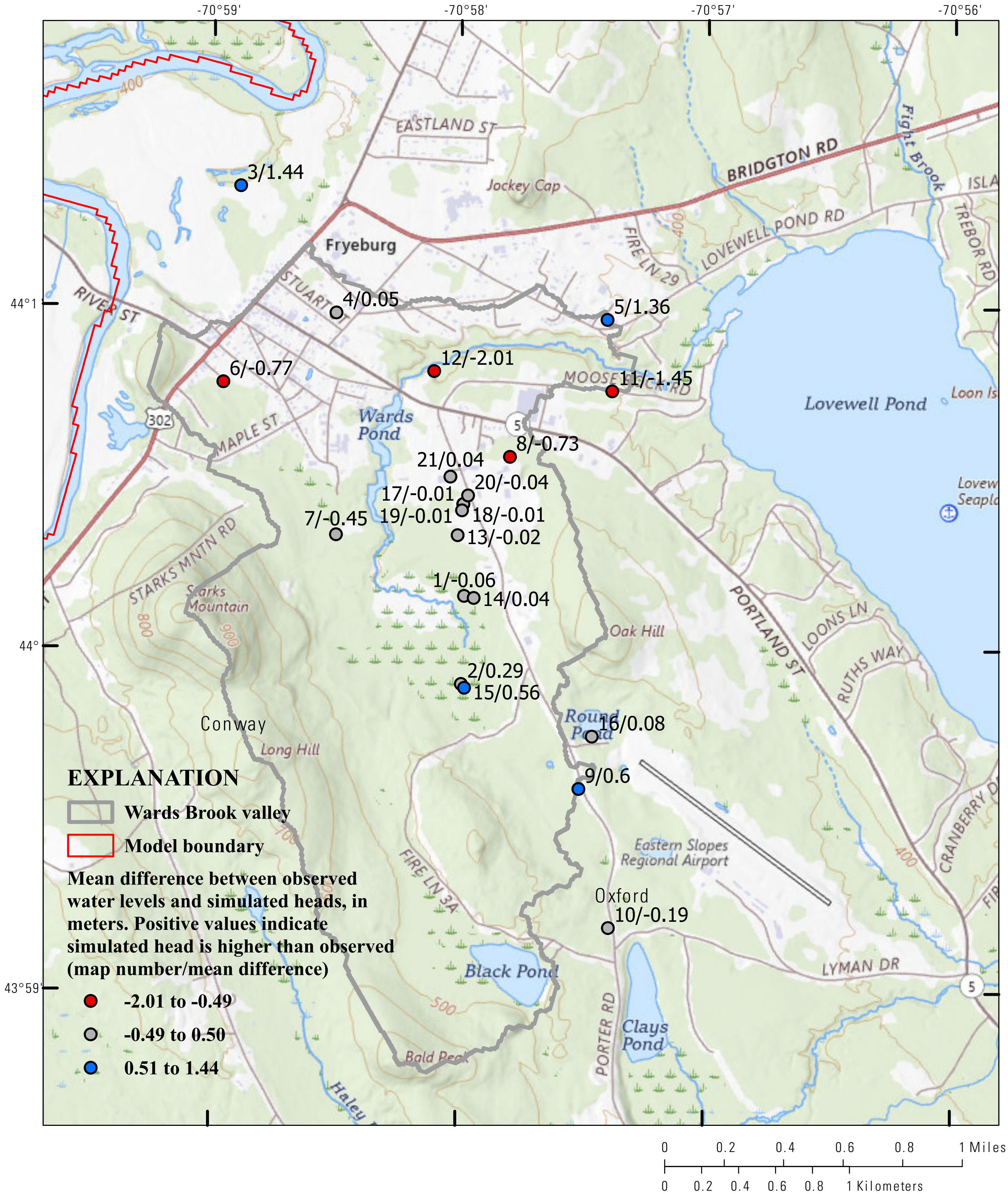


Figure 1-4. Map showing comparison of observed and simulated groundwater levels, Wards Brook model area, 2016 – 2021.

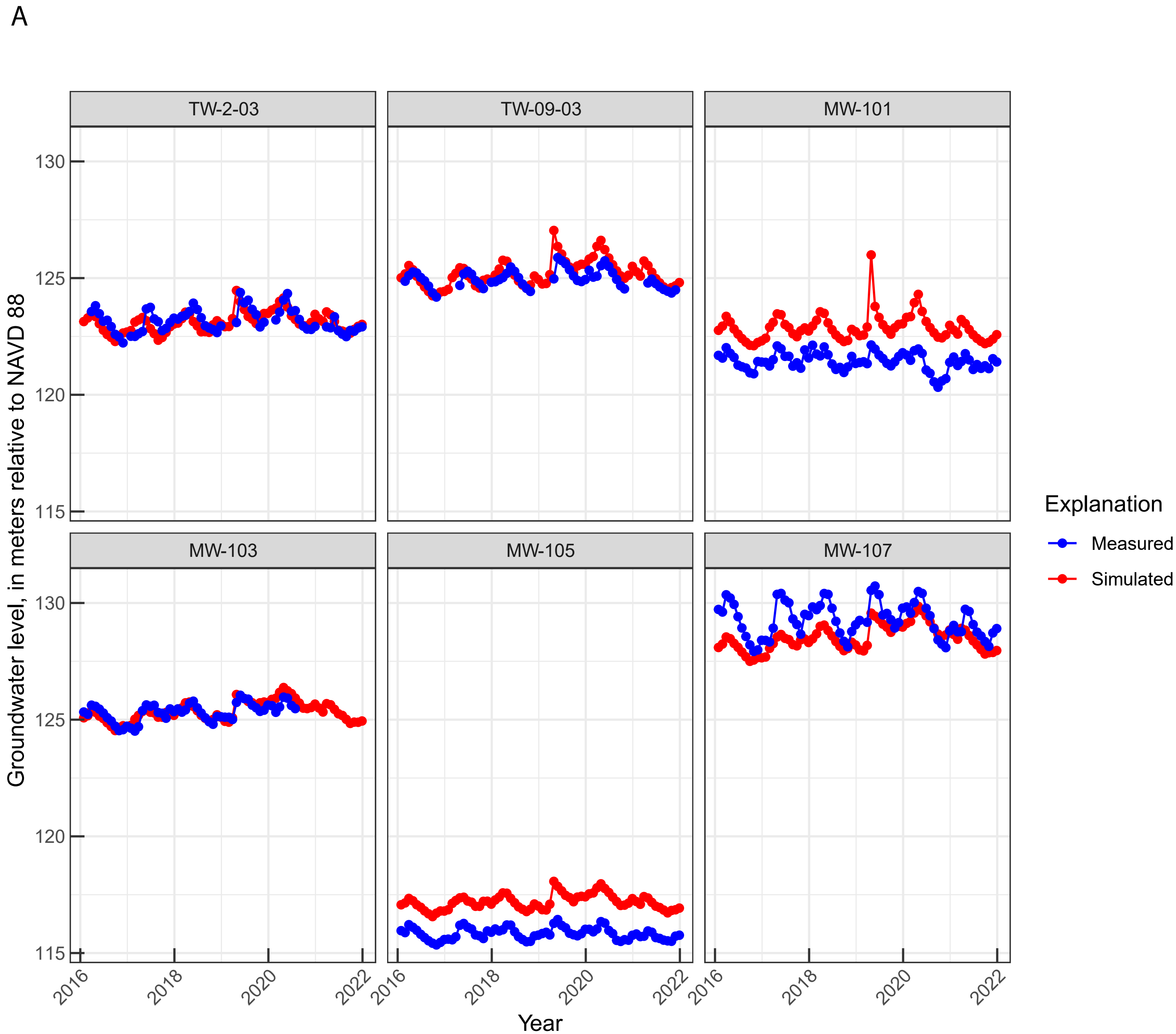


Figure 1-5. A-C, Graphs showing comparison of observed and simulated monthly groundwater levels, Wards Brook model area, 2016 – 2021. [m, meter; NAVD 88, North American Datum of 1988]

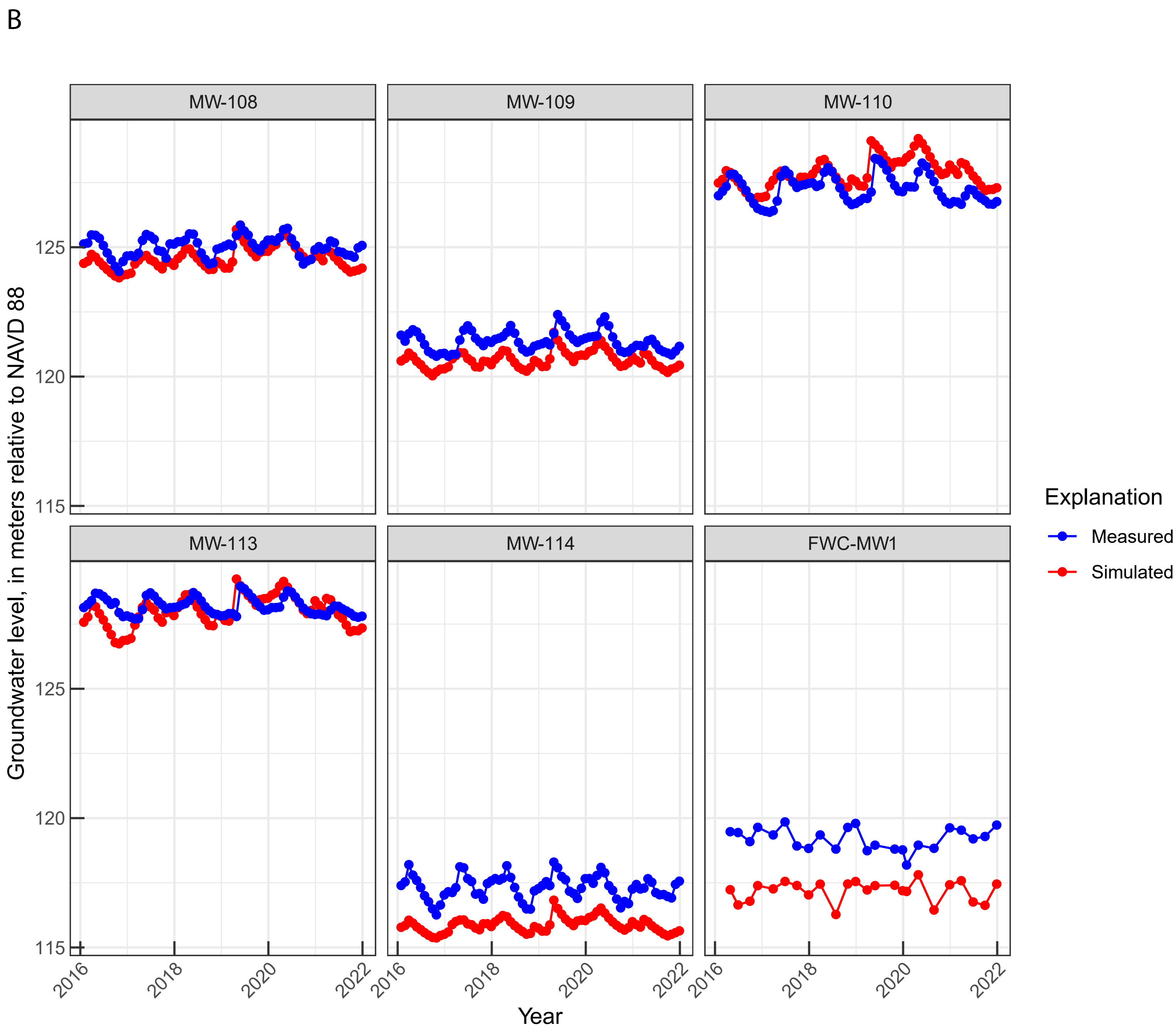


Figure 1-5. A-C, Graphs showing comparison of observed and simulated monthly groundwater levels, Wards Brook model area, 2016 – 2021. [m, meter; NAVD 88, North American Datum of 1988]

C

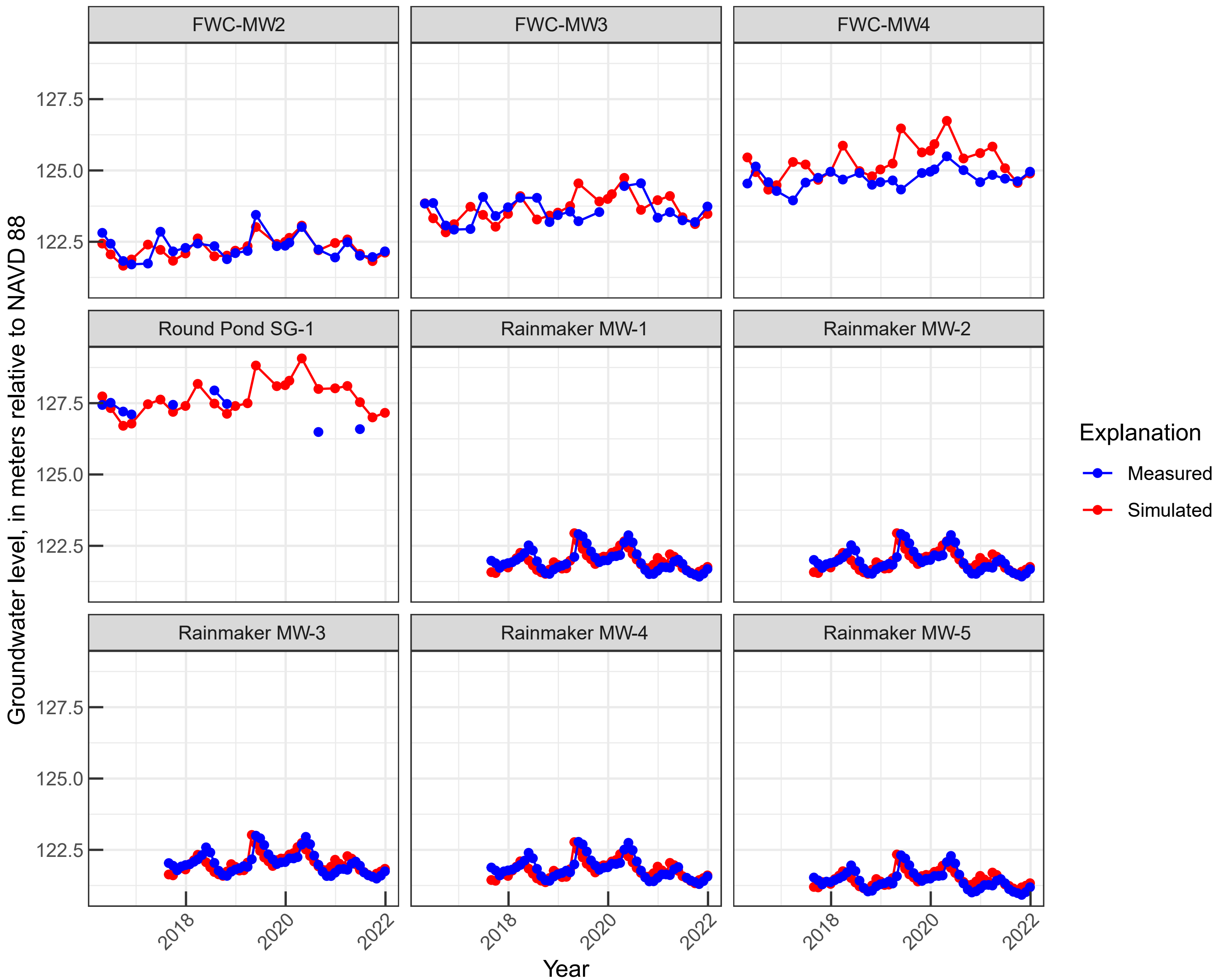


Figure 1-5. A-C, Graphs showing comparison of observed and simulated monthly groundwater levels, Wards Brook model area, 2016 – 2021. [m, meter; NAVD 88, North American Datum of 1988]

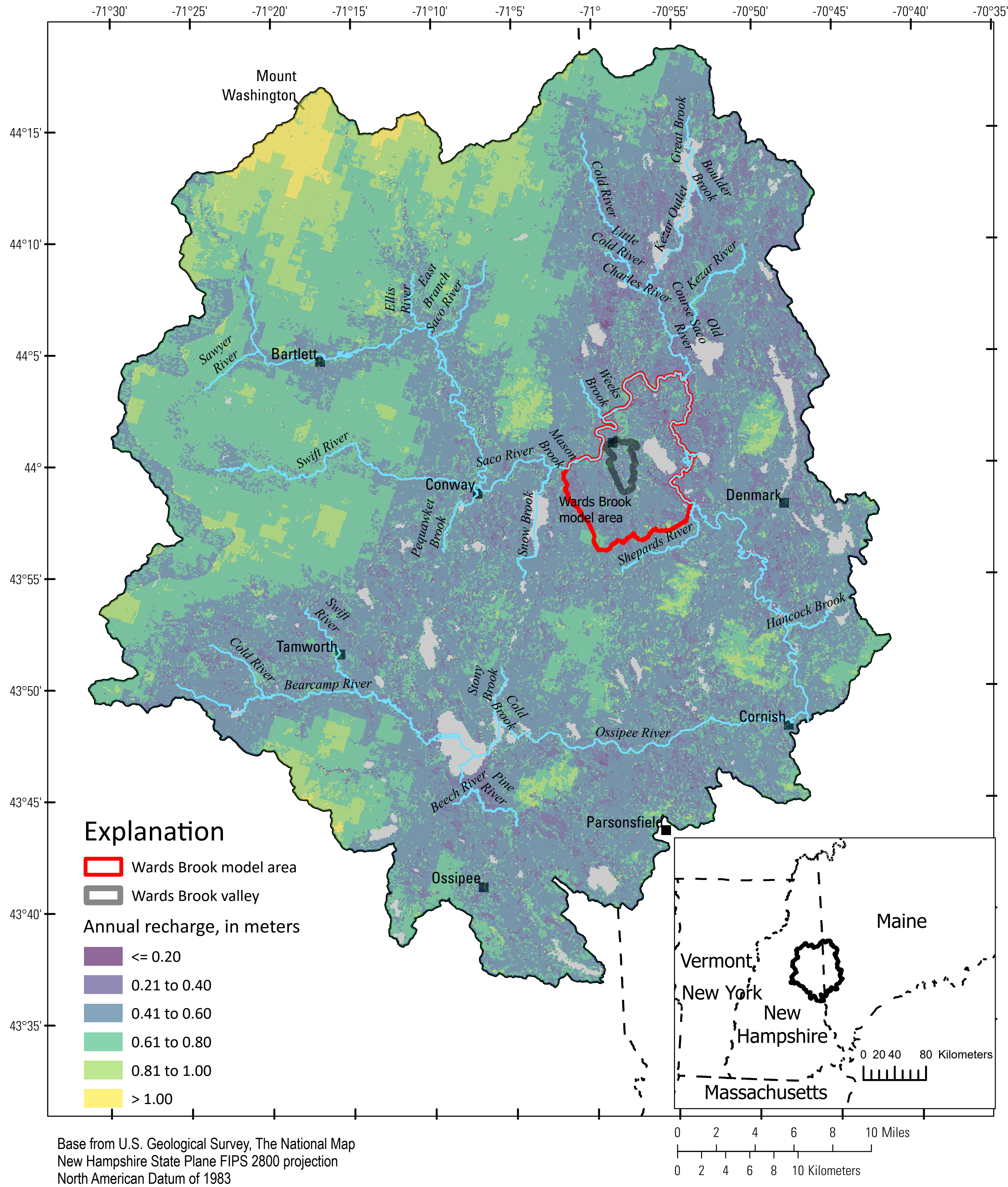
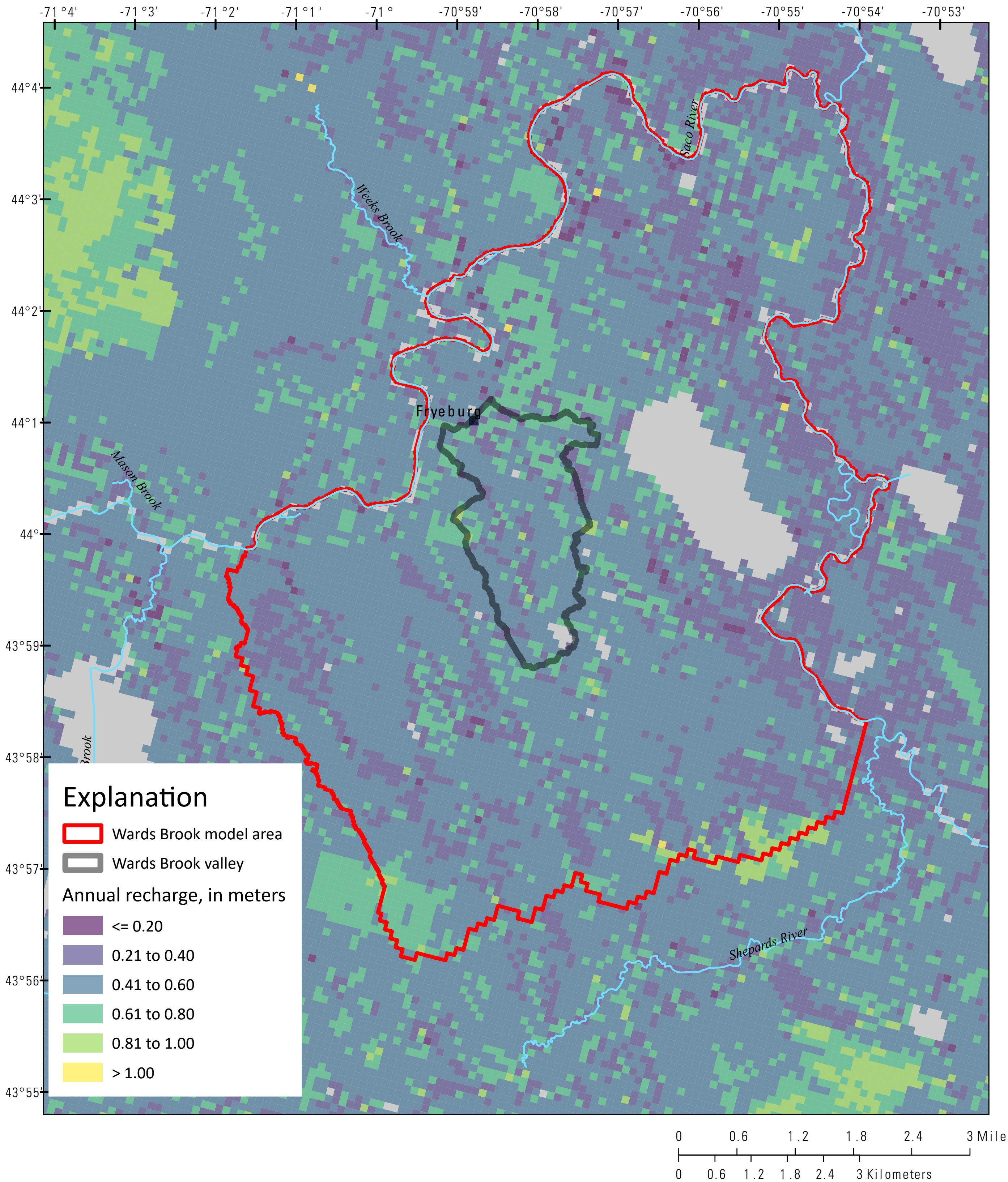


Figure 2-1. Mean annual recharge from the soil-water-balance model, Saco River headwaters, 2000 – 2021. Grey cells indicate "Open Water" cells that did not have an estimated recharge.



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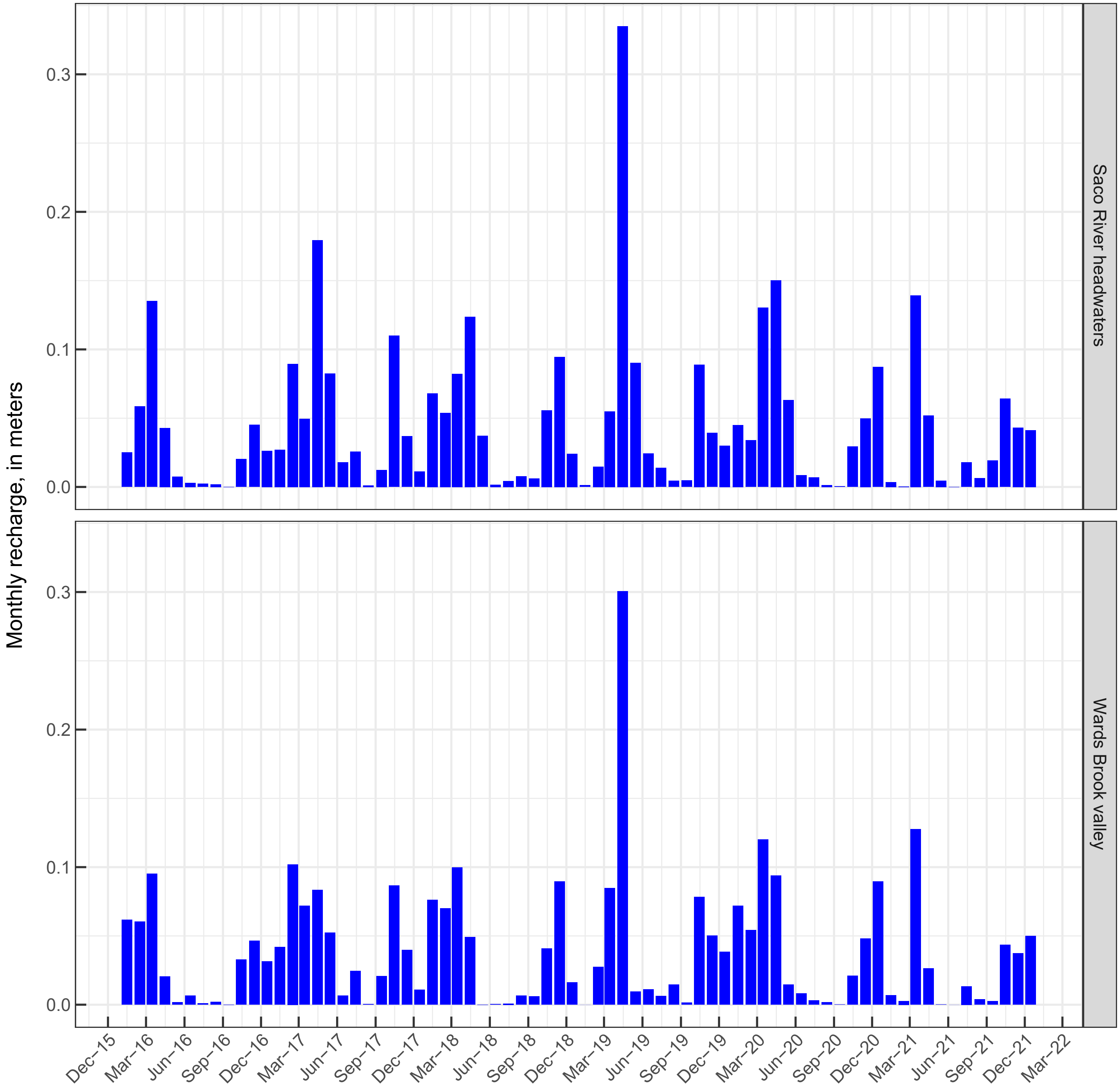


Figure 2-3. Monthly recharge estimated using the soil-water-balance model, Saco River headwaters and Wards Brook valley, 2016 – 2021.

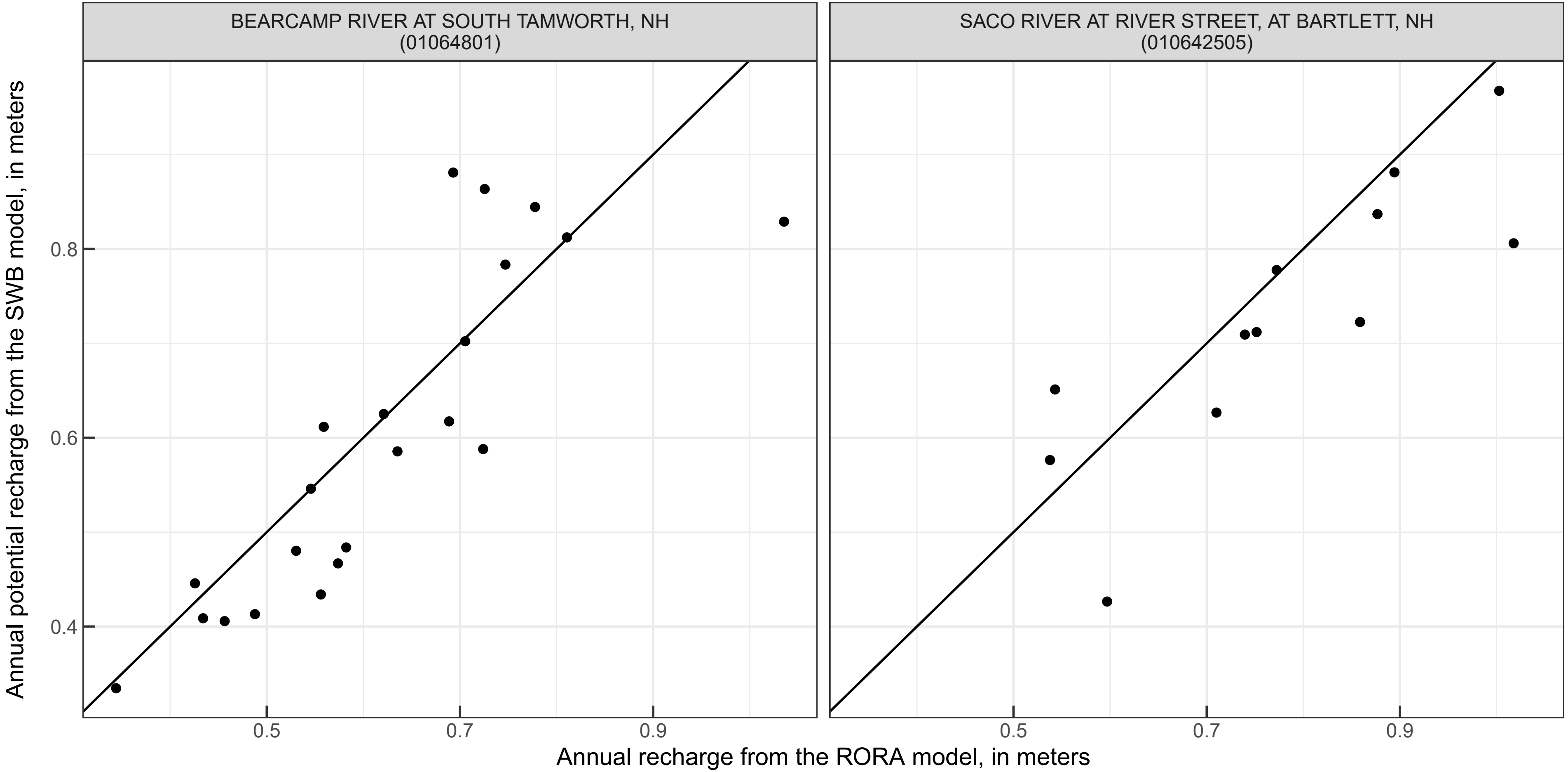


Figure 2-4. Comparison of mean annual net infiltration from the soil-water-balance model with estimates of annual recharge from analysis of streamflow data determined from the RORA model for two Saco River headwaters watersheds, 2000 – 2021, Saco River headwaters, New Hampshire (U.S. Geological Survey, 2022). [Station number on the top of the graphs are for the USGS streamgages and watersheds shown in fig. 1; SWB, soil-water-balance].

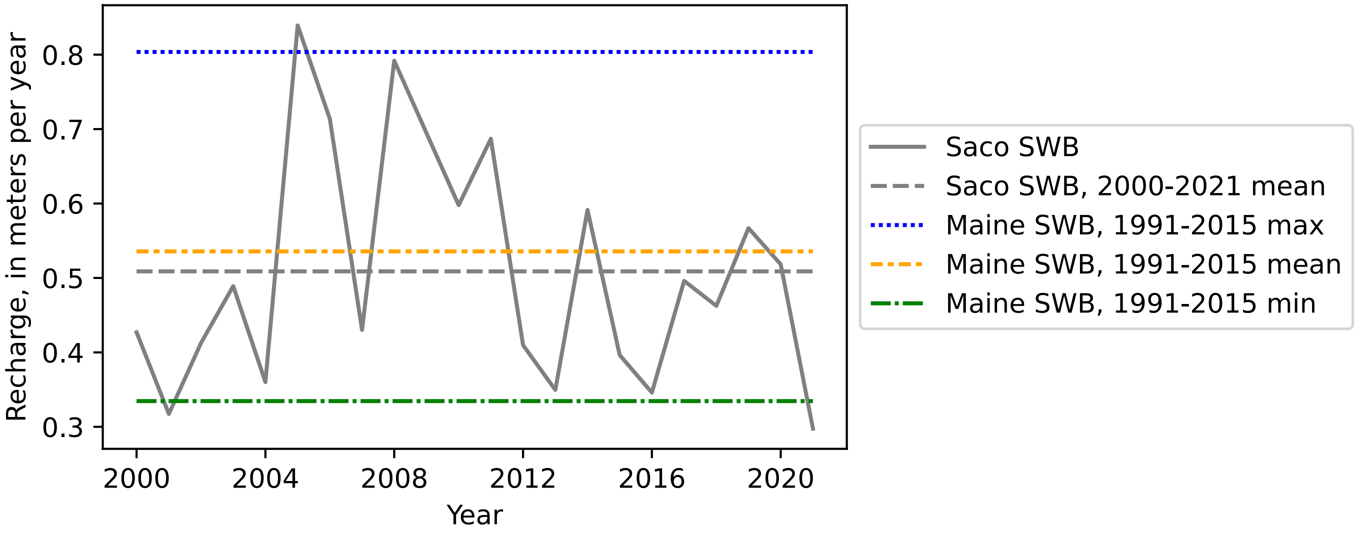


Figure 2-5. Comparison of Saco River headwaters soil-water-balance (SWB) model annual recharge for 2000 – 2021 with Maine SWB model recharge summary statistics for 1991 – 2015.

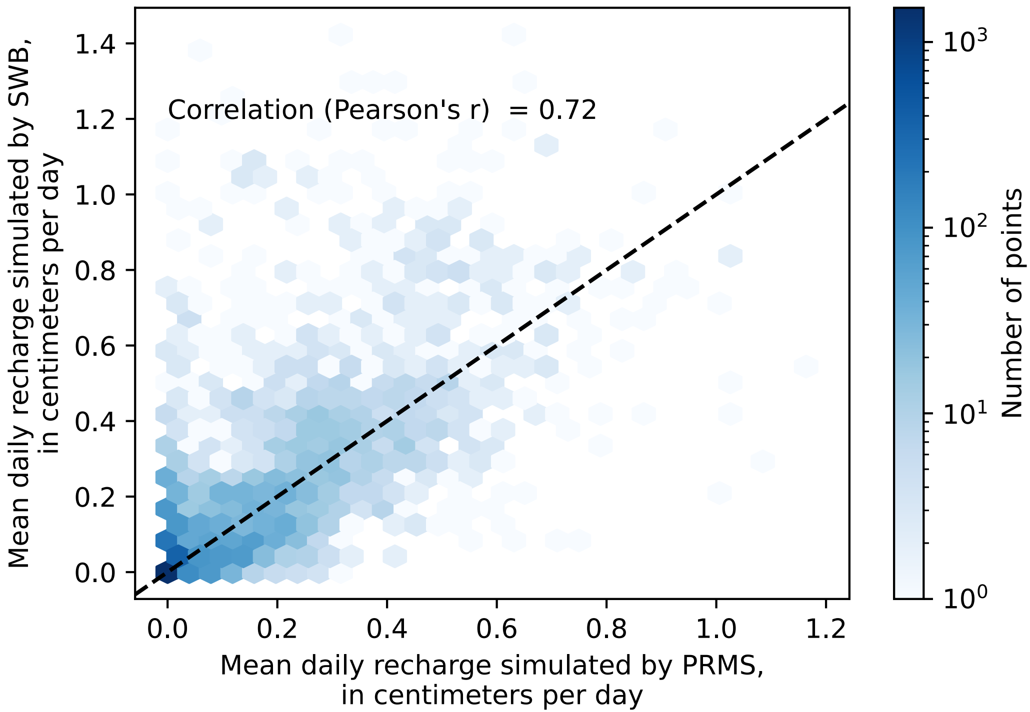


Figure 2-6. Comparison of soil-water-balance (SWB) model recharge results to Precipitation-Runoff Modeling System (PRMS) model recharge results summarized by hydrologic response unit for 2000 – 2005.

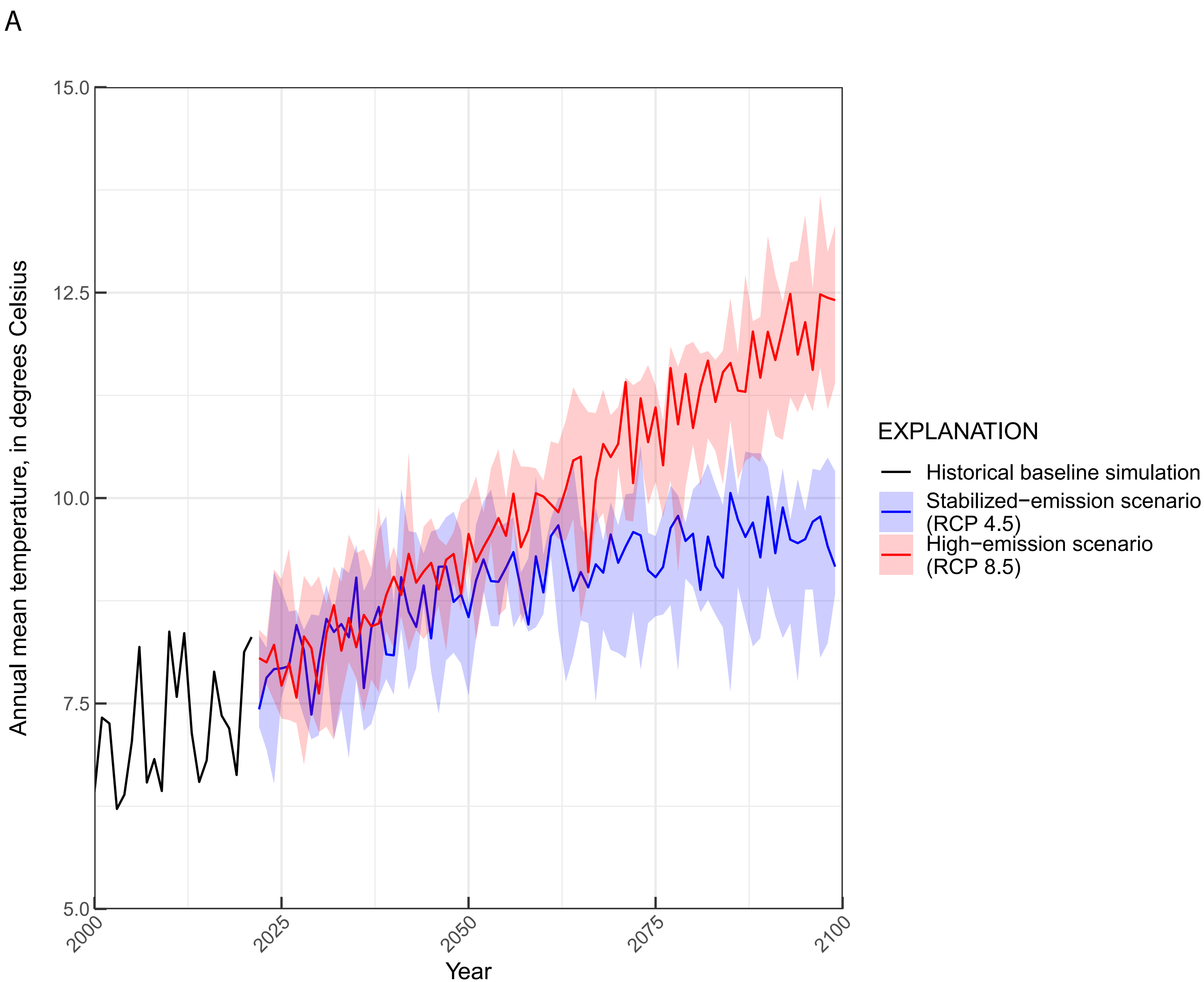
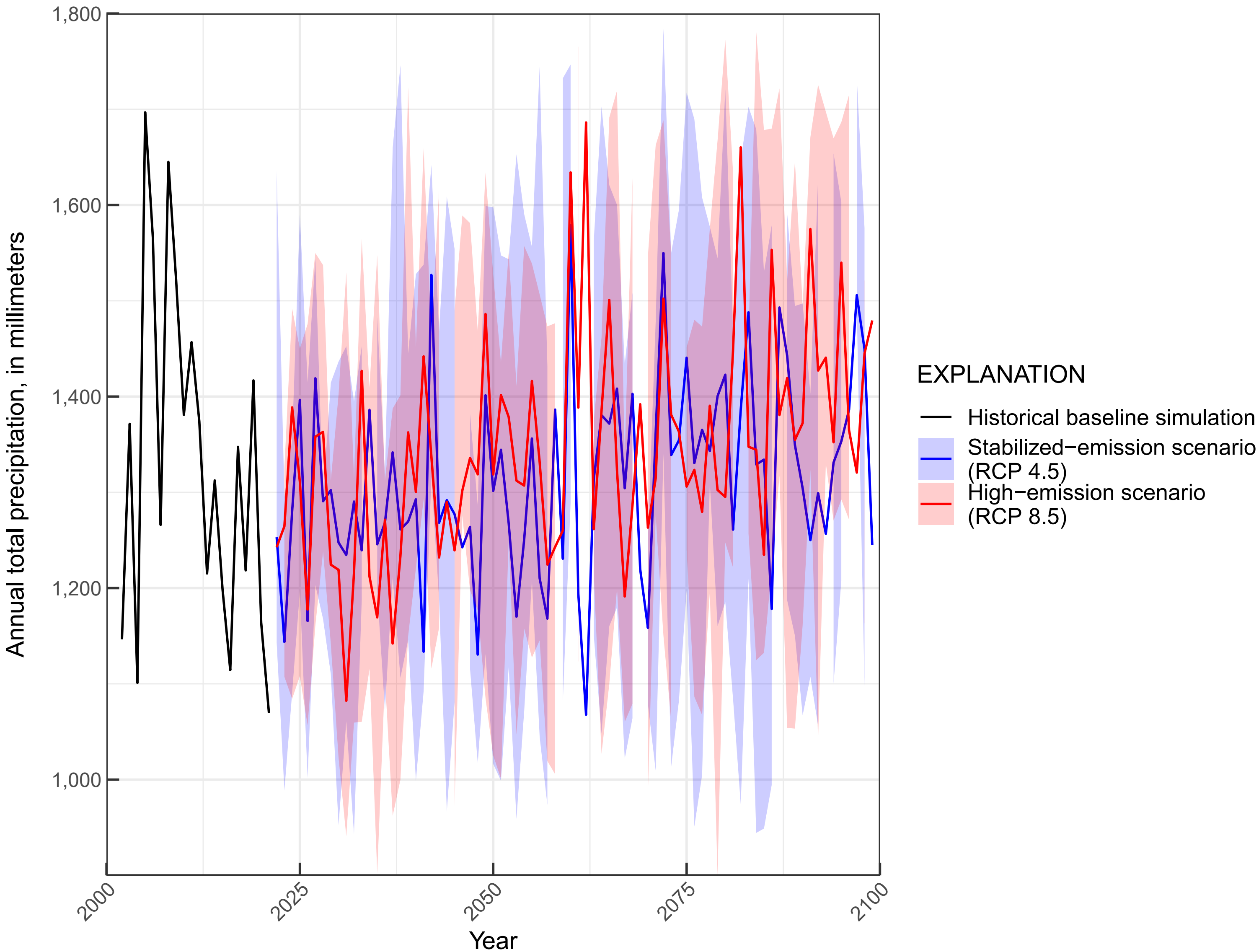


Figure 2-7. Summary of historical meteorological data (2000 – 2021) and two emission scenarios (2022 – 2100) for the groundwater-flow model area in the Saco River headwaters, for A. annual mean temperature, and B. annual total precipitation. Results from the emission scenarios are shown as the range of minimum and maximum values (shaded) and the median values (line) of the five General Circulation Models [RCP, Representative Concentration Pathway].

B



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