PLATTE RIVER RECOVERY IMPLEMENTATION PROGRAM (PRRIP -or- PROGRAM)

DRAFT Wet Meadow Hydrology Report

Questions for the ISAC

Section numbers refer to DRAFT Wet Meadow Hydrology Report, January 2023

Hydroregime (Section 3)

- Is there a better way to summarize area-based statistics? The current method calculates relevant statistics for points and then interpolate to generate areas on which subsequent statistics are calculated. Another option would be to interpolate surface for each timeseries value and calculate statistics for each interpolated grid-cell.
- Previous studies on wet meadow hydroregimes have lacked datasets that incorporate high-spatiotemporal coverage. Is there anything obvious we can do with this data (i.e., hourly groundwater levels) that we haven't already to learn something about wet meadow hydroregimes? Statistically or otherwise?
- Is there anything in this section that seems novel enough for publication?

Vegetation-Groundwater Links (Section 4)

- This section uses results from a previous study (Henszey et al., 2004) to evaluate hydrology and vegetation at two Program managed study sites. Are there any aspects of methodology that could be improved?
- Does this seem like a useful management screening tool? If so, what would be a way to package it? White paper?

Modeling (Section 5)

- General note: The model described in this section is most useful for deriving calibrated hydraulic parameters that can be used to make predictions. Calibration results include a series of calibrated K and S values that can be input for predictions about how stage changes will affect groundwater levels. It probably shouldn't be used to predict groundwater response to precipitation because it lacks even basic accounting for surface drainage or variably saturated flow, though model results through time show decent fits. If this were to be published, it would require more detailed calibration information and testing.

River-Ground surface elevation analysis (Section 6)

- This method came about from intuition that the elevation of the river surface relative to wet meadow topography was an important control for water levels.
- Is there a way to quantify relationships between elevation difference and L7th surface rasters? Perhaps a cell-by-cell regression?
- Is there another standalone analysis that we could perform with the data that could be used to evaluate river-ground surface relationships with respect to wet meadows? Perhaps a simple method that could go hand in hand with this one to support / verify the general findings.

2023 Wet Meadow Hydrology Study

DRAFT REPORT

January 2023

Prepared by Kristen Cognac

Executive Director's Office

Platte River Recovery Implementation Program

Fort Collins, CO

Executive Summary

This report provides a summary of data and analysis conducted as part of the Platte River Recovery Implementation Program (PRRIP or the Program) Wet Meadow Hydrologic Study. The study was designed to utilize hydrologic and climatological data that was collected at two Program managed wet meadow study sites (Fox and Binfield) between 2013 and 2021. Study objectives were to improve the understanding of wet meadow hydrology using the 8-year dataset (i.e., contribute to a growing body of literature), to quantify relationships between processes dominating wet meadow hydrology, and to develop tools and methods that inform management and restoration of wet meadow sites throughout the Plate River Valley. Pending feedback regarding the significance and relevance of methods and results presented herein, the intent is to prepare and submit a manuscript for publication that incorporates parts of this study.

The main narrative of this report is broken into 7 sections: 1) introduction and background, 2) study area and data collection, 3) quantifying the hydroregime, 4) groundwater-vegetation links, 5) modeling, and 6) river-floodplain elevation analysis. Sections 3 through 6 are structured as stand-alone scientific reports containing methods, results, and discussion. In Section 3, we quantify the hydroregime at two wet meadow sites using data with high spatiotemporal coverage to capture variability. Results reveal non-normal depth to groundwater distributions and a broad range of hydrologic conditions both within and across sites. Section 4 presents a method for predicting vegetation type based on depth to groundwater statistics largely following results of Henszey et al. (2004). By linking vegetation to hydrology, we demonstrate how vegetation targets may be converted to hydrologic management targets and we quantify the changes in hydrology that would need to occur at the study site to support a prevalence of wetland vegetation. Section 5 introduces a simple analytical model that can be used to predict groundwater levels at wet meadow sites based on stage change, precipitation, and ET. We then apply that model to testing two management scenarios for achieving target hydrologic conditions. In Section 6, we present a method that utilizes river and ground surface elevation differencing to make predictions about shallow sub-surface hydrology at wet meadow sites, even where data are absent. Section 7 includes a list of references. Appendices are located after the main narrative.

For detailed information about wet meadows and their associated biological and abiotic characteristics, the reader is referred to Weir and Chavez Ramirez (2011). For a summary of data collection methods and instruments deployed at wet meadow study sites, please see PRRIP (2015)

Contents

List of Figures

List of Tables

1 Introduction and Background

Wet meadows are ephemeral wetlands that lack standing water for most of the year but become inundated during periods of seasonal high water, typically during winter and spring months (Mitch and Gosselink, 2000). Inundation is typically controlled by dynamic linkages between river stage, precipitation, and ET which results in fluctuating shallow groundwater tables that extend above the land surface, particularly in low-lying swales (Brinley Buckley et al., 2021). Shallow groundwater is also essential for maintaining saturated soils during low-water periods, achieving sustained periods of inundation, and supporting a variety of hydrophytic grasses, sedges, rushes, and wetland wildflowers (Library of Congress 2006, US EPA 2006; Henszey et al., 2004; Tiner, 2016, Brinley Buckley et al., 2021).

For the Platte River Recovery Implementation Program (Program or PRRIP), understanding the role of hydrology at wet meadow sites in the Central Platte River Valley (CPRV) has been a long-term goal under broader objectives to manage land and water resources to benefit four threatened and endangered species (Target Species) (Cooperative Agreement,1997). During the First Increment, fully one third of the lands (10,000 acres) to be acquired by the Program were set to be managed as wet meadow habitat. At that time, it was thought that flow releases to benefit wet meadows could consume a major fraction of the Environmental Account (EA) water budget (PRRIP, 2012). However, as water and land priorities shifted along with the growing understanding of physical and biological systems, water-use for wet meadows decreased in priority.

Despite the shift in priorities, the Program continues to recognize wet meadows as vital components of the CPRV. With highly fertile soil and wide-ranging hydrology and vegetation, wet meadows support rich and diverse ecosystems that provide habitat for over 150 species of birds and other wildlife (e.g., Krapu, 1981; Currier, 1994; Davis et al 2006, Meyer and Whiles 2008; Franke 2006). Over the past century, an estimated 75% of wet meadows have been lost or severely degraded due to human development (Currier et al., 1985; Sidle et al., 1989). Drastic changes in land-cover and hydrology have resulted from agricultural development, dams, diversions, and wells. It is estimated that the annual mean and peak discharge on the Platte River has been reduced by up to 70% since pre-development times (Williams 1978, Eschner et al., 1981, Davis et al., 2006; USBR, 2004). Accompanying changes in channel width and vegetation have profoundly impacted riparian hydrology, including the lowering of riparian groundwater tables that are critical for supporting wet meadows (Hurr 1983, Currier and Ziewitz 1986, Wesche et al. 1994). With climate change expected to increase the severity of droughts over the coming decades (Williams et al., 2015), further impacts to wet meadows can be expected (e.g., Joyce et al., 2016).

Today, the Program manages over 5,000 acres of low-land grassland, with some fraction being wet meadows. Management actions include rotations of seeding, livestock grazing, haying, mowing, and prescribed fires. At least one Program-managed site (Fox) underwent topographic recontouring to bring the land surface closer to the groundwater table to restore hydrologic conditions. Although Program EA water isn’t currently applied in direct benefit to wet meadows, hydrologic management actions certainly affect wet meadows throughout the CPRV due to the critical role that hydrology plays in supporting various biological and ecological processes.

Unfortunately, limited management guidelines exist for hydrologic requirements at wet meadow sites. Several authors have characterized hydrology and its controls and restoration efforts at CPRV wet meadow sites (e.g., Hurr, 1983; Wesche et al., 1994; Currier, 1995; Whiles and Goldowitz, 1998; Brinley Buckley et al., 2021). However, both scientific and restoration results have varied substantially across studies, limiting the utility of results decision-making. To this end, the Program has invested almost 10 years in collecting data and developing a hydrologic study to improve the understanding of hydrology at wet meadow sites to support management. This report summarizes results from data analysis and modeling performed improve the characterization and understanding of CPRV wet meadow hydrology. Methods presented herein will hopefully provide utility for managers of other wet meadow sites throughout the CPRV.

2 Study Sites and Monitoring Networks

The study area includes two wet meadow tracts located on vegetated islands along the Platte River central Nebraska, USA (Figure 1). Both study sites are located on vegetated islands bounded by primary (south) and secondary (north) channels of the Platte and are managed by the Platte River Recovery Implementation Program (https://platteriverprogram.org/) as part of a broader effort to restore and protect habitat for Program Target Species. The Binfield Site (i.e., Binfield) is a natural, non-degraded wet meadow tract that is currently managed through grazing, haying, controlled burns, and rest periods. The Fox Site (i.e., Fox) is considered a degraded wet meadow tract as it was formerly converted to cropland and has since been restored to a wet meadow. The Fox Site is managed through seeding (2012 and 2017), grazing, haying, controlled burns, and rest periods. Topographic re-contouring was also performed in 2014 to improve hydrologic conditions and likelihood of inundation at the site. This involved excavating five elongate depressions to depths that were determined to result in seasonal standing water. Despite management actions at Fox, vegetation field surveys document decreased species diversity through time (2014 to 2022) by about half, low floristic quality index (FQI) values (2022: Fox = 16; Binfield = 31), and a slowly decreasing ratio of exotic to native forb species that is nearing 1:1.

Monitoring networks at each study site included 16 shallow observation wells (3-meter depths), 2 river stage gages, and a weather station that records precipitation, temperature, relative humidity, solar radiation, wind speed, and other meteorological parameters necessary for estimating reference evapotranspiration (ET). Wells and stage gages were equipped with vented pressure transducers (accuracy ±0.35 cm / resolution 0.0175 cm) programmed to log continuous hourly groundwater and surface water levels. Manual water level measurements were made periodically to calibrate transducers and correct for sensor drift. Transducer data were downloaded regularly via a telemetry system.

Stage gages were anchored to the streambed using existing structures or steel rods that were manually driven into the streambed. Significant data gaps existed at all stage sites due to difficulties with stabilizing sensors in the dynamic braided channel. To fill gaps, linear regressions were developed for study-site stage gages and nearby USGS maintained stage gages. Resulting regression equations were used to estimate river stage at study sites during gap periods. Methodology and regression equations are provided in the 2021 Wet Meadows Data QC Report.

Weather stations were maintained by the High Plains Regional Climate Center and meteorological data and method details are publicly available (https://hprcc.unl.edu/awdn/). HPRCC provides daily potential evapotranspiration (ET_0) values estimated using the daily Penman equation with the wind function for reference crop alfalfa. The final data collection period spanned from March 2013 to July 2021, spanning approximately 8 full growing seasons (2013-2020).

Within the study region, the Platte forms a wide, braided channel with shallow depths (typically < 1m), extensive sand bars, and grassland floodplains. Large, vegetated riparian islands are common. Individual wet meadow tracts were selected for this study based on site history, management status, and data availability. Groundwater flow is approximately parallel to the Platte River, from west to east. The riparian groundwater table is generally highest in the spring and lowest in the summer to early fall (Hurr, 1983; Wesche et al., 1994). Well drilling logs indicate that shallow sediments consist primarily of well-graded sand with occasional gravel and small amounts of silty sand near the surface. The CPRV is characterized by a continental semiarid climate. Between 2013 and 2021, annual rainfall ranged from 36 to 79 cm, with an average of 54 cm per year according to weather stations at the study sites. The majority of precipitation falls between April and September, which largely overlaps with the typical growing season which extends from mid-April to mid-October (HPRCC). During the study period, the largest precipitation events occurred during June and September, with other significant events in April and early-October. The two wettest years for rain and snow were 2018 and 2019, though 2015, 2016 and 2019 recorded the greatest annual discharge at the nearby Kearney gaging station located approximately 5 miles to the west of Fox (USGS Station No. 06770200).

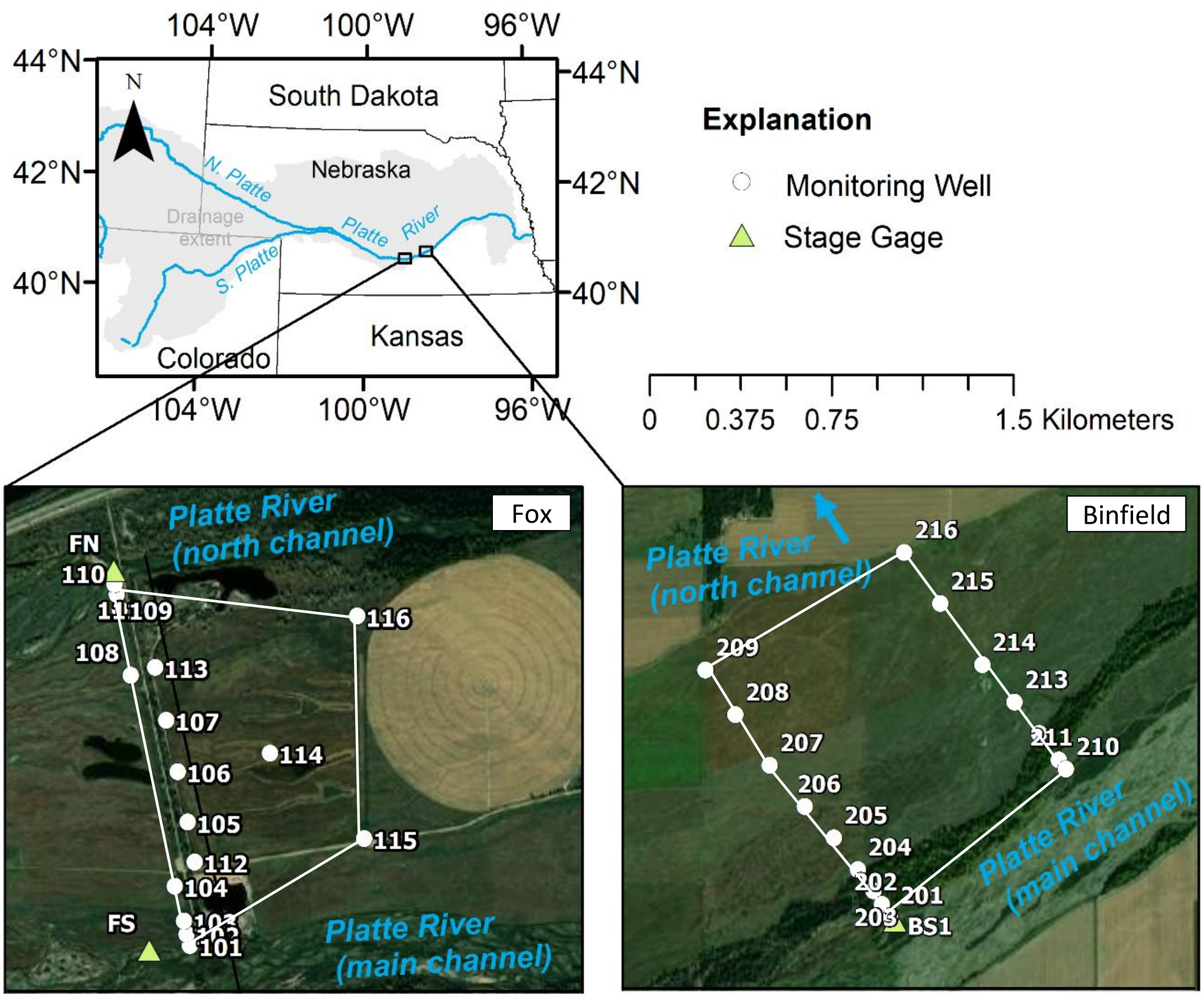


Figure 1 - Wet meadow study sites Fox (restored, formerly cropland) and Binfield (native) with associated monitoring networks.

3 Characterizing the Hydroregime

Wet meadows are characterized by significant spatiotemporal variability in soil moisture and groundwater depth. Temporal variability is imparted by large seasonal changes in groundwater level controlled by, in order of decreasing importance, river stage, precipitation, and ET (Wesche et al., 1994; Brinley Buckley et al., 2021, Chen, 2007). Spatial variations imparted largely by topographic gradients associated with ridge-swale topography in which alternating uplands (ridges) and lowlands (swale) oriented parallel to historic flow reflect relic braided river deposits. The undulating landscape overlies a relatively subdued (i.e., flatter) groundwater table, resulting in spatially varying groundwater depths and inundation dynamics. Since groundwater contributes to soil moisture, soil saturation, and inundation frequency, topographic variations also control which species of vegetation and biota are present

(Henszey et al., 2004; Davis, 2006). Wet meadows appear as a heterogeneous patchwork of vegetation and biota that are adapted to locally varying conditions in hydrology and soil (Henszey et al., 2004; Whiles and Goldowitz, 2001, 2005; Davis et al., 2006).

Characterizing hydrologic dynamics to support decision-making for management and restoration of wet meadow sites has been challenging. Wesche et al. (1994) utilized 4-year groundwater and climatological datasets to evaluate controls and relevant hydrologic metrics at wet meadow tracts. Using correlation analysis, they identified stage as having a dominant influence on groundwater levels, followed by precipitation. However, data and conclusions were limited by manual collection methods available at the time. Brinley-Buckley et al. (2021) used timelapse imagery and wavelet coherence to evaluate inundation frequency and its relationship to hydrologic variables. They similarly identified stage as the dominant control on groundwater levels and underscored the importance of shallow groundwater for inundation dynamics. However, time-lapse imagery captures only a small area fraction of wet meadow sites, limiting the coverage and representation of results.

An objective for this study was to develop a method for characterizing the hydrologic regime (i.e., hydroregime) at wet meadow sites that accounts for spatiotemporal variability by incorporating high-frequency and high-spatial coverage datasets. The Program's existing long-term dataset collected at the Fox and Binfield study sites provides a robust starting point for this objective. The 8+ year dataset contains high-frequency (hourly) groundwater measurements that span large areas of wet meadow tracts. We calculate depth-to-groundwater and hydroperiod metrics that are commonly used to summarize the hydroregime using high-frequency point-based groundwater elevation data, interpolated groundwater elevation surfaces, and high-resolution LIDAR elevation data. The result is a robust characterization of the hydrology at each site that accounts for broad spatiotemporal variability. Data are compared to results from other studies to generate conclusions about hydrology and management for wet meadow sites throughout the CPRV.

As noted, wet meadow groundwater elevations and depths are highly variable through time and space. Statistical metrics were used to summarize groundwater depths and surface inundation (i.e., the hydroregime) at the two study sites using hourly groundwater data from monitoring wells and LIDAR elevation data. LIDAR data was acquired using an airborne laser system with a mean areal resolution of 0.7m, vertical and horizontal accuracy of ≤9.2cm and ≤0.6m respectively, and final gridded areal resolution of 0.91m. Although point measurements are well distributed across study sites (Figure 1), point-based groundwater elevation results are expected to contain bias due to the high variability in ground surface elevation. Both point- and area-based statistics were calculated to account for these spatial variations across the site.

3.1 Depth-to-Groundwater

Point-based statistics, calculated for subsets of hourly depth-to-groundwater (DTGW) measurements at each monitoring well include: the minimum, maximum, mean, median, range, and standard deviation for each year and total for the study period. DTGW was calculated by subtracting measured groundwater elevation (GWE) from the surveyed ground surface elevation (GSE) at each well location.

Point data from well locations were then used to generate DTGW surfaces for which area-based statistics were calculated. First, the annual mean, median, maximum, and minimum groundwater elevation were calculated for each well and year during the study period (2013-2020). From the minimum and maximum values, a range was also calculated. Next, a groundwater elevation surface representing each annual statistic (i.e., the mean annual groundwater elevation) was generated by applying a linear spatial interpolation algorithm to estimate gridded values between wells (e.g., Figure 2). For all but the range, a corresponding depth-to-groundwater (DTGW) surface was then generated by subtracting the interpolated groundwater elevation surface from the LIDAR ground surface elevation resulting in 24 DTGW surfaces and 8 groundwater range surfaces for each site. The spatial resolution was set to maintain that of the LIDAR, totaling approximately, 1,250,000 grid cells per surface. Since area-based statistics include a grid of DTGW values, for each statistic surface, median of all grid cells was calculated for summary tables. As discussed in results, the median was selected due to the finding of non-normal DTGW distributions through both time and space. The final statistics are considered representative of the entire site throughout the study period, overcoming the location bias present in well data.

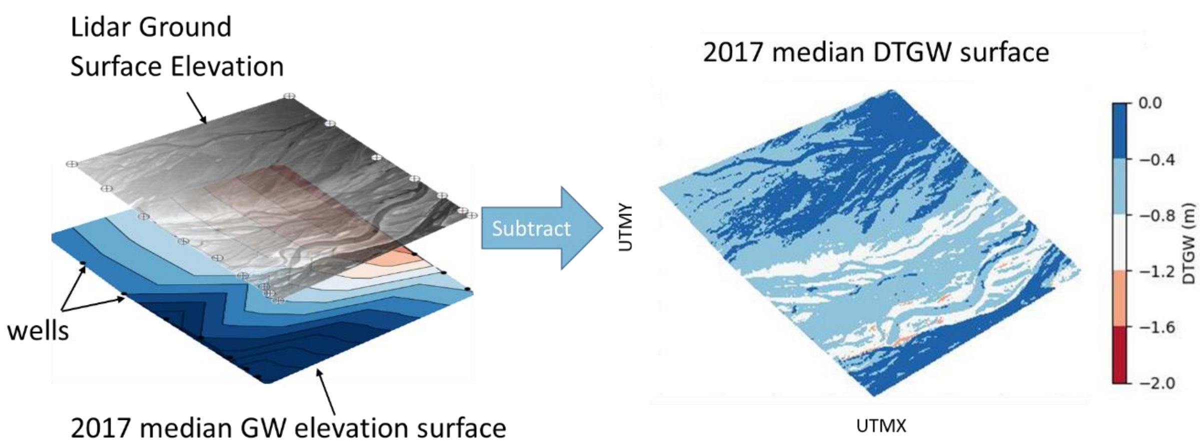


Figure 2- Illustration of method for generating spatial depth to groundwater statistics.

Groundwater depths for point locations (i.e., wells) and area surfaces (i.e., DTGW surfaces) are summarized in Tables 1 and 2 respectively. The mean and median DTGW values for wells and surfaces were similar, with surfaces having slightly smaller (i.e., shallower) mean DTGWs. This indicates that point data (wells) decently capture mean and median DTGW values for the sites. Unsurprisingly, surfaces recorded a wider range of DTGW values, indicating that area-based methods capture a broader range of groundwater depth conditions that occur at wet meadow sites. Three wells at Fox, and thirteen at Binfield, recorded minimum (i.e., shallowest) DTGWs that were above ground surface (i.e., negative values). Note, positive values indicate groundwater levels below ground surface. Summary statistics for all years indicate that Fox had statistically greater mean, median, and annual ranges of DTGW values (two-tail t-Test alpha = 0.05, t=7.32, 7.10, and 7.33, p=6.92e-8, 6.68e-8, and 3.77e-8) suggesting Fox had deeper groundwater levels overall during the study period. Additional plots exploring relationships between groundwater levels and distance from the channel are included in Appendix A.

DTGW distributions were tested for normalcy by plotting histograms and performing statistical tests for normal distributions including Kolmogrov Smirnov, Lilliefors, and Shapiro Wilk. Annual and daily subsets of DTGW values were tested to evaluate normalcy of spatial and temporal distributions respectively. Annual subsets of data tested included yearly timeseries of groundwater depth values for individual wells (i.e., to test temporal distributions) and daily sets of gridded values from interpolated DTGW surfaces (to test spatial distributions).

Example histogram results are shown in Figure 3. P-values for all statistical tests were less than 0.05, indicating non-normal distributions. Non-normal data distributions are characterized by outliers or multiple modes and are best summarized using nonparametric statistics that are considered resistant, or relatively unchanged by outliers. Resistant statistical metrics (e.g., median, interquartile range, median absolute deviation) are therefore recommended for summarizing groundwater depth data at wet meadow sites. Notably, despite this finding, mean and median DTGW values for the duration of the study period were similar as reported in Tables 1 and 2. This indicates that over longer time periods, groundwater depth data may converge to a more normal distribution. However, statistical tests calculated for subsets of daily mean DTGW spanning the entire study period for individual wells also indicated non-normal distributions.

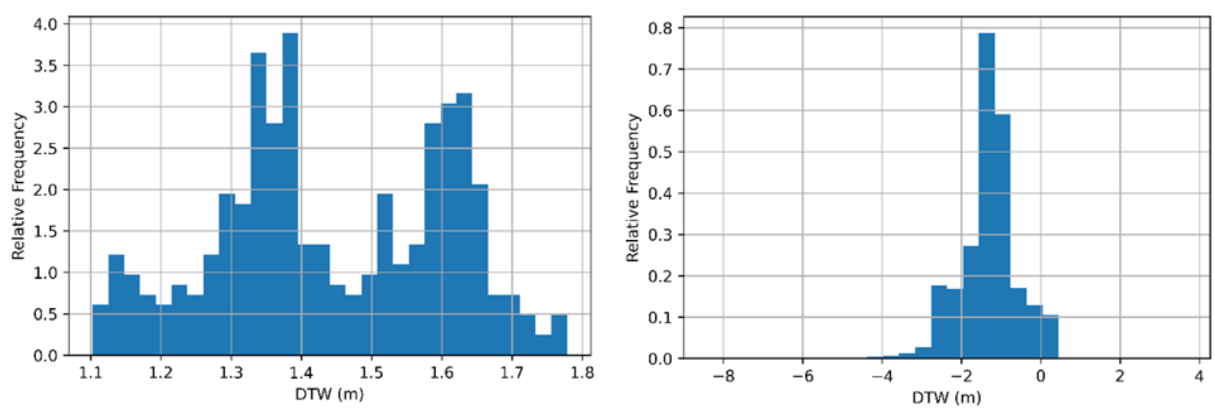


Figure 3– Histograms exemplifying distributions for subsets of DTGW data from the Fox site. Left subset includes all DTGW measurements for well 105 during 2014, and right subset includes spatially interpolated groundwater depths at the Fox site on 7/1/2020.

Table 1 – Point-based groundwater depth statistics for the period March 2013 to June 2021.

	Loc	Max (m)	Min (m)	Mean (m)	Median (m)	Range (m)
Fox	101	2.6	0.73	1.7	1.69	1.88
	102	2.42	0.48	1.53	1.52	1.94
	103	2.61	0.65	1.77	1.76	1.96
	104	1.64	-0.35	0.87	0.87	1.99
	105	2.05	0.21	1.33	1.33	1.83
	106	2.07	0.22	1.36	1.36	1.85
	107	2.16	0.29	1.45	1.44	1.88
	108	2.96	1.11	2.32	2.32	1.85
	109	2.33	-0.03	1.71	1.71	2.36
	110	2.59	0.15	1.96	1.96	2.43
	111	2.32	-0.13	1.68	1.68	2.45
	112	1.9	0.08	1.22	1.18	1.82
	113	2.21	0.37	1.57	1.57	1.84
	114	1.91	0.28	1.18	1.18	1.64
	115	2.35	0.43	1.5	1.5	1.92
	116	2.42	0.03	1.68	1.71	2.39
	Site Mean	**2.28**	**0.28**	**1.55**	**1.55**	**2.00**
Binfield	201	0.99	-0.42	0.36	0.35	1.4
	202	1.32	-0.12	0.69	0.67	1.44
	203	1.08	-0.35	0.47	0.45	1.43
	204	1.68	-0.08	1.08	1.07	1.76
	205	1.39	-0.56	0.78	0.79	1.94
	206	1.41	-0.58	0.78	0.8	1.99
	207	1.72	-0.43	1.1	1.1	2.15
	208	1.31	-0.56	0.68	0.68	1.87
	209	1.38	-0.4	0.73	0.73	1.78
	210	1.53	0.23	0.99	0.99	1.3
	211	1.67	0.39	1.14	1.16	1.28
	212	1.66	0.07	1.09	1.07	1.59
	213	1.52	-0.28	0.93	0.93	1.8
	214	1.29	-0.65	0.71	0.71	1.94
	215	1.33	-0.83	0.58	0.55	2.16
	216	1.13	-0.82	0.53	0.51	1.95
	Site Mean	**1.40**	**-0.34**	**0.79**	**0.79**	**1.74**

Table 2 – Area-based groundwater depth statistics for the period March 2013 to June 2021

Location	Year	Max (m)	Min (m)	Mean (m)	Median (m)	Range (m)
Fox	2013	9.09	-4.25	1.70	1.67	3.42
	2014	8.93	-4.36	1.57	1.56	3.10
	2015	8.66	-4.66	1.30	1.37	4.79
	2016	8.63	-4.30	1.25	1.28	3.16
	2017	8.76	-3.97	1.43	1.44	2.89
	2018	8.74	-4.08	1.44	1.43	2.82
	2019	8.53	-4.79	1.27	1.32	3.42
	2020	8.79	-4.23	1.43	1.45	2.67
	Site Mean	**8.77**	**-4.33**	**1.42**	**1.44**	**3.28**
Binfield	2013	2.73	-0.76	0.87	0.87	3.20
	2014	2.60	-0.64	0.71	0.70	2.72
	2015	2.40	-1.39	0.54	0.55	4.14
	2016	2.49	-0.83	0.67	0.68	3.14
	2017	2.50	-0.78	0.62	0.63	2.22
	2018	2.45	-0.74	0.63	0.61	1.83
	2019	2.34	-1.18	0.49	0.50	2.98
	2020	2.61	-0.80	0.61	0.67	3.71
	Site Mean	**2.51**	**-0.89**	**0.64**	**0.65**	**2.99**

3.2 Depth-to-Groundwater area proportions

The proportion of the study site that falls within a given range of groundwater depths over a set time period is a useful metric for comparing sites with strong spatial groundwater depth variability (e.g., Wesche et al., 1994). We calculated the area proportions for the minimum, maximum, spring (Mar-May) median, and summer (Jun-Aug) median groundwater depths at the two study sites for the period between March 2013 to September 2020. DTGW surfaces were generated as described previously, except with the minimum, maximum, spring median, and summer median water level assigned at each well. Area proportions, reported as percent, were calculated as the area that fell within a given DTGW range divided by the total area of the surface.

A major objective of the area proportion exercise was to compare hydrology at the two study sites to other wet meadows in the CPRV. We replicated the methodology of Wesche et al. (1994) to the degree possible, though their study period was shorter (4-years) and surface elevations were manually surveyed leading to reduced resolution. NOAA climate records for a nearby Kearney station (Site No. USC00254335) report similar precipitation and temperature records during the two study periods despite several years separation. Despite the differences, we expect comparisons to be useful. A map with locations of study sites relative to Wesche et al. (1994) sites is presented in Figure 4.

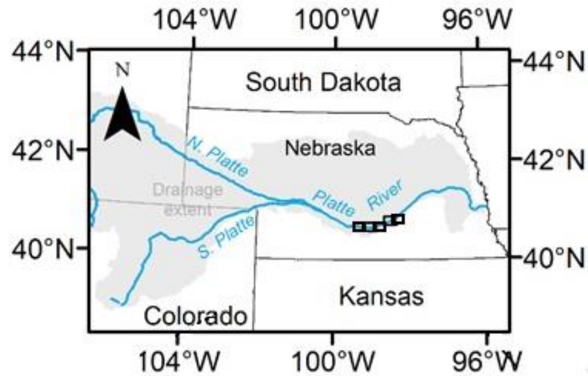


Figure 4 – Map showing the location of Binfield and Fox sites relative to the three Wesche et al. (1994) study sites. From west to east, sites (represented by squares) are Elm Creek, Fox, Rowe Sanctuary, Binfield, and Crane Meadows.

Area proportion results for the Binfield and Fox study sites, along with those of Wesche et al. (1994), are presented in Table 3. The datasets reveal the broad range of groundwater depths encountered within and across wet meadows. The wettest site from the group is Crane Meadows, for which 94% of the site area recorded median spring groundwater depths within 0.6m of the surface. Sites become increasingly dryer towards the west, with Binfield recording median spring groundwater depths with 0.6m across 73% of the site, Rowe Sanctuary recording 31%, Fox recording 15%, and Elm Creek recording less than 0.5%.

Table 3 – Area-based groundwater depth statistics. Binfield and Fox summarize the period from March 2013 to June 2021. Elm Creek, Rowe Sanctuary, and Crane Meadows summarize the period from July 1988 to September 1992 and are from Wesche et al., 1994.

Proportion of each study area within each DTGW range (percent)

	Depth-to-Groundwater (m)			min	max	spring median	summer median
Binfield			> 0	0	89	3.7	2
	0.0	to	-0.3	0	6	23	5
	-0.3	to	-0.6	0.35	3	46	10
	-0.6	to	-0.9	6.5	1	24	57
	-0.9	to	-1.2	44	0	4	23
	-1.2	to	-1.5	36	0	0	4
	-1.5	to	-1.8	12	0	0	0
		<	-1.8	1.2	0	0	0
Fox			> 0	0.35	21	5.7	4
	0.0	to	-0.3	0.1	21	4	4
	-0.3	to	-0.6	2.0	28	5	4
	-0.6	to	-0.9	4.3	10	11	6
	-0.9	to	-1.2	3.9	5	23	17
	-1.2	to	-1.5	4.9	5	21	28
	-1.5	to	-1.8	13	6	11	15
		<	-1.8	72	3	20	23
Elm Creek		>	> 0	0	1	0.0	0
	0.0	to	-0.3	0	9	0	0
	-0.3	to	-0.6	0	38	<.5	0
	-0.6	to	-0.9	<.5	39	6	2
	-0.9	to	-1.2	10	11	36	21
	-1.2	to	-1.5	46	2	50	55
	-1.5	to	-1.8	37	0	8	20
		<	-1.8	7	0	<.5	2
Rowe Sanctuary			> 0	<.5	4	<.5	<.5
	0.0	to	-0.3	<.5	46	<.5	<.5
	-0.3	to	-0.6	<.5	37	31	<.5
	-0.6	to	-0.9	8	12	58	33
	-0.9	to	-1.2	68	1	11	56
	-1.2	to	-1.5	33	0	<.5	11
	-1.5	to	-1.8	1	0	0	<.5
		<	-1.8	0	0	0	0
Crane Meadows			> 0	0	46	24	0
	0.0	to	-0.3	0	0	53	<.5
	-0.3	to	-0.6	6	11	17	38
	-0.6	to	-0.9	55	2	5	45
	-0.9	to	-1.2	30	1	1	14
	-1.2	to	-1.5	8	<.5	<.5	2
	-1.5	to	-1.8	1	0	0	1
		<	-1.8	<.5	0	0	<.5

3.3 Hydroperiod

Hydroperiod is defined as the number of days for which surface inundation (i.e., standing water) occurs at a given wet meadow site. Whiles and Goldowitz (2001) reported hydroperiod durations ranging from 94 to 365 days at 5 study sites on Mormon Island and Wild Rose Ranch using manual measurements during a 1-year study period. Brinley-Buckley et al. (2021) recorded hydroperiod at Mormon Island over a 6-year period using timelapse imagery positioned to record water extent at a single location. They report a mean hydroperiod duration of 141 days, typically spanning from 10 December to 1 May, and generally peaking in the early spring. They also found a bimodal distribution, peaking again in late spring during wetter years and rapid response to precipitation events, strong relation to streamflow, and observed that an elevated groundwater table was necessary for sustained inundation.

As stressed throughout this report, groundwater elevation and inundation dynamics vary significantly across wet meadow sites. One would expect the hydroperiod in one wet meadow swale to vary from the hydroperiod in another, even if the separation were only a few hundred meters. To account for spatiotemporal variability in quantifying the hydroperiod, in addition to the number of days with standing water, we calculate the proportion of each study site with standing water for each day during the study period, which we then summarize using statistical metrics. The daily percent was calculated using DTGW surfaces described above, as the area of grid cells that recorded a negative DTGW (i.e., groundwater elevation above ground surface) divided by the total area of the gridded surface. The mean, median, minimum, and maximum proportion for each year and study site were calculated.

Hydroperiod metrics are summarized in Tables 4 and 5. The recorded hydroperiod duration was 365 days per year at Fox and 315-365 days per year at Binfield, indicating both sites recorded standing water on part of the site for the entire year. Note, 2013 and 2020 data are reported but not included in summary metrics because part of the data were missing. These hydroperiod duration results are not typical. Whiles and Goldowitz (2001) reported hydroperiod duration of 296 days while Brinley Buckley et al. (2021) reported a mean hydroperiod duration of 141 days, lasting from 10 December to 1 May, with annual variations. However, it is not surprising that the area-based method yielded these results since both sites have topographic lows that likely have groundwater at or near the surface for most of the year. At Fox, topographic lows are concentrated at excavations created as part of restoration efforts described previously. At Binfield, topographic lows are concentrated near the channel and within an inland swale that is likely a historical channel remnant.

To reduce the influence of perennially inundated depressions on results, a threshold area was determined for each site representing a theoretical lower limit that separates ponded water and temporary standing water. Manually determined thresholds were 4% and 0.5% at Fox and Binfield respectively, approximately corresponding to the area of permanent inundation. The selection of these thresholds impacts the results and may require careful consideration during future applications. Although thresholds add uncertainty to this method, it is worth noting that other methods for assessing hydroperiod typically collect data for a single location (i.e., one swale) and do not account for an entire wet meadow site. Hydroperiod calculations with the threshold applied were closer to other studies.

Table 4– Hydroperiod summary table for the Fox site.

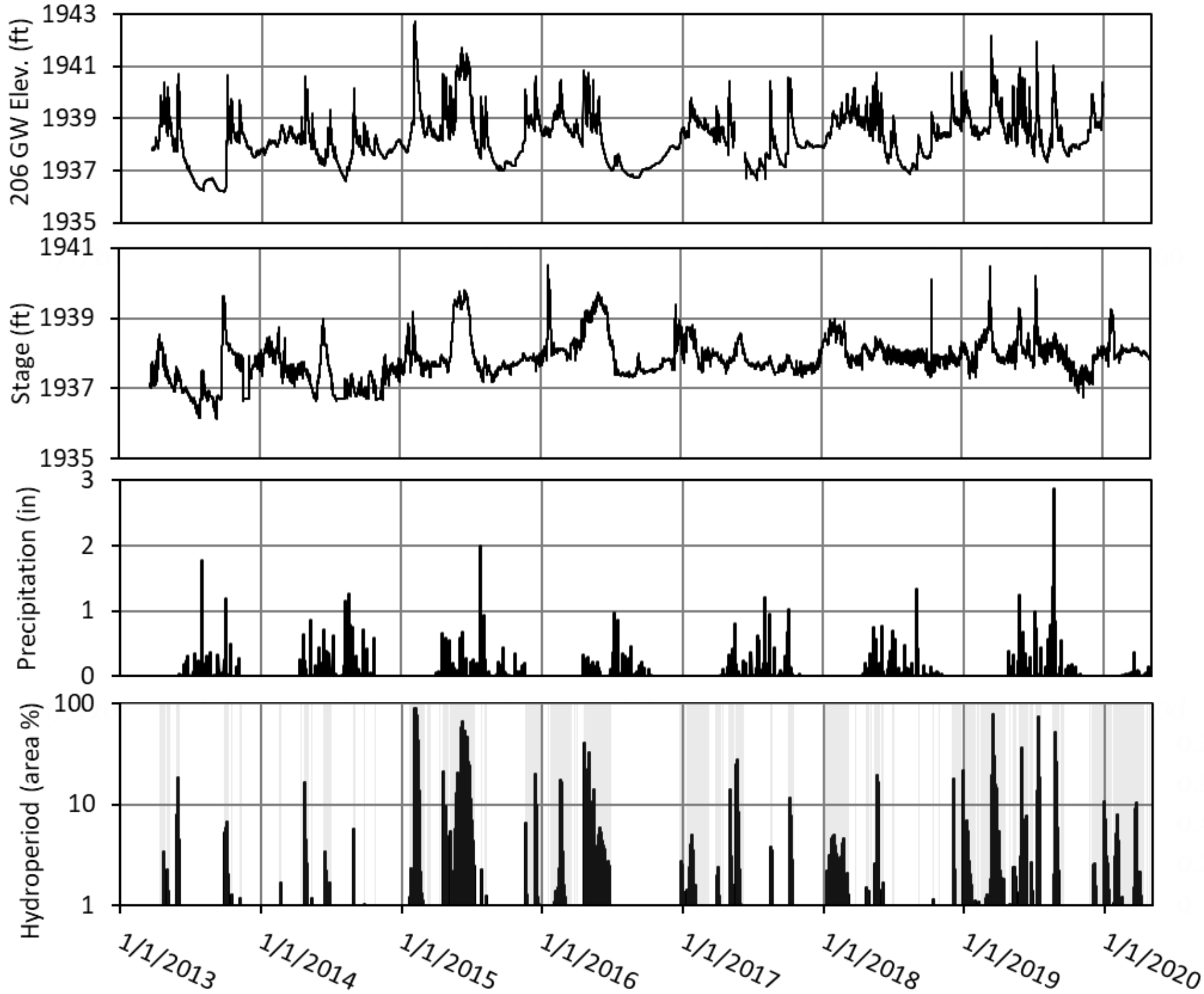
	Year	# Days	# Days (>4%)	Mean (%)	Median (%)	Min (%)	Max (%)
Fox	2013	295	24	1.45	0.62	0.05	6
	2014	365	50	2.15	2.05	0.47	7
	2015	365	237	5.91	4.59	2.40	22
	2016	366[1]	276	6.45	6.10	2.25	13
	2017	365	187	4.28	4.23	1.09	9
	2018	365	173	3.48	3.93	0.56	7
	2019	365	269	5.83	5.34	2.99	13
	2020	366[1]	176	3.69	3.16	0.52	15

Note: [1]366 days indicates leap year

Table 5– Hydroperiod summary table for the Binfield site.

	Year	# Days	# Days (>0.5%)	Mean (%)	Median (%)	Min (%)	Max (%)
Binfield	2013	180	46	0.45	0.05	0	18
	2014	324	36	0.28	0.09	0	17
	2015	364	164	5.30	0.39	0	89
	2016	365	138	1.6	0.25	0	40
	2017	349	121	0.96	0.15	0	28
	2018	350	122	0.92	0.20	0	22
	2019	364	202	3.1	0.60	0	79
	2020	315	108	0.81	0.04	0	35

Figures 5 and 6 show plots of hydroperiod as percent area along with groundwater elevation (at a select well), stage, and precipitation. Gray areas in the hydroperiod plot indicate times where the area percent was above the threshold. Notably, hydroperiod peaks during winter, spring, or early summer, depending on the year. The inundated area percent of the site is highly variable, showing periodic wetting and drying cycles rather than a slow rise and fall marking the start and end of the hydroperiod. Like Brinley Buckley et al. (2021), our data record a bimodal distribution during wet years (e.g., 2015 and 2016), peaking again in late spring during wetter years. They identified a relationship between hydroperiod, precipitation and streamflow during these periods, which can also be observed in our data (e.g., Figures 5 and 6).



459

460 *Figure 5 – Groundwater elevation (Well 206), stage, precipitation, and hydroperiod area percent for the*
461 *Binfield site. Gray shading in the hydroperiod plot indicates area percent was above the threshold value.*

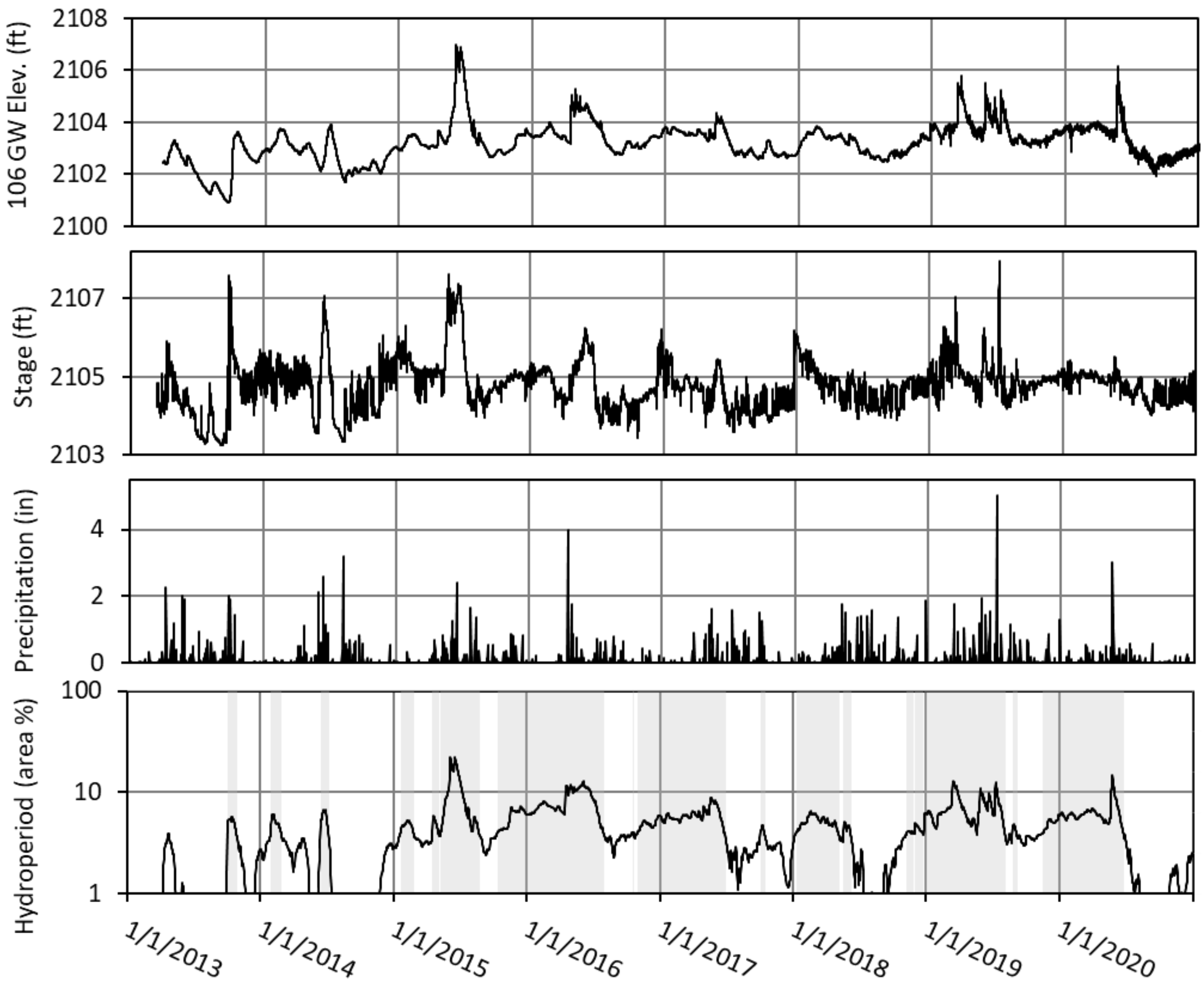


Figure 6 – Groundwater elevation (Well 106), stage, precipitation, and hydroperiod area percent for the Fox site. Gray shading in the hydroperiod plot indicates area percent was above the threshold value.

3.4 Discussion / implications

What emerges from this dataset is a broad spectrum of wet meadow hydrologic conditions with respect to groundwater depth. The two Program study sites (Fox and Binfield) fall somewhere in the middle of the recorded ranges reported by Wesche et al. (1994). Both Wesche et al. (1994) and Brinley Buckley et al. (2021) identified Crane Meadows as being an especially "wet" site. Brinley Buckley et al. (2021) and others (anecdotally) also describe Crane Meadows as an "archetypal" wet meadow. This analysis highlights uncertainties about what defines an "archetypal" CPRV wet meadow. The inherent variability within and across sites makes it challenging to define ideal hydrologic, vegetation, and biotic conditions. Future work is needed to evaluate whether these sites can be used as realistic targets for wet meadow restorations elsewhere.

When tasked with managing wet meadow sites, it is important to recognize the spectrum of conditions that exist across wet meadows in the CPRV. Realistic targets will likely fall somewhere within the limits of reported conditions, but will ultimately be tied to site specific constraints, likely related to topography and hydrology.

4 Groundwater-vegetation links

In 1994, Wesche et al. published a hydrologic study that utilized depth-to-groundwater duration curves, i.e., cumulative frequency distributions, to summarize highly variable groundwater data collected at wet meadow sites in the CPRV. Cumulative frequency distributions show the number of days (i.e., the percent of time) that groundwater was at or above a particular level. Wesche et al. (1994) recommend that future studies combine depth-to-groundwater duration curves with plant species response at wet meadows to predict how water management actions might affect vegetation.

In 2004, Henszey et al. published such a study quantifying the relationship between groundwater depth and vegetation. They collected groundwater depth and vegetation density data across a range of moisture and topographic gradients at wet meadow study sites in the CPRV and developed a series of non-linear models that can be used to predict which of four wet meadow vegetation groups (emergent wetland, sedge meadow, mesic prairie, and dry ridge) are likely to be present based on groundwater depth statistics for a given location.

Although the utility of this study was recognized by Wesche et al. (1994) for management applications and by Weir and Chaves Ramirez (2011) for defining wet meadows, to our knowledge, few if any studies have applied results of Henszey et al. (2004) to manage or evaluate wet meadows in the CPRV. One of the remaining challenges for managing wet meadow is that hydrologic targets are highly uncertain due to spatiotemporal variability and complexity. The Henszey et al. study provides a clear response variable (i.e., vegetation) that can be used to assess wet meadow hydrology and therefore aid in managing sites.

According to the U.S. Army Corps of Engineers (USACE), the U.S. Environmental Protection Agency, and the State of Nebraska, "Wetlands are areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions.” (USACE 1987). Amongst other rigorous criteria, the presence and “prevalence” of wetland vegetation is key indicator of wetland status. For wet meadow sites in the CPRV, Henszey et al. (2004) identified four key wet meadow vegetation groups that occur across the topographic and moisture gradients described previously. They are, from wettest to driest, emergent, sedge meadow, mesic prairie, and dry ridge. Of these, emergent and sedge meadow groups are considered to represent “wetland” vegetation, as they contain the majority of obligate and facultative wetland species present at wet meadow sites. The prevalence of emergent and sedge meadow vegetation at wet meadow sites may therefore be considered a critical requirement for a site to be distinguished as a wet meadow. Further, since many historic wet meadow sites have been degraded due to agricultural and other land uses through time (e.g., Currier et al., 1985; Sidle et al., 1989), if one were able to quantify the necessary hydrologic requirements for supporting such vegetation, hydrologic conditions, rather than vegetation prevalence, could be substituted as a target for restoration and management efforts.

We extend the work of Wesche et al. (1994) and Henszey et al. (2004) and test this theoretical framework by evaluating whether the present hydrologic conditions at two wet meadow sites are sufficient to support a prevalence of critical wet meadow vegetation types. We first develop spatially continuous predictions for vegetation type based on hydrology across the two wet meadow study sites after Henszey et al. (2004). We then evaluate whether wetland prevalence meets thresholds delineated by the USACE definition for wetlands. We then consider whether management actions could be used to alter the hydrology and associated vegetation to improve wet meadow status. In doing so, we demonstrate a methodology that has broader utility for managing wet meadow sites throughout the CPRV.

4.1 Methods

Henszey et al. (2004) identified the growing season 7-day moving average maximum groundwater-level (L7th) as the strongest predictor for type of wet meadow vegetation (emergent, sedge meadow, mesic prairie, or dry ridge). They tested a range of groundwater depth statistics and determined the L7th to be the best predictor using the Akaike information criterion. Their results included a series of non-linear models that use the L7th as an input variable to predict which of four key wet meadow vegetation groups is likely to be present. The models also included a series of L7th depth ranges associated with each vegetation group. Vegetation groups and L7th ranges, summarized in Figure 7, were from wettest to driest, >20 cm for emergent (i.e., wetland), 20 to -30 cm for sedge meadow, -30 to -135 cm for mesic prairie, and < -135 cm for dry ridge. Positive values for the L7th indicate maximum growing-season groundwater levels that are above ground surface. Henszey et al. (2004) noted that the emergent category closely matched the 1989 interagency manual criterion for wetland hydrology (Federal Interagency Committee for Wetland Delineation, 1989).

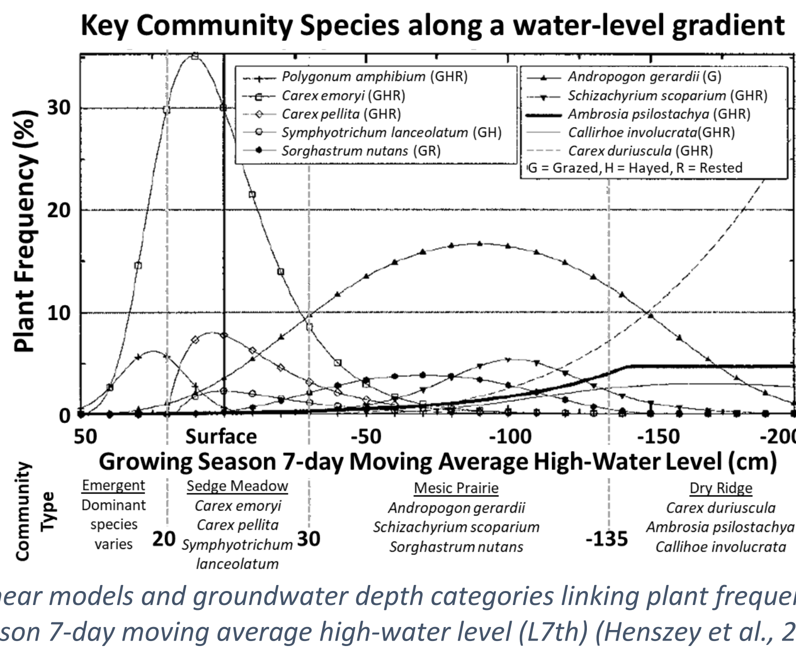


Figure 7 – Nonlinear models and groundwater depth categories linking plant frequency to the growing season 7-day moving average high-water level (L7th) (Henszey et al., 2004).

We calculated the L7ths for a series of groundwater depth surfaces to develop spatially continuous predictions of vegetation group across the two Program managed wet meadow study sites. Groundwater elevation surfaces were created using methods described previously in this report with the following exceptions. First, annual subsets of the 7-day moving average groundwater elevation timeseries were calculated for each well. Following Henszey et al. (2004), annual subsets were limited to the growing season period (15 March to 15 October). Linear interpolation was then performed with the 7-day moving average timeseries data assigned at well points to generate a moving average groundwater elevation surface for each day during the growing season. Next, the maximum value for each grid cell was calculated using an algorithm developed to successively update raster grid cells whenever a new cell value exceeded a previous value. This maximum moving average groundwater elevation surface was subtracted from the LIDAR ground surface elevation to obtain the DTGW. The result was a single gridded raster surface per growing season and year in which each grid cell represents the L7th for that location. The vegetation group that corresponds to the L7th DTGW was then determined, and the area proportion of each site that fell within each L7th and vegetation group was calculated for growing seasons during the study period.

4.2 Results

L7th DTGW area proportion results are summarized in Table 6 and L7th DTGW raster surfaces are presented in Figures 8 and 9 respectively. For this section, groundwater depths are reported with negative values indicating groundwater below ground surface, differing from the previous section to maintain consistency with Henszey et al. (2004). L7th values at Binfield, in order of decreasing prevalence, occur within ranges -0.3 to 0.2m, -1.35 to 0.3m, and 0.2 to 2m, corresponding to vegetation groups sedge meadow, mesic prairie, and emergent. During all study years, the Binfield site records almost no (<1%) L7ths in the -5 to -1.35m range corresponding to dry ridge vegetation. L7th values at Fox, in order of decreasing prevalence, occur within ranges -1.35 to 0.3m, -5 to -1.35m, -0.3 to 0.2m, , and 0.2 to 2m, corresponding to vegetation groups mesic prairie, dry ridge, sedge meadow, and emergent. Notably, Fox records considerable area proportions within the dry ridge category (11 to 34%) during all years.

In comparing the two Sites, Binfield exhibits greater proportions of sedge meadow and lesser proportions of emergent, mesic prairie, and dry ridge than Fox during all study years. Since Fox is generally dryer (i.e., deeper groundwater table), it was expected to have lesser emergent cover than Binfield. However, L7th figures show that the emergent zones at the Fox Site are primarily concentrated around the perimeter of excavations that were created to bring the ground surface closer to the water table. This result would suggest that recontouring efforts improved conditions at the Fox Site. However, it is worth noting that the Fox site supports a lower proportional area of sedge meadow than the Binfield site and most of the site remains as mesic prairie or dry ridge, suggesting that larger scale recontouring efforts would be needed to significantly alter the hydrology at the site.

Table 6 – Area proportion results for corresponding L7th depth and vegetation category (E=emergent, S=sedge meadow, M=mesic prairie, D=dry ridge) by growing season.

	Binfield (Area %)				*Fox (Area %)*			
Veg.	E	S	M	D	E	S	M	D
L7th (m)	(0.2 to 2)	(-0.3 to 0.2)	(-1.35 to -0.3)	(-5 to -1.35)	(0.2 to 2)	(-0.3 to 0.2)	(-1.35 to -0.3)	(-5 to -1.35)
2013	1.1	47	52	0.05	2.4	6.9	58	31
2014	0.31	59	41	0.07	3.4	6.8	60	29
2015	11	81	8.4	0	14	28	46	11
2016	1	74	25	0.03	9.7	11	61	17
2017	0.71	73	26	0.03	5.8	6.6	62	25
2018	0.25	60	39	0.06	3.3	6.6	55	34
2019	4.4	81	14	0.01	9.3	11	62	17
2020[1]	0.53	68	31	0.02	9.0	9.7	63	18

[1]2020 total area for Binfield was slightly less due to lack of data for well 216.

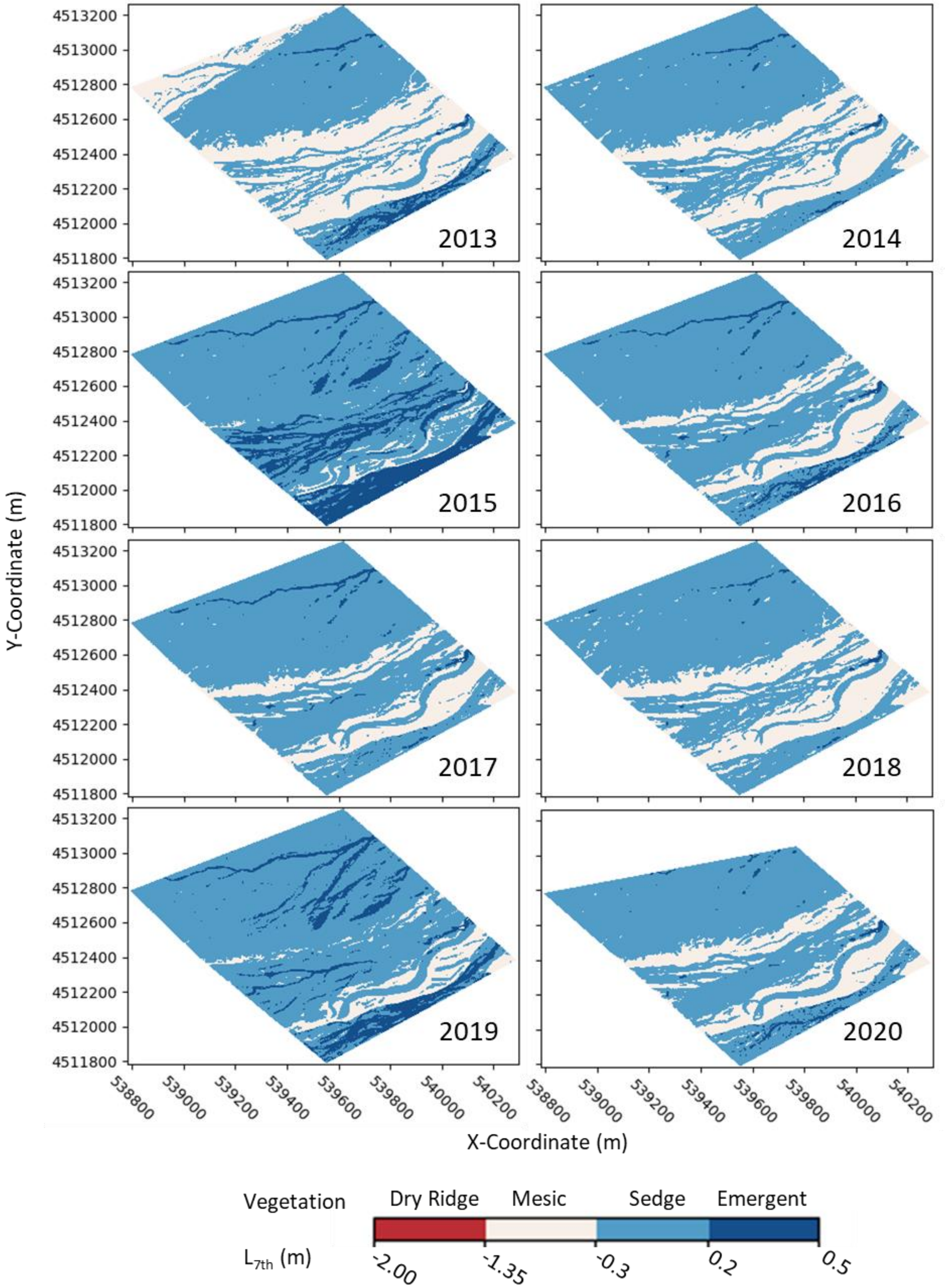
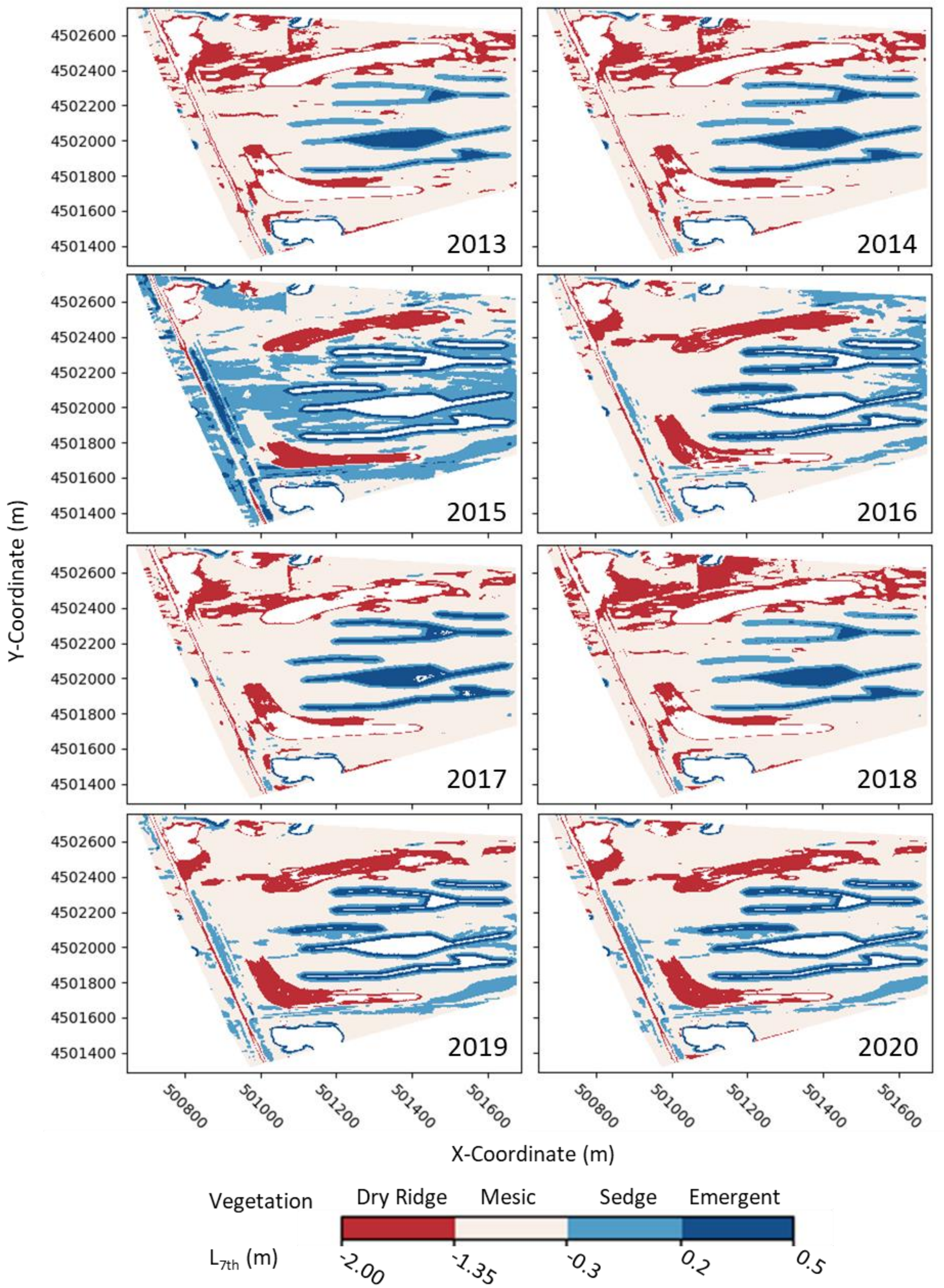


Figure 8 – L_{7th} depth to water surfaces for Binfield site. Color indices correspond to Henszey et al. (2004) vegetation categories. Well 216 DTGW data was unavailable for 2020, hence truncated surface.



583

584 *Figure 9 – L7th depth to water surfaces for Fox site. Color indices correspond to Henszey et al. (2004)*
585 *vegetation categories.*

4.3 Discussion

Significant differences in hydrology and associated vegetation predicted from hydrology were recorded at the two wet meadow study sites. The natural, nondegraded Binfield tract exhibits groundwater conditions likely to support predominantly sedge meadow and mesic prairie vegetation groups whereas the Fox Site is likely to support primarily mesic prairie and dry ridge vegetation. Since wetland vegetation is a key feature of wet meadows, results suggest that the Fox site is unlikely to support a prevalence of wetland vegetation, and therefore, is unlikely to achieve the ecosystem functions that are desired from wet meadows.

Wet meadows are ephemeral wetlands. If we assume that wet meadows should meet requirements for wetland delineation, then conditions require "a prevalence [i.e., >50%] of dominant plant species [that] are adapted to life in saturated soil conditions… where dominance refers to the spatial extent of a species observed in the field" (Nebraska Department of Transportation, 2020). To assess whether study sites meet wetland requirements, we extend conditional requirements for vegetation to hydrology using an area threshold such that ≥ 50% of the site has hydrologic conditions that could theoretically support hydrophytic vegetation (i.e., emergent and sedge meadow vegetation groups). It is important to note that this analysis does not utilize field-based plant observations and therefore does not indicate actual wetland status. For all years but 2013, the Binfield site meets area requirements for wetland designation, with area percentages ranging from 48.1% (in 2013) to 92% of the site having hydrologic conditions that can support hydrophytic species. Conversely, for all study years, the Fox site does not meet criteria for wetland designation, with site area percentages ranging from 9.4% to 42%.

We calculated the required change in L7th that would be necessary to achieve hydrologic conditions that could support wetland vegetation across at least 50% of the Fox Site for years 2013-2020. Results are presented in Table 7. Changes in L7th range from 0.075m during the wettest year (2015) to 0.93m during the driest year (2018), with a mean of 0.59m. This value may be interpreted as the average elevation that that peak sustained (~7 days) groundwater levels would need to rise in order to support wetland vegetation across 50% of the site.

Table 7 – Required changes in L7th and area percent to achieve 50% wetland vegetation.

Year	ΔL7th (m)
2013	0.89
2014	0.84
2015	0.075
2016	0.41
2017	0.73
2018	0.93
2019	0.42
2020	0.45
AVERAGE	**0.59**

Potential management actions that could be used to improve site hydrology would require either changing the land surface elevation or changing site groundwater levels. Topographic recontouring (i.e., lowering the land surface through excavation) is one method of altering the ground surface to improve wetland hydrology. As mentioned, such an effort was previously undertaken at the Fox site which resulted in the addition of 5 elongate depressions on the site. However, this method requires significant quantification of variability in site hydrology to achieve ideal conditions. The existing recontouring effort generated ponds with marginal areas that have ideal hydrologic conditions, and interior areas (ponds) that are too wet to support most wetland vegetation. Also, most of the site is still mesic prairie and dryland, indicating that to perform such an effort successfully would likely require excavating material off broad areas of the site.

The alternative would be to alter groundwater levels through managed aquifer recharge or added flows to the Platte River that would in turn raise riparian groundwater levels. In the next section, we describe and demonstrate the utility of a simple analytical model that can be used to test such management actions.

5 Modeling

Authors have investigated the relative importance of different controls for groundwater levels at CPRV wet meadows. A shallow, fluctuating groundwater table is required for sustained inundation (Brinley-Buckley et al., 2021), wetland vegetation (Henszey et al., 2004), and various animal species (Davis et al., 2006) at wet meadow sites. Previous studies have concluded that river stage exerts the predominant influence on groundwater levels, followed by precipitation, and to a minor degree ET (Whiles and Goldowitz, 1993; Wu, 2003; Wesche et al., 1994; Chen, 2007). Chen (2007) also suggests hydraulic lift caused by vegetative water use to be a relevant control, though the effect is not well constrained or documented.

To test the importance of various hydrologic controls, studies typically use models to predict the response of groundwater levels to a given hydrologic stress. Chen (2007) developed a three-dimensional numerical model to evaluate river-aquifer-vegetation hydraulic connections in the CPRV (approximately 5 miles west of the Fox site) with specified stresses for stage and ET. Chen identified a highly connected hydrologic system with rapid response in groundwater levels to fluctuations in stage. Loheide II and Gorelick (2007) developed a finite element model of variably saturated groundwater flow to assess hydroecologic functions of meadow systems in the Last Chance watershed in North Central California. They identified drainage to the stream as an important control for groundwater levels that in turn affected local vegetation. In 2012, the Program developed an analytical model to test how water management actions in the form of flow release would affect groundwater levels at wet meadow sites (PRRIP, 2012).

Here, we develop a simple analytical model to test how different management actions could affect groundwater levels at wet meadow sites in the CPRV. The purpose and implementation of the model is similar to the 2012 model used by the program (PRRIP, 2012), with added changes to incorporate the effects of precipitation on ET. Notably, the model presented herein is not intended to fully reproduce or describe groundwater flow dynamics at wet meadow sites. Rather, it is intended to be used as a simple tool for predicting groundwater level changes at wet meadow sites given stage, precipitation, and ET inputs. Accomplishing the former task almost certainly requires a numerical groundwater flow model, the development of which requires significant time, effort, and data collection. Numerical flow models also require site specific development, which means they are not easily transferred between sites. Conversely, this analytical model can be readily applied to different sites with relatively minimal parameter inputs.

The resulting model provides a useful tool for testing water management scenarios, with improved predictions as compared to previous models, indicated by relatively good matches between observed and modeled groundwater levels.

5.1 Methods

The model is based off the Glover Bank Storage equation (Eq. 1) (Glover, 1964) for which the change in groundwater level (*s(x,t)*) some distance from a river (*x*) is estimated at time (*t*) given an instantaneous step change in river stage (s_0) (e.g., Figure 10). The parameter α is hydraulic diffusivity defined as α=T/S where T is transmissivity (L^2T^{-1}) and S is storativity (dimensionless). Since the aquifer is unconfined, it is assumed that S is equal to specific yield (Sy), or the volume of water released under gravity from storage per unit cross-sectional area per unit decline in water table (Freeze and Cherry, 1979). Erfc is the complimentary error function.

$$s(x,t) = s_0\, erfc\left(\frac{x}{\sqrt{4\alpha t}}\right) \quad \text{Eq. 1}$$

The principle of superposition was applied to calculate stepwise changes in groundwater level caused by daily changes in stage (Eq. 2). Notably, Equation 2 does not model groundwater flow, but rather stepwise vertical changes in groundwater level due stepwise boundary stresses (i.e., stage changes).

$$s_{(x,t)} = \sum_{i=1}^{n} \Delta s_i\, erfc\left(\frac{x}{\sqrt{4\alpha(t-t_i)}}\right) ; t \geq t_i \quad \text{Eq. 2}$$

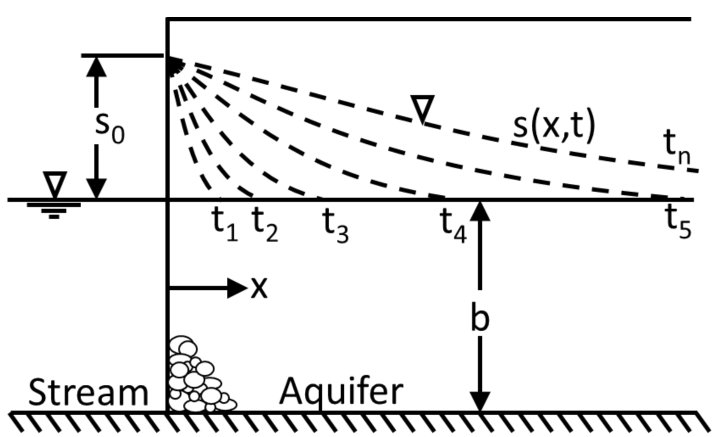


Figure 10 – Schematic illustrating variables in the Glover Bank Storage equation (Sanders, 2001).

Additional terms were added to the right side of Eq. 2 to account for daily effects of precipitation and evapotranspiration on groundwater levels. When precipitation falls on a pervious surface, some fraction is expected to reach the groundwater table as recharge (e.g., does not runoff or evaporate). This fraction is referred to as the recharge or precipitation infiltration coefficient. Groundwater level changes due to precipitation were calculated by multiplying a recharge coefficient (f) by the daily total precipitation (recorded at the on-site weather station) and dividing by the aquifer specific yield (i.e., $s_P(x,t)=fP(t_i)/Sy$). This relationship has been utilized in other study areas with shallow groundwater tables such as wetlands, where the effects of hysteresis and vadose zone transport are negligible on water table response (Gerla, 1992; Rosenberry and Winter, 1997). The high-hydraulic conductivity and use of daily timesteps further support this approach since infiltrated precipitation is expected to reach the water table in less than one day.

Specific yield relates the volume of precipitation to the available pore space in the aquifer and is defined as the volume of water released under gravity from storage per unit cross-sectional area per unit change in water table (Freeze and Cherry, 1979). This method is appropriate for sites with shallow water tables and high soil water content such as riparian wet meadows (Healey and Cook, 2002). Recharge coefficients (f) typically range from 0.1 to 0.6 (Andaulem et al., 2021), though the actual fraction is time-varying and depends on soil hydraulic properties and antecedent moisture. We tested a range of infiltration fractions between 0.1 and 0.6 and found a best-fit between modeled and observed groundwater level changes to occur with fractions between 0.4 and 0.6.

Yue et al. (2016) documented an exponential relationship between evapotranspiration from groundwater (ETg) and groundwater depth at the Binfield study site wherein the rate of ETg decreases as groundwater depth increases. Wesche et al., (1994) identified a similar exponential relationship between groundwater and ET, and this relationship has been found elsewhere in riparian zones (e.g., Lurtz et al., 2019). We calculate changes in groundwater level due to ETg as a weighted fraction of daily potential evapotranspiration divided by specific yield ($s_{ET}(x,t)= \beta ETo/Sy$) wherein a daily weight is calculated based on the depth to groundwater from the previous timestep. The weight, β, representing the ratio of ETg/ETo, is calculated using Equation 3 where ε and λ are fitting parameters describing the shape of the exponential relationship, as defined by Yue et al. (2016).

$$\beta=\varepsilon\exp(\lambda * DTW_{t-1}) \quad \text{Eq 3}$$

The final model equation is thus:

$$s_{(x,t)} = \sum_{i=1}^{n} \Delta s_i \, erfc\left(\frac{x}{\sqrt{4\alpha(t-t_i)}}\right) + \frac{fP(t_i)}{Sy} - \frac{\beta ET_0(t_i)}{Sy} \; ; t \geq t_i \quad \text{Eq. 4}$$

The analytical model was calibrated for each well location to achieve a best-fit between modeled and observed groundwater levels for growing seasons during the study period. Calibration involved varying hydraulic parameters (K, Sy, f) until a visual best-fit was achieved. Subsequently, ET parameters (ε and λ; Eq. 3) were determined through calibration implemented in Python using the SciPy optimization function curve-fit with the trust region method. This method applies non-linear least squares algorithm to calculate optimal parameters given a set of input data and a function. Calibration resulted in a range of location specific parameter estimates, which are discussed subsequently.

5.2 Results

Model calibration parameters are summarized in Table 8. All values are within the expected range for alluvial sediments. ET parameters were allowed to vary by location, as was found by Yue et al. (2016). Calibration would benefit from future site-specific measurements to justify the observed heterogeneity. Alternatively, site-specific parameters could be enforced (e.g., one K value for the entire site) while select parameters could be tested through calibration. The calibration process could be performed manually in Excel for a single location to make location specific predictions. However, if the model is intended to represent groundwater level changes across a site, it would be best to test the model for a range of parameters bounded by minimum and maximum values to test the sensitivity of results to calibration. This task has not been performed but is within the scope of possible next steps.

Figures 11 and 12 present model results for Wells 112 and 212 at the Fox and Binfield Sites for years 2014 through 2020 (note, insufficient data from 2020 were available for well 112). These wells were selected because results exemplify model performance across commonly occurring scenarios. Model results for all well locations and growing seasons are provided in Appendix B. The plots for years 2014 and 2018 show that the model reproduces general trends in groundwater level very well. Minor rises and falls are almost entirely captured and in sync with field observations. Plots for 2015 and 2017 show that the model is not reproducing groundwater levels during high-water table events. This could be an issue for predictions since it suggests that the model will under-predict scenarios where the water table is expected to be higher than recorded data. The 2017 plot also reveals significant variations in groundwater level during summer months which are not predicted by the model. While the cause of this discrepancy can't be certain, it is known that during this time the Program experimented with applying pumped well water to the site from on-site wells (communication with Justin Brei). Though pumped and applied water volume records were not compiled for this study, they may explain differences between modeled and observed values.

742 Table 8 – Model calibration results

Well	K (ft day^{-1})	Sy	f	ε	λ
101	400	0.15	0.4	0.50	-0.44
102	400	0.15	0.4	0.50	-0.45
103	600	0.15	0.5	0.50	-0.36
104	600	0.15	0.5	0.50	-0.40
105	600	0.12	0.5	0.50	-0.38
106	600	0.12	0.6	0.73	-0.42
107	600	0.12	0.6	1.35	-0.49
108	600	0.12	0.6	1.59	-0.34
109	650	0.12	0.5	0.58	-0.24
110	650	0.12	0.5	1.18	-0.31
111	400	0.12	0.5	0.50	-0.40
112	150	0.08	0.5	0.50	-0.32
113	150	0.08	0.5	0.66	-0.33
114	250	0.08	0.5	0.50	-0.30
115	150	0.08	0.5	0.50	-0.25
116	100	0.1	0.6	0.50	-0.16
201	800	0.25	0.8	0.98	-1.24
202	800	0.25	0.6	0.50	-0.50
203	800	0.2	0.7	0.50	-0.59
204	800	0.15	0.8	1.41	-0.33
205	800	0.1	0.8	1.58	-0.63
206	600	0.1	0.8	1.38	-0.63
207	800	0.08	0.8	2.00	-0.41
208	800	0.08	0.7	0.67	-0.33
209	600	0.1	0.8	0.59	-0.27
210	800	0.25	0.6	0.50	-0.82
211	800	0.25	0.6	0.68	-0.67
212	1000	0.1	0.6	2.00	-0.50
213	400	0.08	0.6	1.10	-0.32
214	800	0.08	0.8	1.04	-0.56
215	800	0.08	0.8	0.66	-0.37
216	800	0.1	0.7	0.72	-0.26

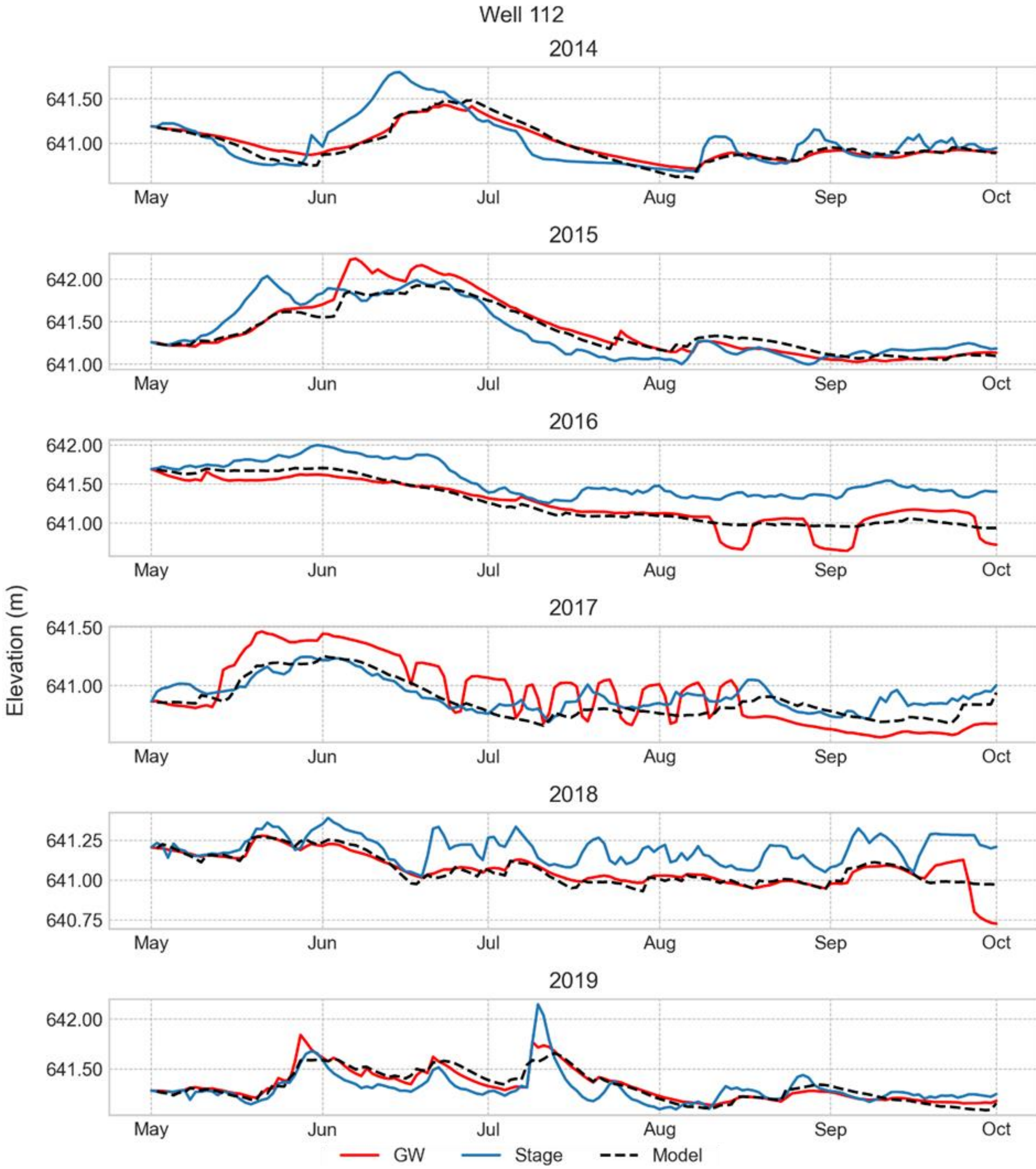
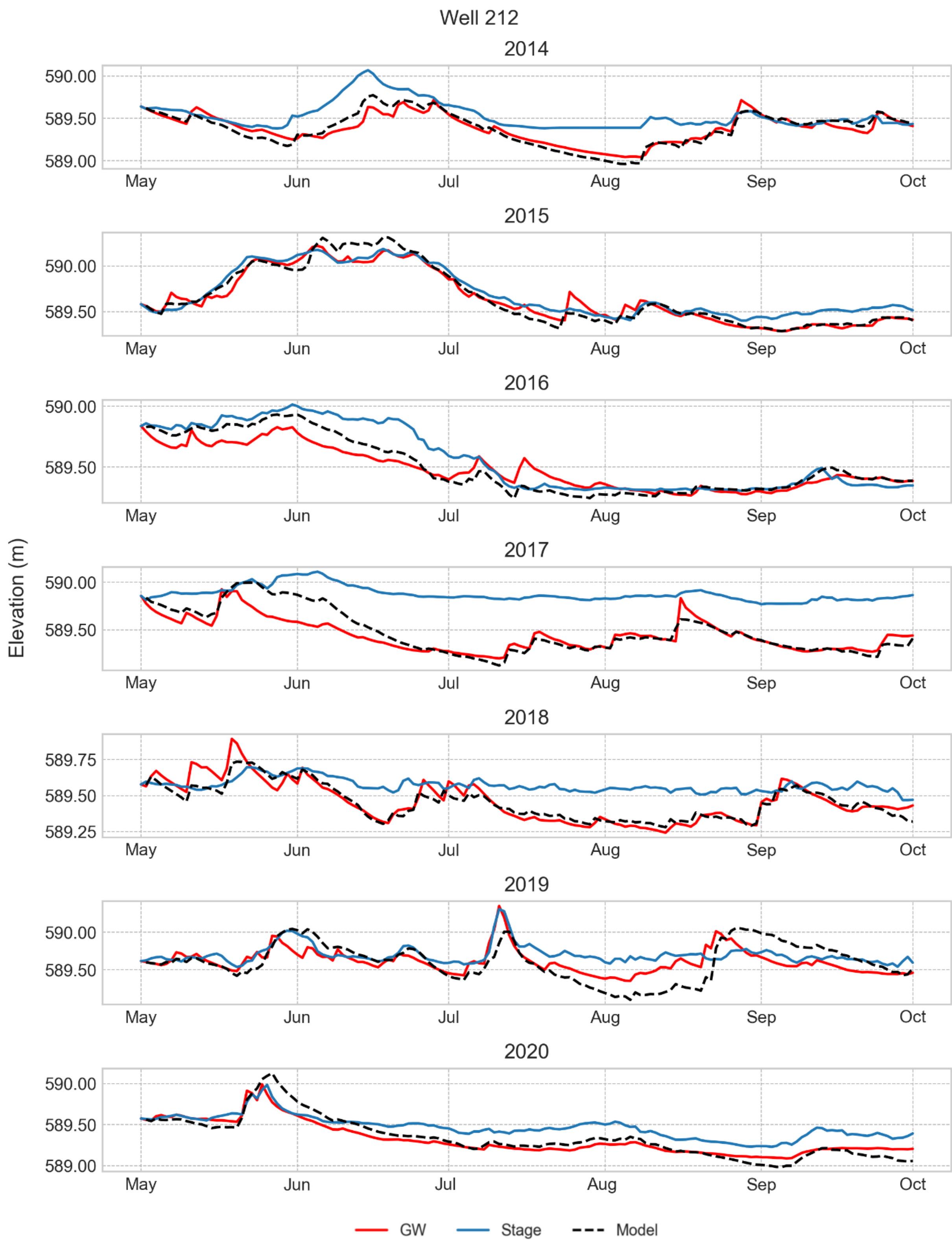


Figure 11 – Well groundwater levels, stage, and modeled groundwater levels for well 112.


Well 212
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
590.00
589.50
589.00
589.75
589.25
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

745

Figure 12 – Well groundwater levels, stage, and modeled groundwater levels for well 212.

5.3 Discussion

The model was applied to evaluate the effects of hypothetical management actions during a low-groundwater period in 2018. As described previously, groundwater depths at the Fox site were found to too deep to support key wetland vegetation groups as estimated using the L7th DTGW statistic. Changes of up to 0.93 m were estimated as necessary to achieve a hypothetical management target of 50% spatial coverage with wetland vegetation during 2018. One possible management action for achieving increased groundwater levels is to increase flow, and in turn stage, on the Platte River. Groundwater depths for well 116 are plotted in Figure 13, demonstrating the especially low spring groundwater levels that occurred in 2018. Well 116 was chosen because it is an inland well away from excavations, but this analysis could be performed for any well.

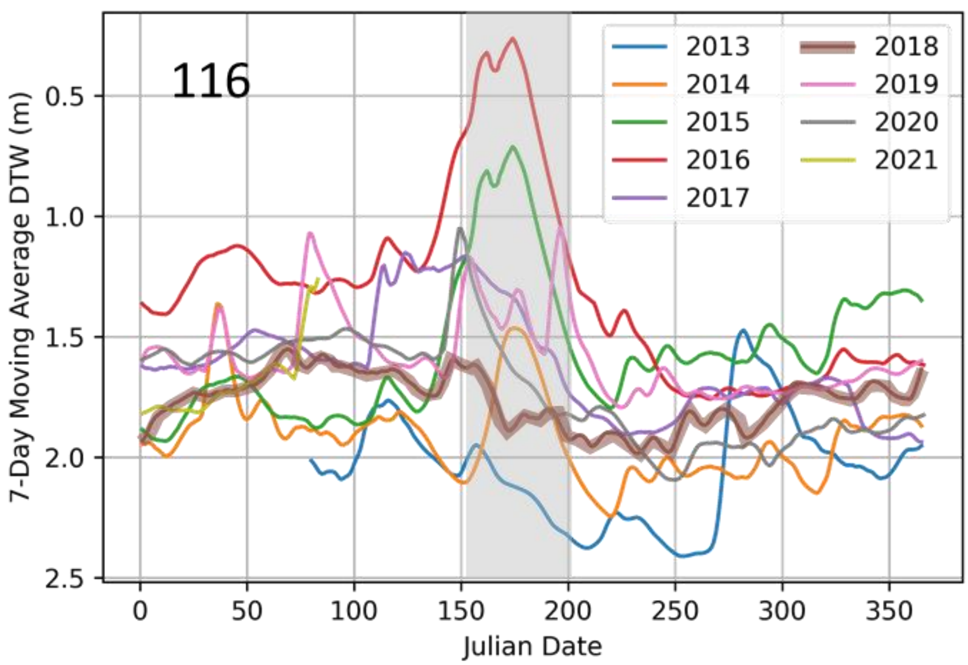


Figure 13 – Groundwater levels for well 116 highlighting low-water period during summer of 2018.

We used the model to estimate groundwater level changes at well 116 by inputting the range of calibration parameters and increasing river stage for approximate two weeks using a gaussian shaped discharge curve. During June of 2018, Platte River discharge ranged from just under 200 cfs to 2,500 cfs. Figure 14 demonstrates how a 1m increase in stage would raise groundwater levels by about 0.25m at well 116. A 1m stage increase corresponds to an increase in discharge of over 10,000 cfs based on stage-discharge curve reported for the USGS Kearney gage (USGS #: 06770200) (Figure 15). Regulatory flood stage at this gage station, 5 miles upstream of the Fox Site, is 7 feet, corresponding to a discharge of 17,915 cfs.

Similarly, we calculated groundwater depths assuming an 78-day surface recharge event, modeled by adding a constant value of 0.25 inches per day to the precipitation term, to estimate how surface recharge could raise groundwater levels. The Fox site is approximately 247 acres (1.07 km^2), assuming 0.25in day^{-1} were applied across the site, the delivery rate would need to be 1,200 gpm or 5.3 acre-ft per day. The model predictions show a similar increase in groundwater level to the stage change, but possibly an unrealistic retaining of water in the aquifer. This effect would need to be addressed if long-term groundwater level predictions were required. However, since the goal is to test actions for raising the L7th which is a relatively short-term maximum value, predictions may be adequate.

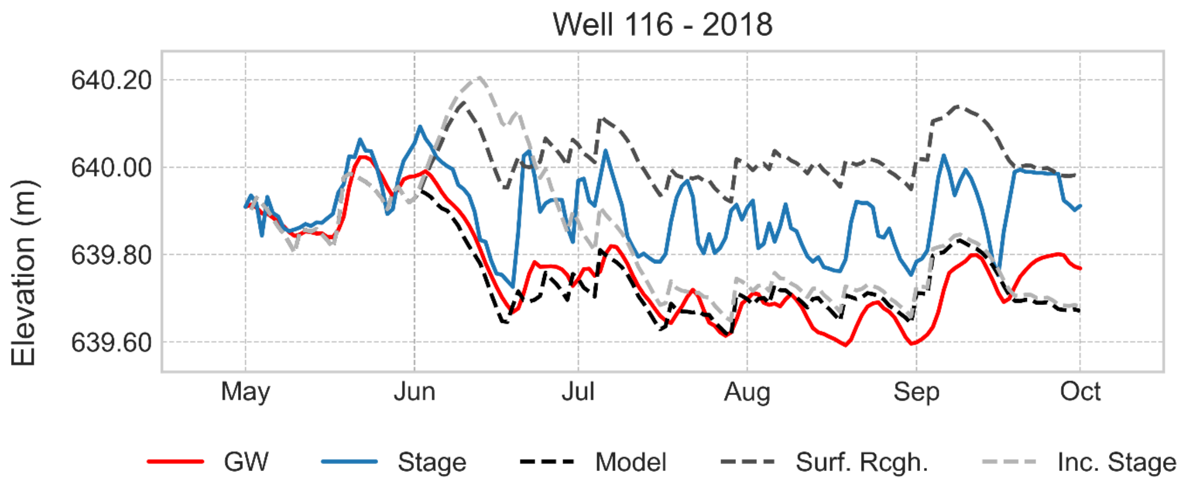


Figure 14– Plot demonstrating two management scenarios on predicted groundwater levels at well 116.

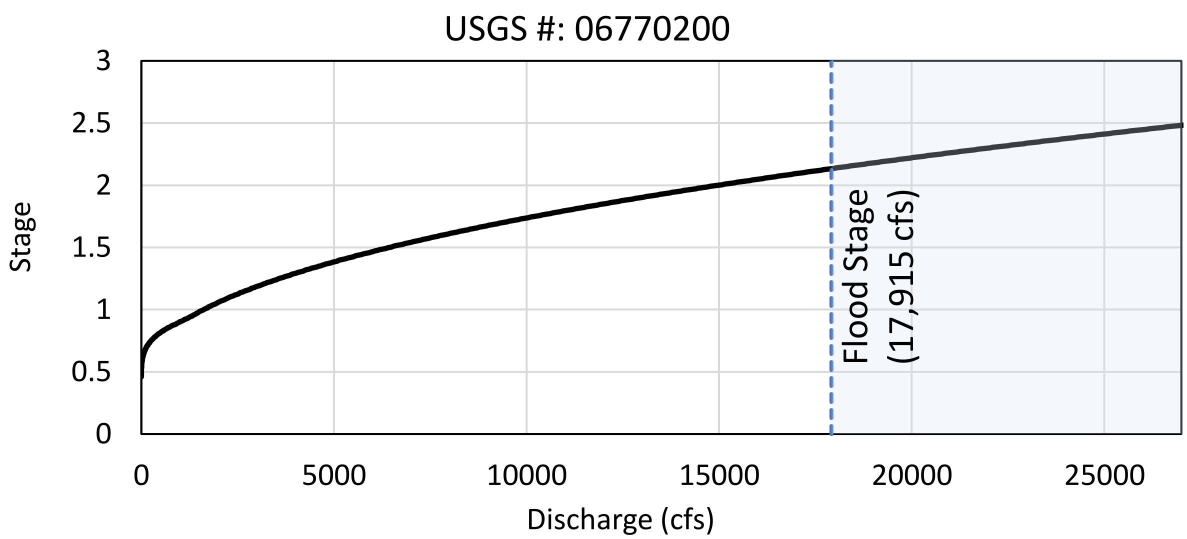


Figure 15 – Stage-discharge relationship for the USGS Kearney gage on the Platte River.

6 River-floodplain elevation analysis

One of the goals for this study was to generate results that would inform management at other CPRV wet meadow sites. In Sections 1-3 of this study, we demonstrate and discuss the influence of river stage and topography on groundwater levels. However, one aspect of this relationship that was not mentioned is the relationship between the absolute elevation of the river and ground surfaces. Previous studies identified strong correlations between temporally varying river stage and groundwater levels (e.g., Wesche et al., 1994), but correlations do not contain information about spatial relationships between the two. Intuitively, one would expect riparian zones where the ground and river surfaces were similar to have shallower groundwater levels compared to zones where the river is further below the adjacent floodplain. Similarly, if the river were to incise through time, one might expect adjacent groundwater levels to decrease.

If clear relationships between absolute river elevation and wet meadow ground surface elevations could be quantified for archetypal wet meadow sites, those relationships could be extended to other areas to inform predictions about hydrology even if field and well data were sparse or unavailable.

6.1 Methods

We produced differenced maps of river and ground surface elevations to test whether characteristic relationships were present. River stage surfaces were generated using SRH2D models for a range of relevant discharges (750-3500 cfs) that span representative discharges for the study period (2013-2020) and discharge targets designated by the USFWS as having potential benefits for wet meadows (Table 9). The modeled river stage was extrapolated laterally into the floodplain using a nearest neighbor interpolation method. The interpolated surface was then subtracted from the LIDAR ground surface elevation, producing a raster surface representing the elevation difference between ground surface and interpolated river stage.

Table 9 – USFWS target flows with wet meadow related benefits (PRRIP,2012)

USFWS Target Flow	USFWS Beneficial Effect	Hydrologic Condition	Target Flow (cfs)	Exceedance
February 15 to March 15 Pulse Flow	Maintain and enhance occurrence of soil moisture and pooled water during the growing season for lower tropic levels of the food chain in low grasslands and for biologically diverse communities in the ecosystem over the long term.	Normal and wet	3,350[1]	Exceeded in 75% of Years
		Dry	2,250[1]	Exceeded in 100% of Years
May 20 to June 20 Pulse Flow	Bring ground water levels in grasslands near to the soil surface in most areas of grassland and above the soil surface in some surface depressions in grasslands. One effect of this is to bring up soil organisms to near or above the soil surface for predation by migratory birds and other animals and provide pooled water for other aquatic food organisms.	Wet	3,700[1]	Exceeded in 33% of Years
		Normal	3,400[1]	Exceeded in 75% of Years

[1] Based on "Fixed Daily Target Flows" from Appendix E of the Program's Water Plan Reference Materials (Program Water Plan, Attachment 5, Section 11).

6.2 Results and Discussion

Results for two locations with vegetated islands (containing Binfield and Fox study sites) are presented in Figures 16 through 19 for discharges of 750 cfs, 1,200 cfs, 2,000 cfs, and 3,500 cfs respectively. Differenced elevation values range from -11 ft to 57 ft, with negative values indicating areas where ground surface is below the river surface elevation. The abrupt change in differenced values in the western Binfield site is likely due to a LIDAR inconsistency and will be addressed in future iterations of this method.

Visual observations of resulting maps reveal important patterns. Off-channel zones within 1km of the channel tend to have more similar elevations to the river surface, whereas inland areas tend to be higher in elevation. Off-channel zones where north and south channels converge also tend to be closer in elevation to the river surface. Relic channels, as indicated by channel patterns within inland areas, are typically within 3ft (1m) of the river surface, even at lower flows (750cfs). Notably, the results contain very similar patterns to those identified in previous sections of this report. Zoomed in maps of previously discussed sites, Fox, Binfield, and Crane meadows, are reproduced along with elevation difference results in Figures 20 through 23. For reference, results from Section 4 for L7th DTGW surfaces are also included in Figure 24. L7th maps were produced through extensive, long-term data collection and subsequent processing, yet patterns are remarkably similar to the differenced elevation results, particularly for 2,000 and 3,500 cfs, which required no field data collection and minimal data processing.

The utility of this method can also be seen in the distinct signature of the Crane Meadows site. As previously demonstrated, and shown by Wesche et al. (1994), the Crane Meadows site is an especially "wet" wet meadow site, affording it a range of characteristics that site managers have tried to mimic at other sites through restoration and management. The differenced elevation surfaces reveal that a significant proportion of the site has ground surface elevations below river stage, even at the lowest analyzed flow of 750cfs. The extent to which this occurs is unmatched in any other region of the study area, making Crane Meadows a truly unique site. It also highlights the fact that restoration activities at other sites may never be able to recreate characteristics of Crane Meadows, due to the site's unique topography.

While qualitative (for now), this analysis presents a method for making predictions about site hydrology that are relevant for wet meadows, without the need for extensive field data collection. From the results, one can clearly compare the likelihood of groundwater levels being higher or lower across sites. By performing the analysis at existing wet meadows, as we have done, locations with similar river-ground surface elevation relationships can be identified throughout the CPRV as viable wet meadow sites. Several grassland sites currently managed by the Program may even be considered for future restoration or management as wet meadows using this method. Figure 25 includes the area around the Binfield site, showing additional tracts that exhibit similar hydrologic relationships as the Binfield site.

Future work might look at other aspects of channel geometry and elevation in relation to wet meadow occurrence and hydrology. The sites considered in this analysis all occurred on vegetated islands within the Platte River, but it would be useful to consider this analysis along consolidated segments of the channel and floodplain. Additionally, areas where channel incision has occurred may have different channel-floodplain elevation relationships than non-incised regions.

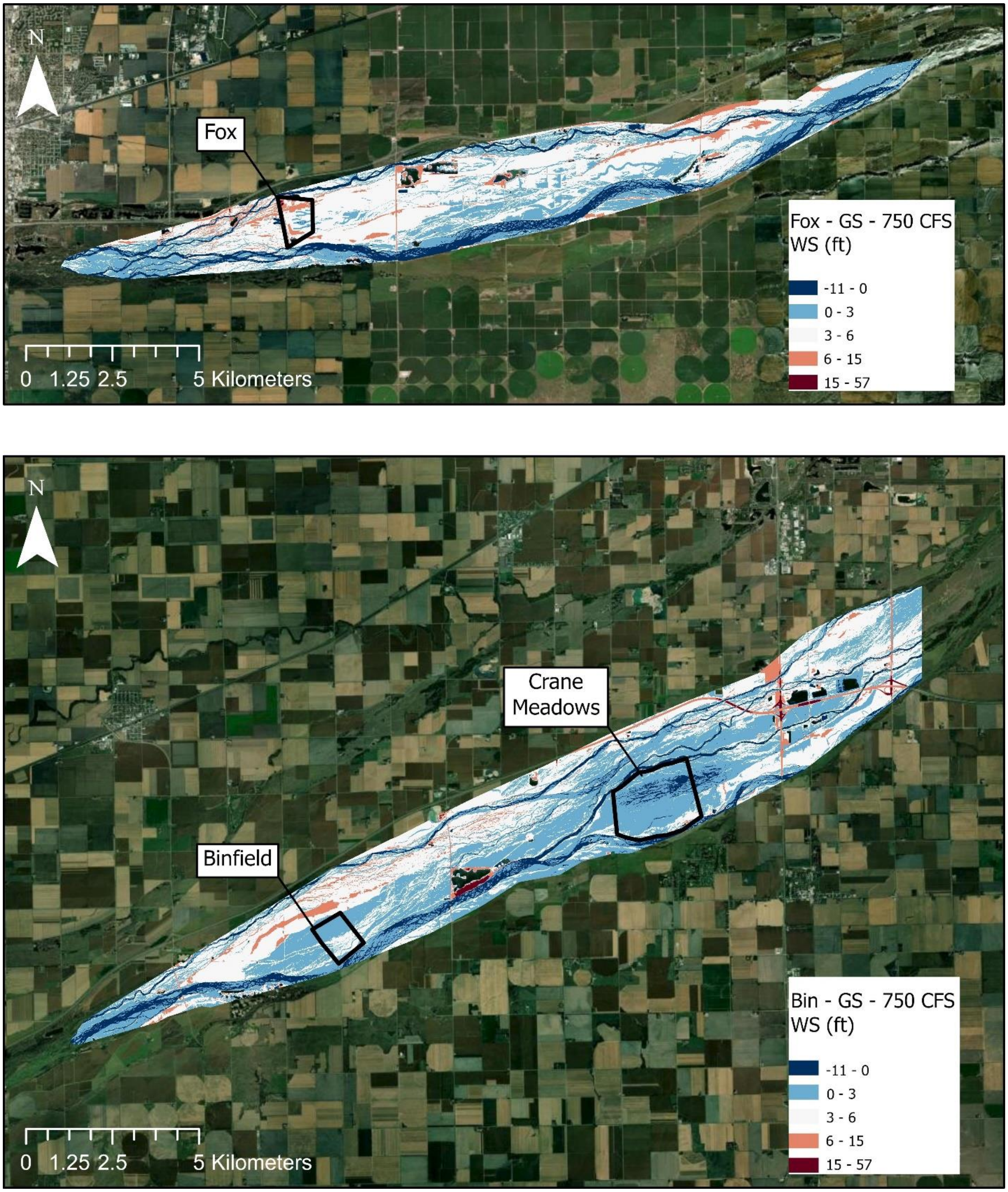
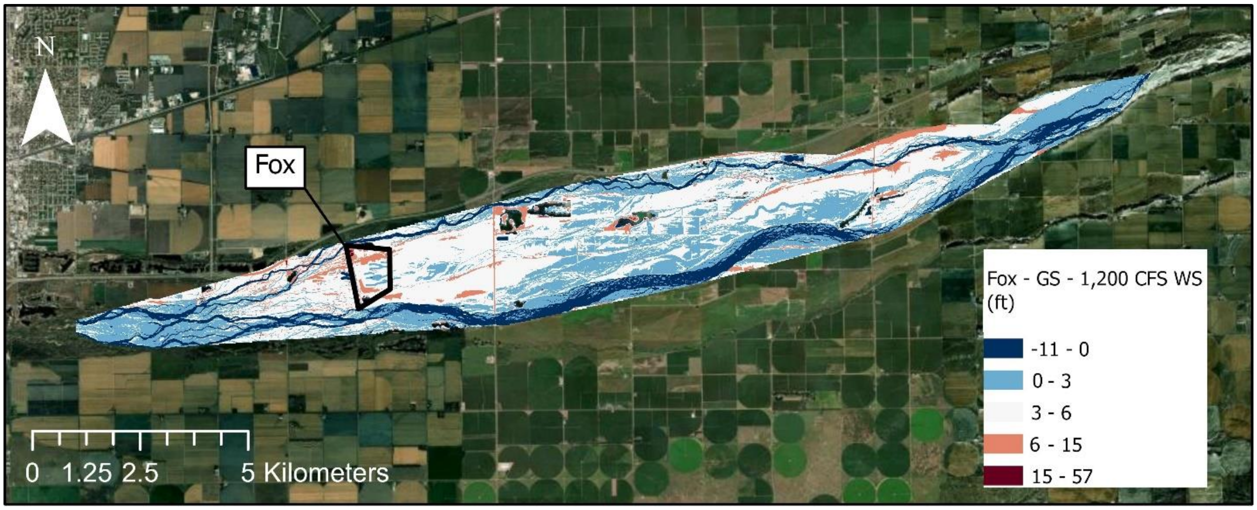


Figure 16 – Differenced river and ground surface elevation raster for discharge of 750 cfs.



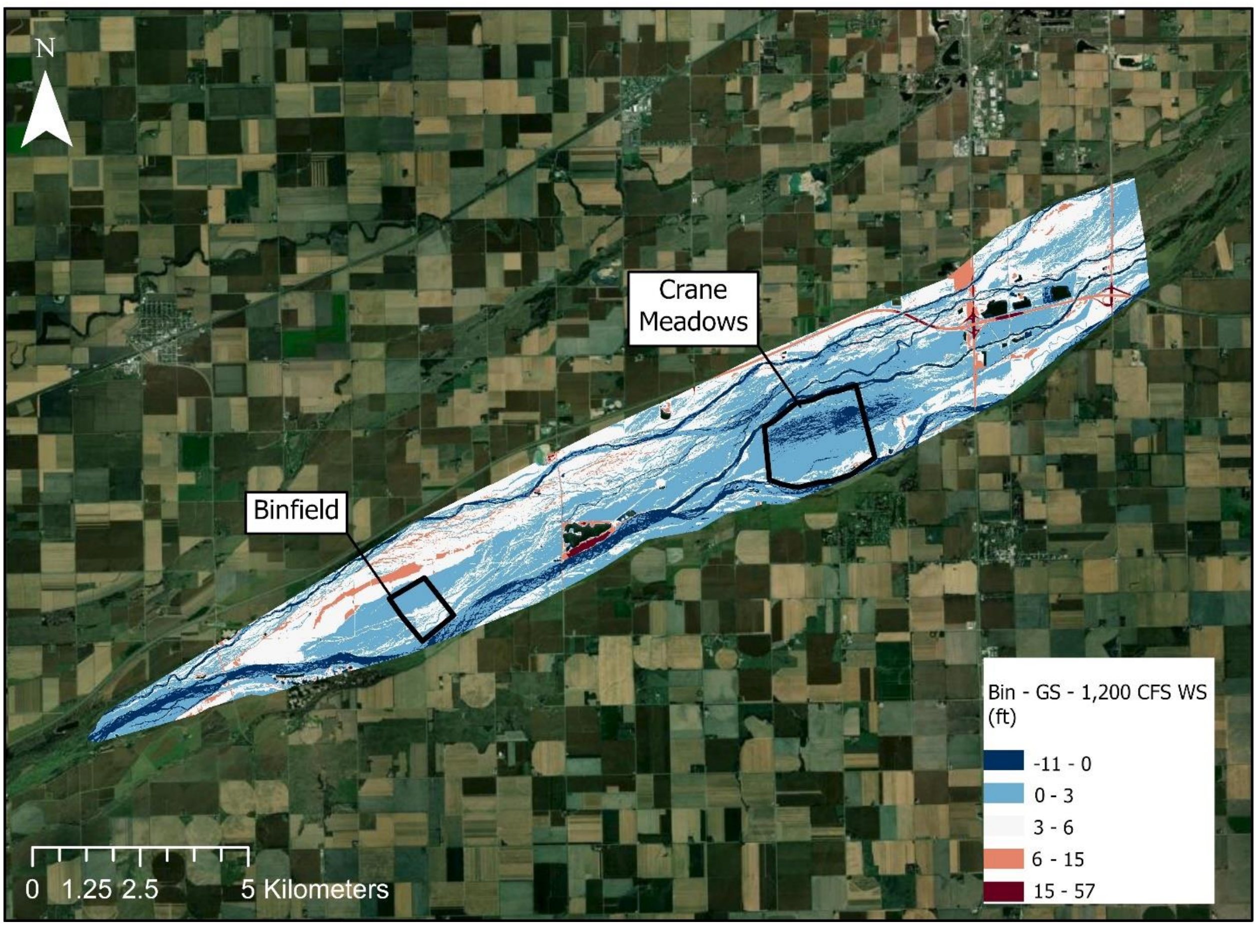
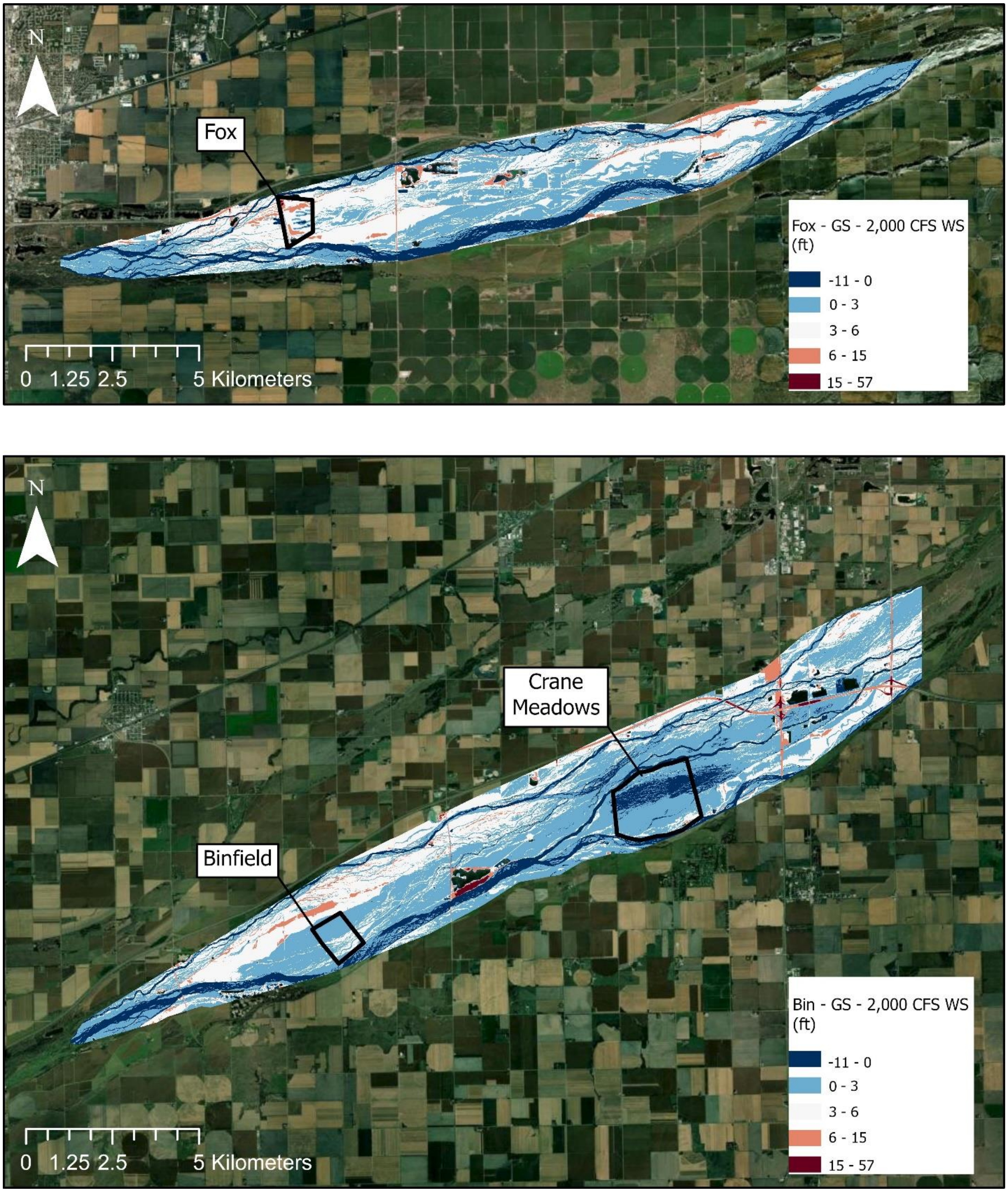


Figure 17 – Differenced river and ground surface elevation raster for discharge of 1200 CFS.



849

850 *Figure 18 – Differenced river and ground surface elevation raster for discharge of 2000 cfs.*

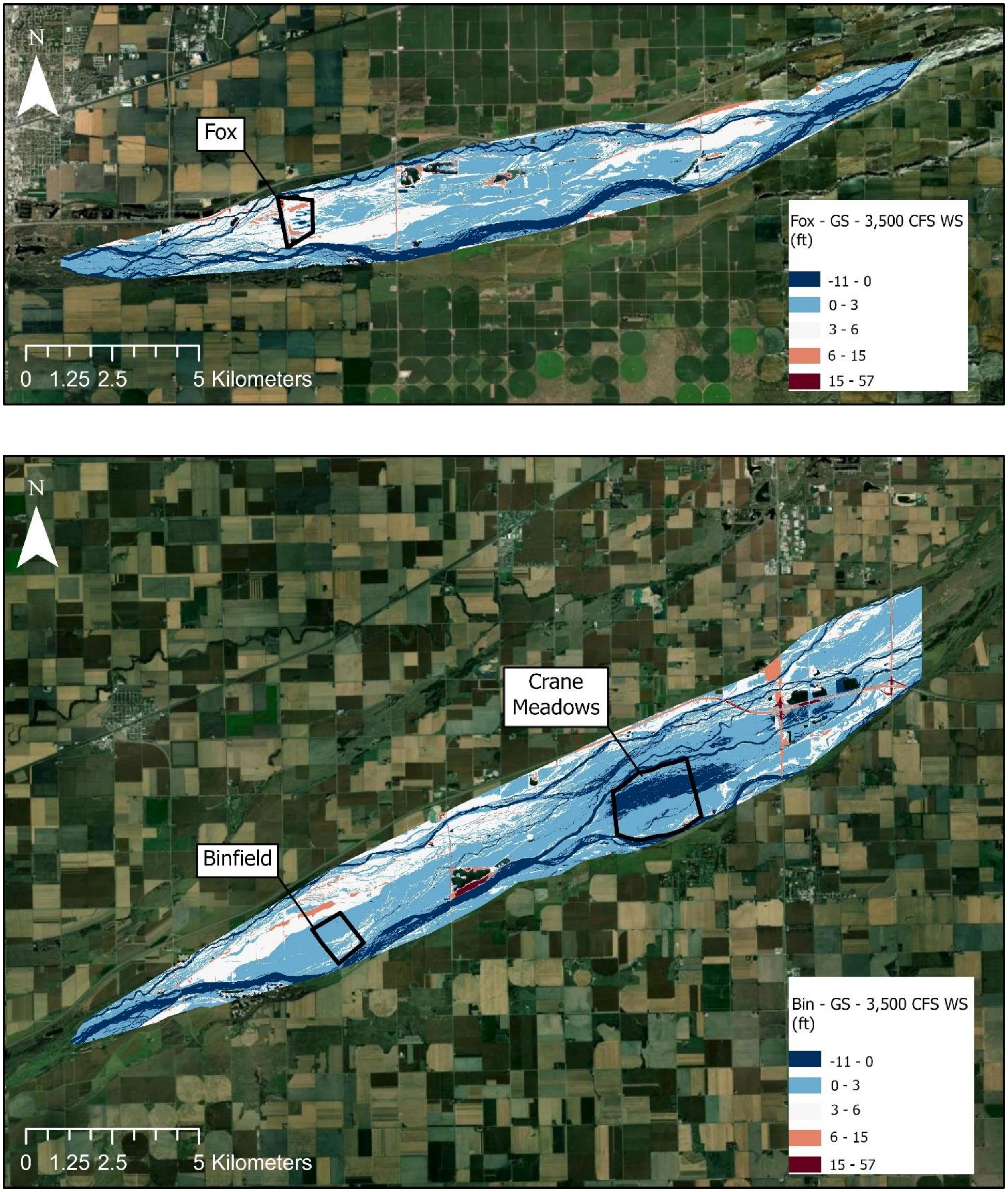


Figure 19 – Differenced river and ground surface elevation raster for discharge of 3,500 cfs.

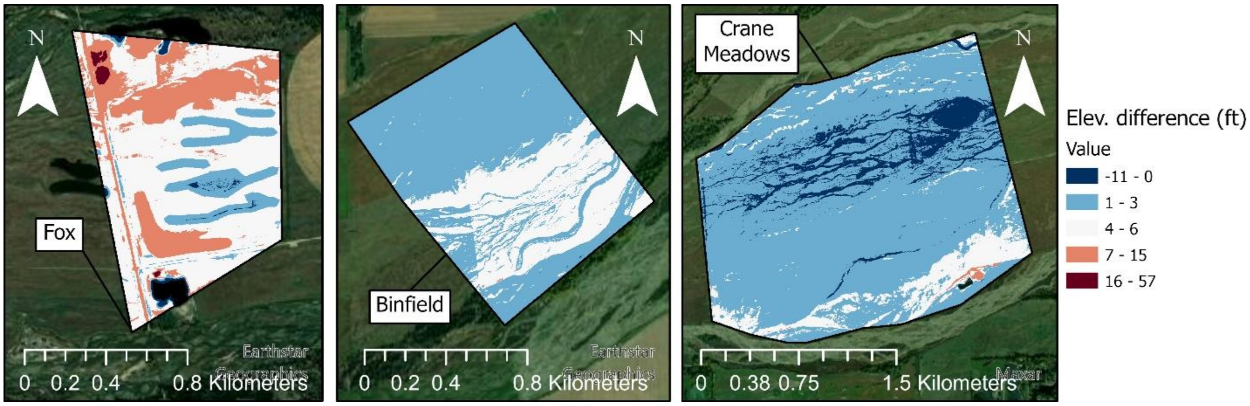


Figure 20 – Zoomed in differenced river and ground surface elevation raster at Fox, Binfield, and Crane Meadows at discharge of 750 cfs.

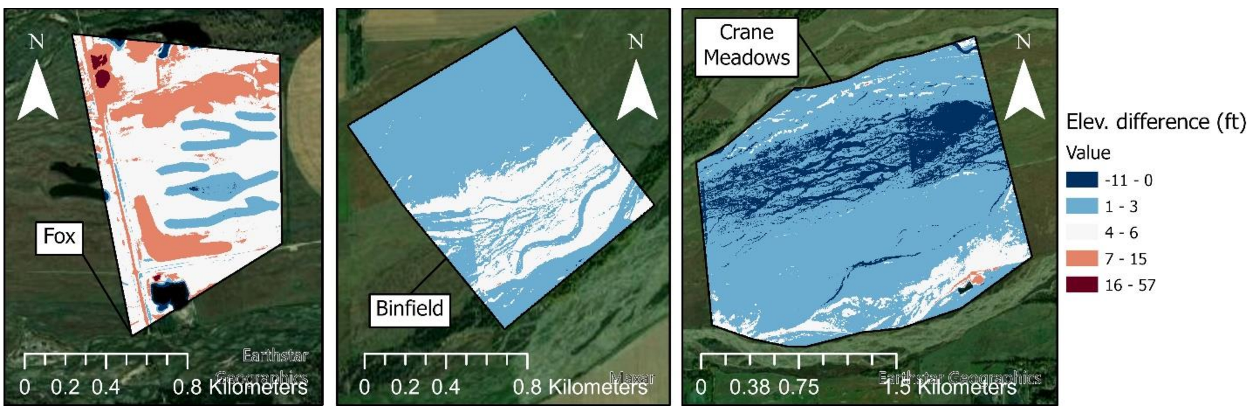


Figure 21 – Zoomed in differenced river and ground surface elevation raster at Fox, Binfield, and Crane Meadows at discharge of 1,200 cfs.

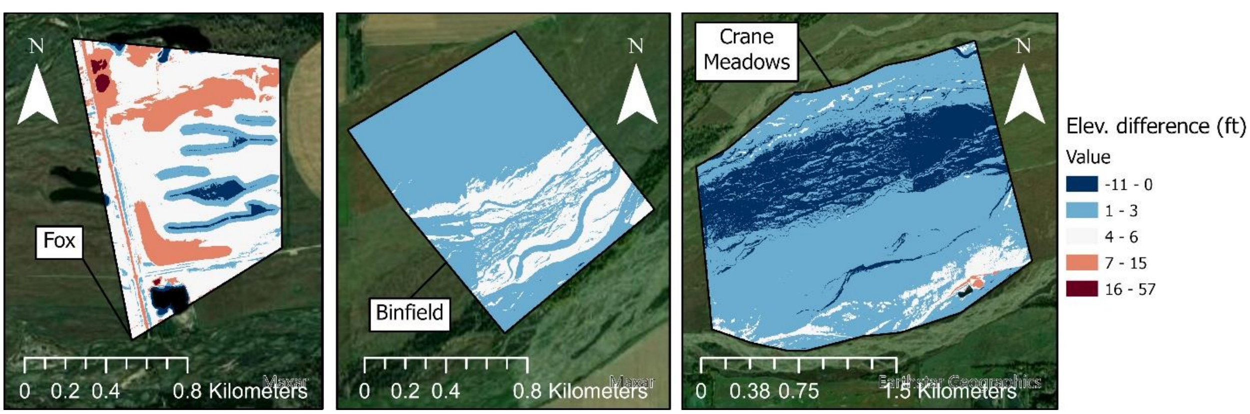


Figure 22 – Zoomed in differenced river and ground surface elevation raster at Fox, Binfield, and Crane Meadows at discharge of 2,000 cfs.

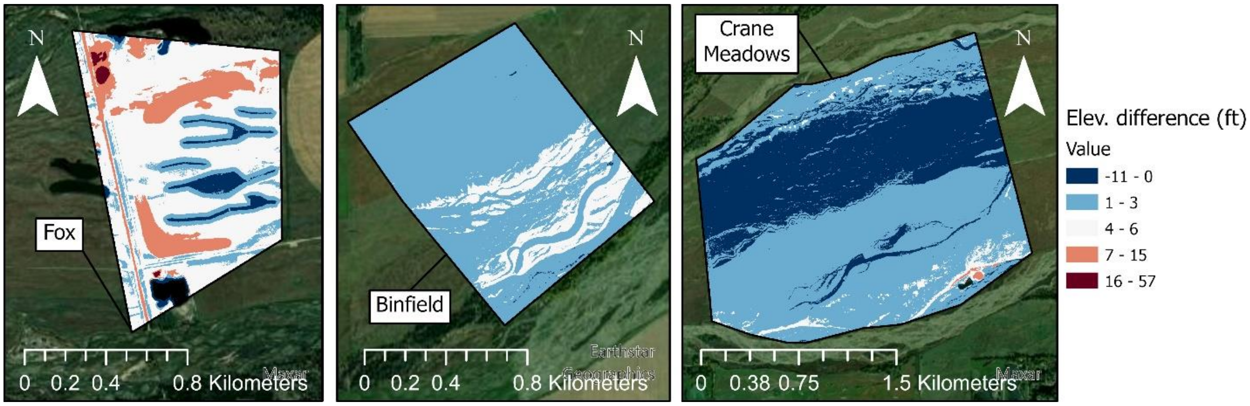


Figure 23 – Zoomed in differenced river and ground surface elevation raster at Fox, Binfield, and Crane Meadows at discharge of 3,500 cfs.

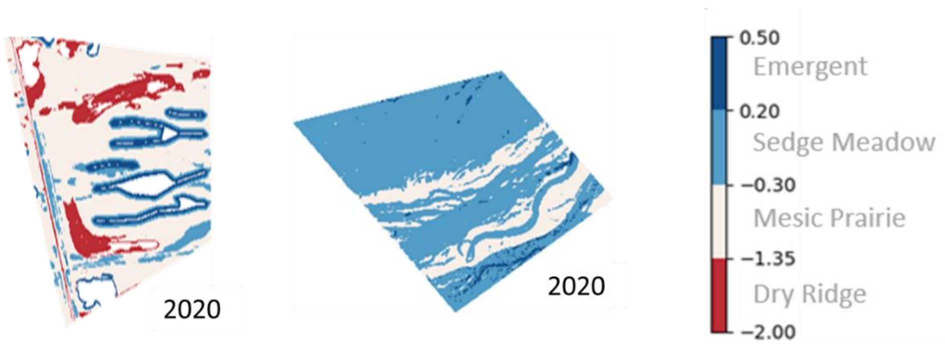
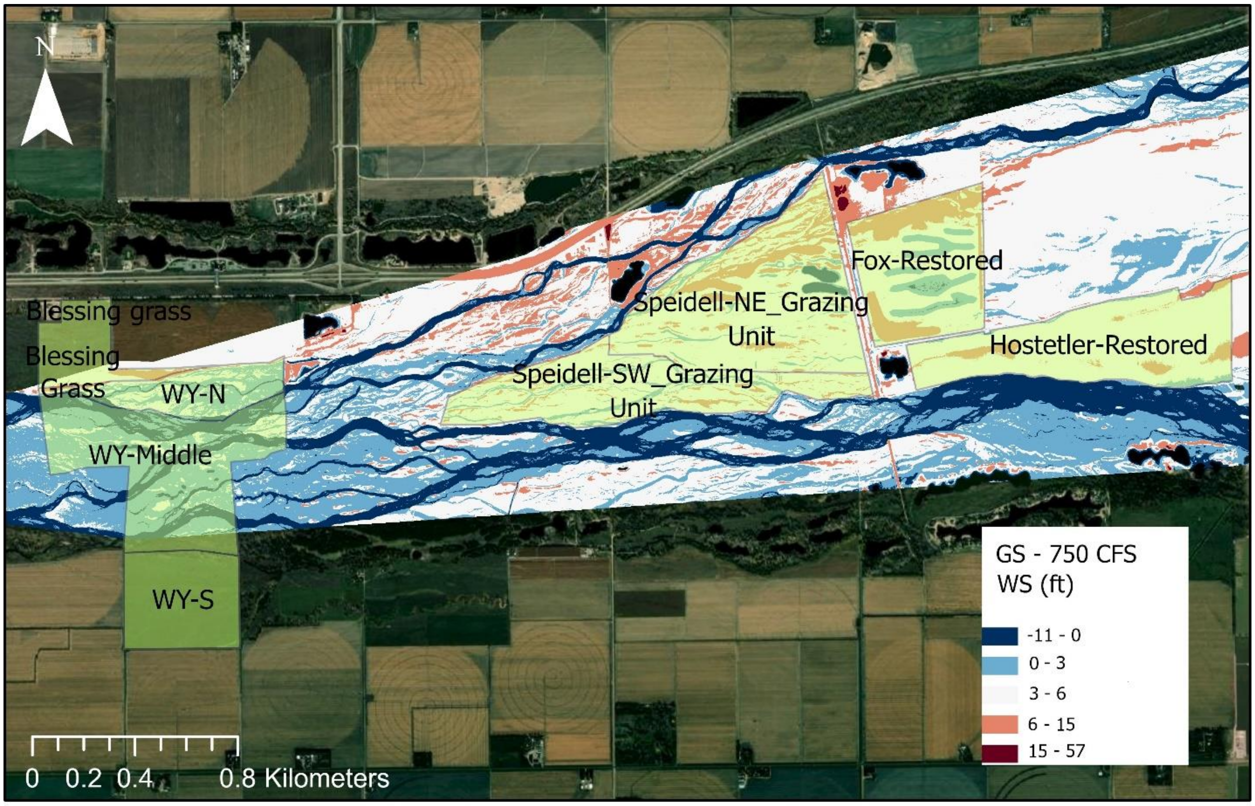
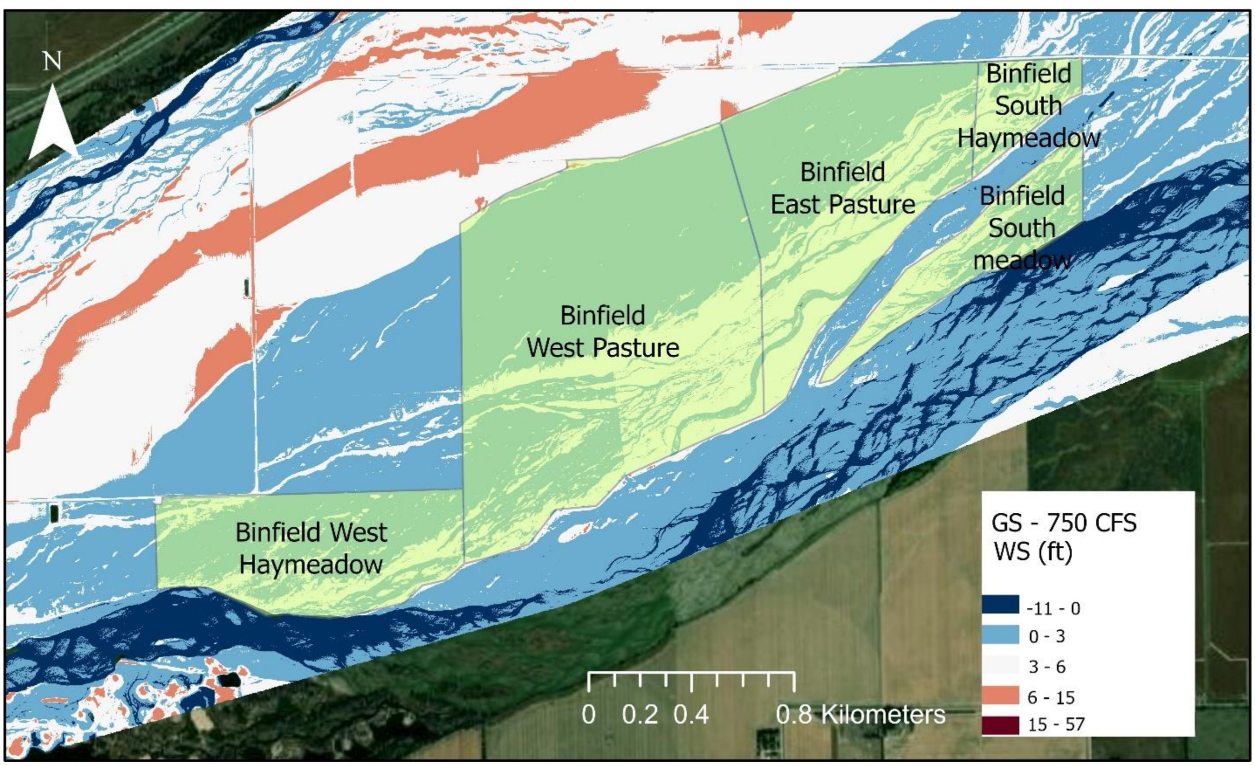


Figure 24 – L7th DTGW raster surfaces for 2020 duplicated from Section 4


N
Blessing grass
Blessing Grass
WY-N
WY-Middle
WY-S
Speidell-NE_Grazing Unit
Speidell-SW_Grazing Unit
Fox-Restored
Hostetler-Restored
GS - 750 CFS
WS (ft)
-11 - 0
0 - 3
3 - 6
6 - 15
15 - 57
0 0.2 0.4 0.8 Kilometers


N
Binfield South Haymeadow
Binfield East Pasture
Binfield South meadow
Binfield West Pasture
Binfield West Haymeadow
GS - 750 CFS
WS (ft)
-11 - 0
0 - 3
3 - 6
6 - 15
15 - 57
0 0.2 0.4 0.8 Kilometers

870

Figure 25 – Map of differenced raster surfaces along with other grassland sites managed by the Program.

7 References

Andualem, T. G., Demeke, G. G., Ahmed, I., Dar, M. A., & Yibeltal, M. (2021). Groundwater recharge estimation using empirical methods from rainfall and streamflow records. Journal of Hydrology: Regional Studies, 37, 100917.

Brinley Buckley, E. M., A. J. Caven, J. D. Wiese, and M. J. Harner. (2021). Assessing the hydroregime of an archetypal riverine wet meadow in the central Great Plains using time-lapse imagery. Ecosphere 12(11):e03829. 10.1002/ecs2.3829

Chen, X. (2007). Hydrologic connections of a stream–aquifer-vegetation zone in south-central Platte River valley, Nebraska. Journal of Hydrology 333:554–568.

Cooperative Agreement. (1997). Cooperative agreement for Platte River research and other efforts relating to endangered species habitats along the central Platte River, Nebraska. Signed July 1, 1997 by the Secretary of Interior and Governors of Nebraska, Wyoming and Colorado, USA.

Currier, P. J., & Davis, C. A. (2000). The Platte as a prairie river: a response to Johnson and Boettcher. Great Plains Research, 69-84.

Currier, P., & Ziewitz, J. (1985). Application of a Sandhill Crane Model to the Management of Habitat Along the Platte River. In: Crane Workshop, 1985. 315-325.

Currier, P. J., G. R. Lingle, and J. G. VanDerwalker. (1985). Migratory bird habitat on the Platte and North Platte rivers in Nebraska. Pages 177. Platte River Whooping Crane Critical Habitat Maintenance Trust, Grand Island, Nebraska, USA.

Currier, P. J. (1989). Plant species composition and groundwater levels in a Platte River wet meadow. Proceedings of the North American Prairie Conference11:19–24.

Currier, P. J. (1994). Restoration of functioning wet meadows on the Platte River—experimenting with reseeding, constructed wetlands, and hydrology. Contract, 14-16.

Davis, C. A., J. E. Austin, and D. A. Buhl. (2006). Factors influencing soil invertebrate communities in riparian grasslands of the central Platte River floodplain. Wetlands 26: 438– 454.

Derby, C., & Strickland, D. (2001). Platte River Cooperative Agreement and proposed program: efforts to protect, restore, and manage habitat for Whooping Cranes, Least Terns, and Piping Plovers.

Eschner, T. R., Hadley, R. F., & Crowley, K. D. (1981). Hydrologic and morphologic changes in channels of the Platte River basin: a historical perspective (Vol. 81, No. 1125). US Geological Survey.

Federal Interagency Committee for Wetland Delineation. (1989). Federal manual for identifying and delineating jurisdictional wetlands. U.S. Army Corps of Engineers, U.S. Environmental Protection Agency, U.S. Fish and Wildlife Service, and U.S.D.A. Soil Conservation Service, Washington, DC, USA. Cooperative Technical Publication.

Franke, (2006), Survey of Herpetofauna of the Platte River Whooping Crane Maintenance Trust, With Notes on Management Concerns for the Plains Leopard Frog (Rana blairi). (DELETE)

Freeze, R. A., & Cherry, J. A. (1979). Groundwater. Prentice-hall.

Gerla, P.J. (1992). The relationship of water-table changes to the capillary fringe, evapotranspiration, and precipitation in intermittent wetlands. Wetlands 12(2) : 9 1-98.

Glover, R. E. (1964). Ground-water movement (Vol. 31). US Department of the Interior, Bureau of Reclamation.

Henszey, R. J., Pfeiffer, K. & Keough, J. R. (2004). Linking surface and ground water levels to riparian grassland and species along the Platte River in central Nebraska, USA. Wetlands, 24, 665-687.

HPRCC, (2021) High Plains Regional Climate Center, accessed September 2021, link

Hurr, T. (1983). Ground-Water Hydrology of Mormon Island Crane Meadows Wildlife Area Near Grand Island Hall County, Nebraska. In: 1277, U. S. G. S. P. P. (ed.) Hydrologic and Geomorphic Studies of the Platte River Basin.

Joyce, C. B., M. Simpson, and M. Casanova. (2016). Future wet grasslands: ecological implications of climate change. Ecosystem Health and Sustainability 2:e01240.

Krapu, G. L. (1981). Platte River Ecology Study, Special Research Report. . In: U.S. DEPARTMENT OF THE INTERIOR, U. S. G. S. (ed.). Jamestown, ND: USGS.

Loheide, S. P., & Gorelick, S. M. (2007). Riparian hydroecology: a coupled model of the observed interactions between groundwater flow and meadow vegetation patterning. Water Resources Research, 43(7).

Lurtz, M. R., Morrison, R. R., Gates, T. K., Senay, G. B., Bhaskar, A. S., & Ketchum, D. G. (2020). Relationships between riparian evapotranspiration and groundwater depth along a semiarid irrigated river valley. Hydrological Processes, 34(8), 1714-1727.

Meyer, C. K., & Whiles, M. R. (2008). Macroinvertebrate communities in restored and natural Platte River slough wetlands. J.N.Am.Benthol.Soc., 27, 626-639.PRRIP (2012). Platte River Wet meadow Geohydrology and Management through flow releases

Nebraska Department of Transportation, (2020), Wetland and Water Resource Procedure Document. https://dot.nebraska.gov/media/113469/wrrpd_manual_combined.pdf

PRRIP (2012). Platte River Wet Meadow Geohydrology and Management through Flow Releases. ED Office White Paper. Link

PRRIP (2015). Wet Meadows Hydrology Monitoring Approach Chapters. Link

Rosenberry, D. O., & Winter, T. C. (1997). Dynamics of water-table fluctuations in an upland between two prairie-pothole wetlands in North Dakota. Journal of Hydrology, 191(1-4), 266-289.

Sanders, G. (2001). Ground Water and River Flow Analyses Appendices. Technical Report Appendices of the Platte River EIS Team U.S. Department of the Interior, Bureau of Reclamation, Fish and Wildlife Service.

Sidle, J. G., E. D. Miller, and P. J. Currier. (1989). Changing habitats in the Platte River Valley of Nebraska. Prairie Naturalist 21:91–104.

Tiner, R. W. (2016). Wetland indicators: A guide to wetland formation, identification, delineation, classification, and mapping. CRC press.

US Army Corps of Engineers (USACE). (1987). Wetlands Delineation Manual

U.S. Department of the Interior Bureau of Reclamation (USBR). (2004) The Platte River Channel: History and Reclamation. Technical Service Center, Denver, Colorado.

Weir, E., Chavez Ramirez, F. (2011). Wet Meadow Literature and Information Review. Prepared for the Platte River Recovery Implementation Program. Link

Wesche, T. A., Q. D. Skinner, and R. J. Henszey. (1994). Platte River wetland hydrology study: final report. Wyoming Water Resources Center Technical Report. University of Wyoming, Laramie, Wyoming, USA.

Williams, G. P. (1978). The case of the shrinking channels: the North Platte and Platte Rivers in Nebraska (Vol. 781). US Department of the Interior, Geological Survey.

Williams, A. P., Seager, R., Abatzoglou, J. T., Cook, B. I., Smerdon, J. E., & Cook, E. R. (2015). Contribution of anthropogenic warming to California drought during 2012–2014. Geophysical Research Letters, 42(16), 6819-6828.

Whiles, M. R., and B. S. Goldowitz. (2001). Hydrologic influences on insect emergence production from central Platte River wetlands. *Ecological Applications* **11**: 1829– 1842.

Whiles, M. R., and B. S. Goldowitz. (2005). Macroinvertebrate communities in central Platte River wetlands: patterns across a hydrologic gradient. *Wetlands* **25**: 462– 472.

WU, W. (2003). Riverine Landscape of the Middle Platte River: Hydrological Connectivity and Physicochemical Heterogeneity. Doctorate Partial Degree Fullfillment, University of Nebraska at Lincoln.

Yue, W., Wang, T., Franz, T. E., & Chen, X. (2016). Spatiotemporal patterns of water table fluctuations and evapotranspiration induced by riparian vegetation in a semiarid area. Water Resources Research, 52(3), 1948-1960.

Appendix A – Ground surface – groundwater elevation relationships

Ground surface elevation strongly controls the depth to groundwater at a given location. Locations with a higher ground surface have a greater depth to the groundwater table, and vice versa. This is easily visualized with plots of median groundwater depths and ground surface elevation at the Binfield site (Figure A1).

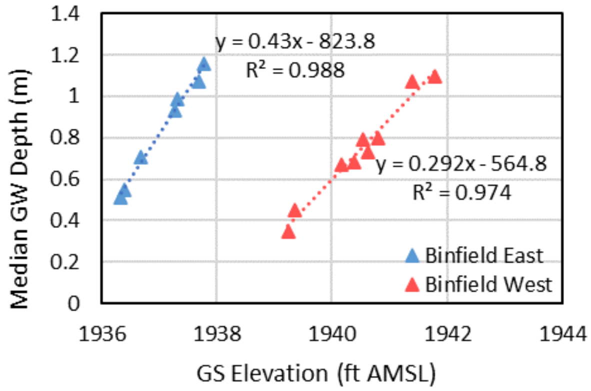


Figure A1 – Plot demonstrating strong-positive relationship between ground surface elevation and median groundwater depth (m) for Fox and Binfield transects.

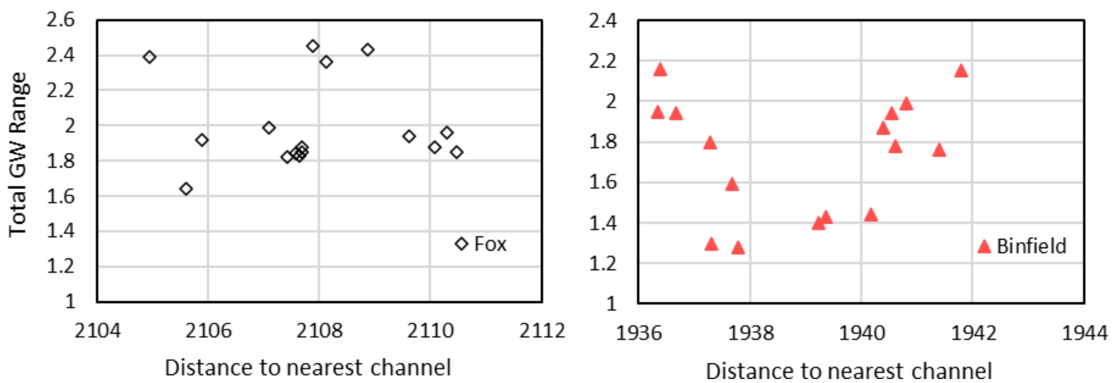


Figure A2 – Plot of distance from channel vs study period GW range.

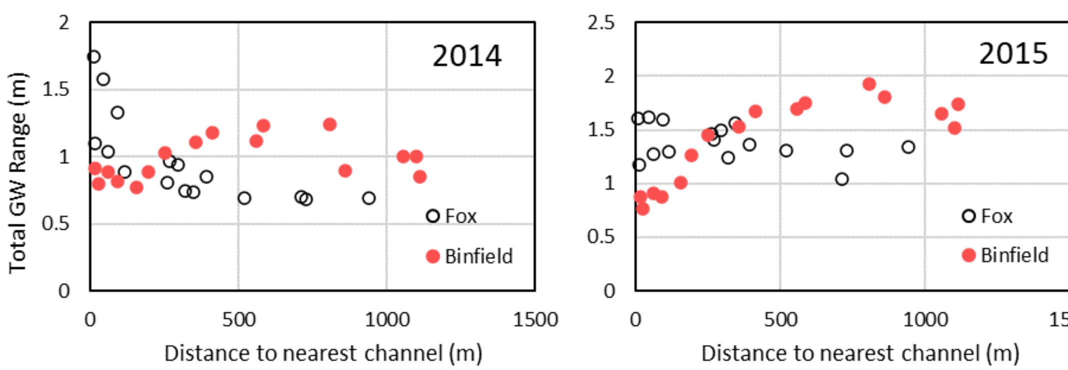
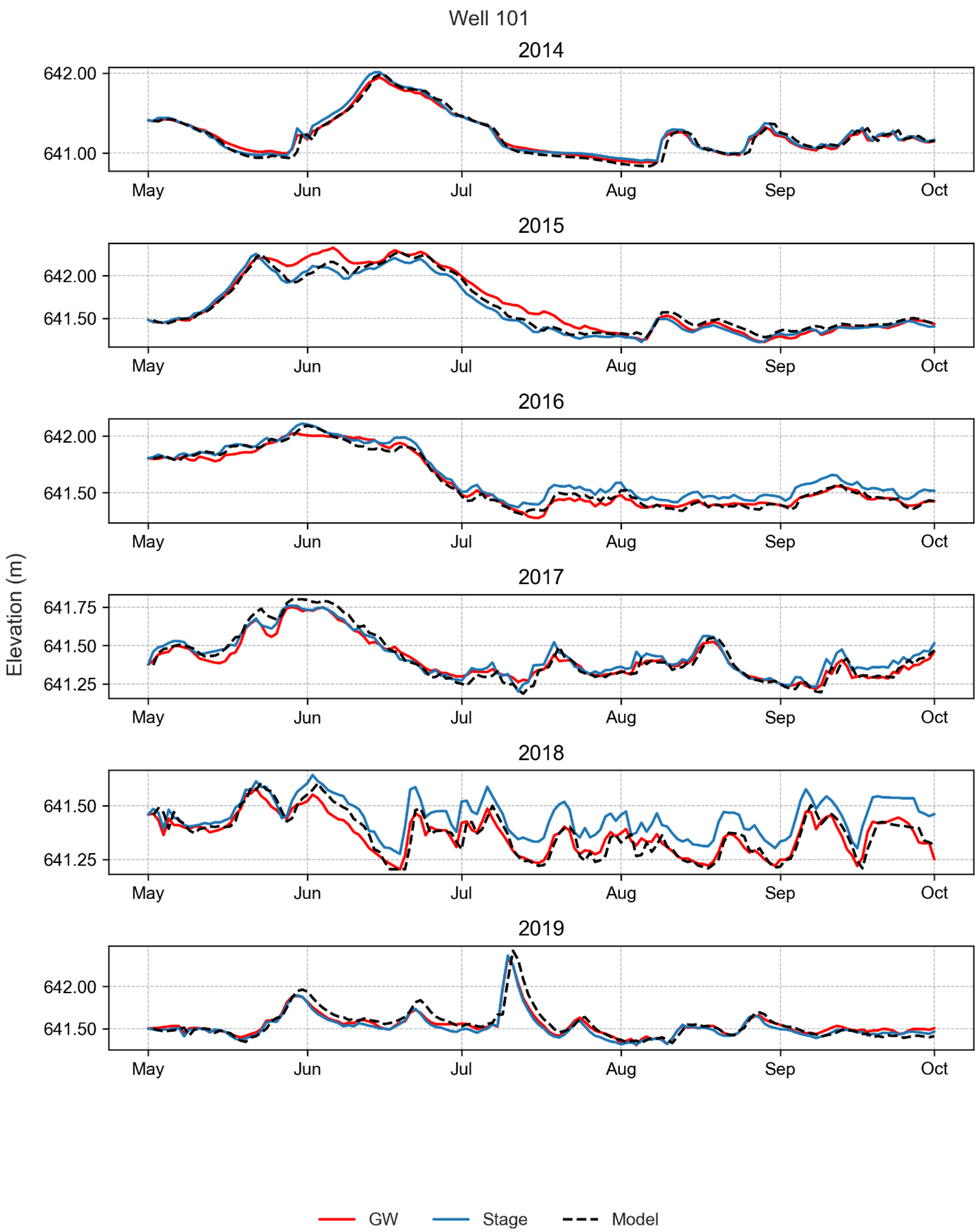


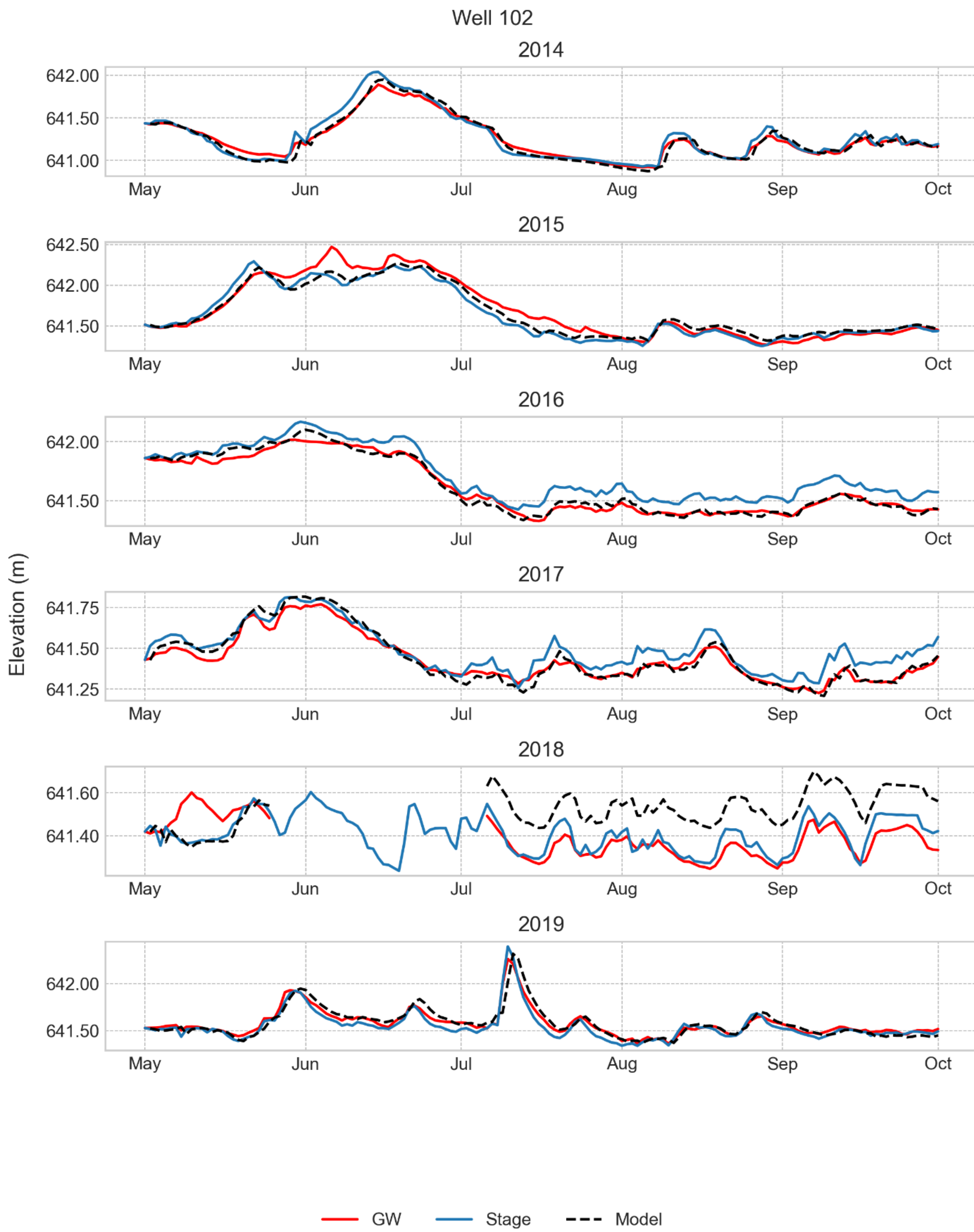
Figure A3 – Plot showing relationship between annual range in groundwater levels and the distance to the channel for a dry year (2014) and wet year (2015).

981 ## Appendix B – Model calibration results

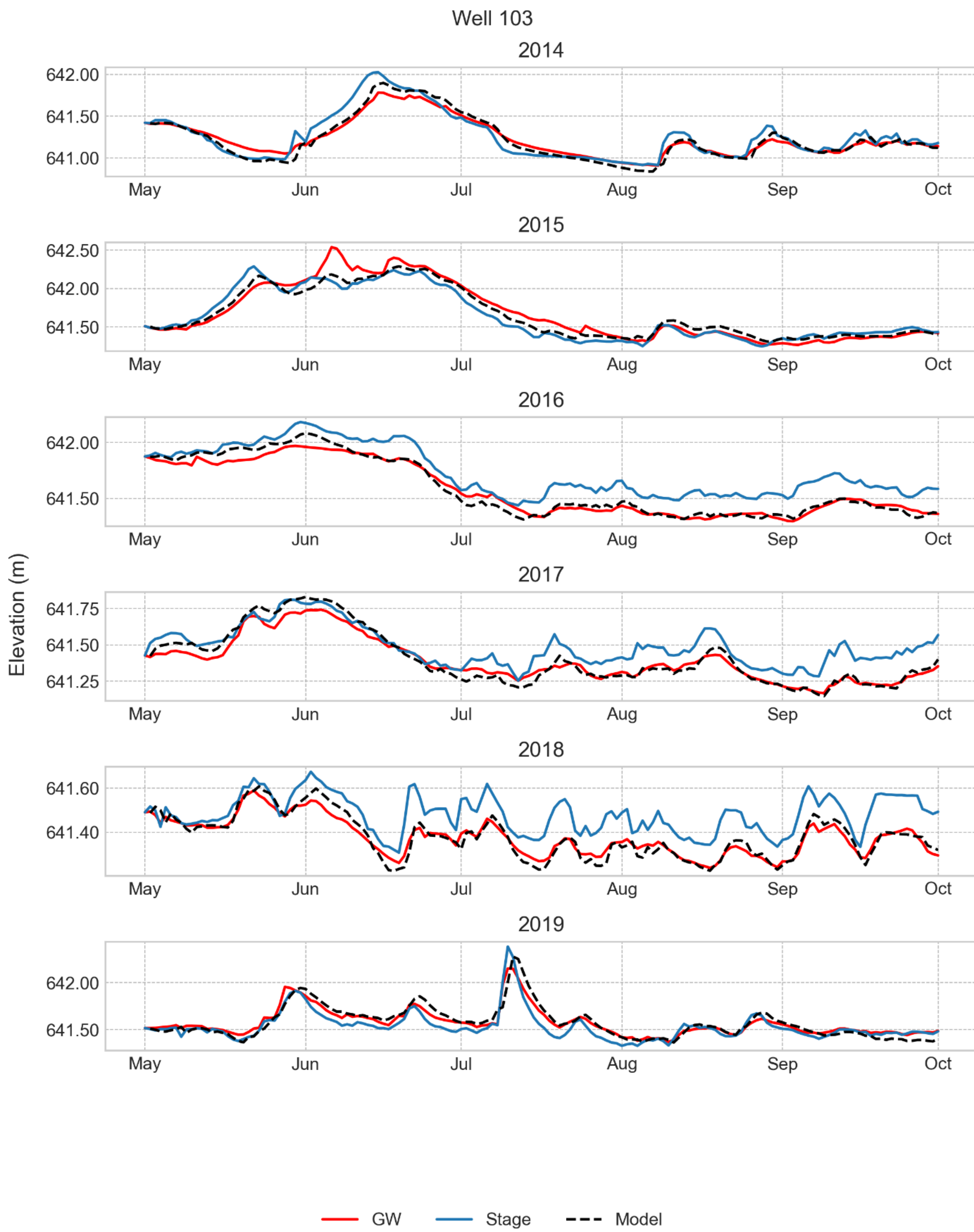


982

983


Well 102
2014
642.00
641.50
641.00
May
Jun
Jul
Aug
Sep
Oct
2015
642.50
642.00
641.50
2016
642.00
641.50
2017
641.75
641.50
641.25
2018
641.60
641.40
2019
642.00
641.50
Elevation (m)
GW
Stage
Model

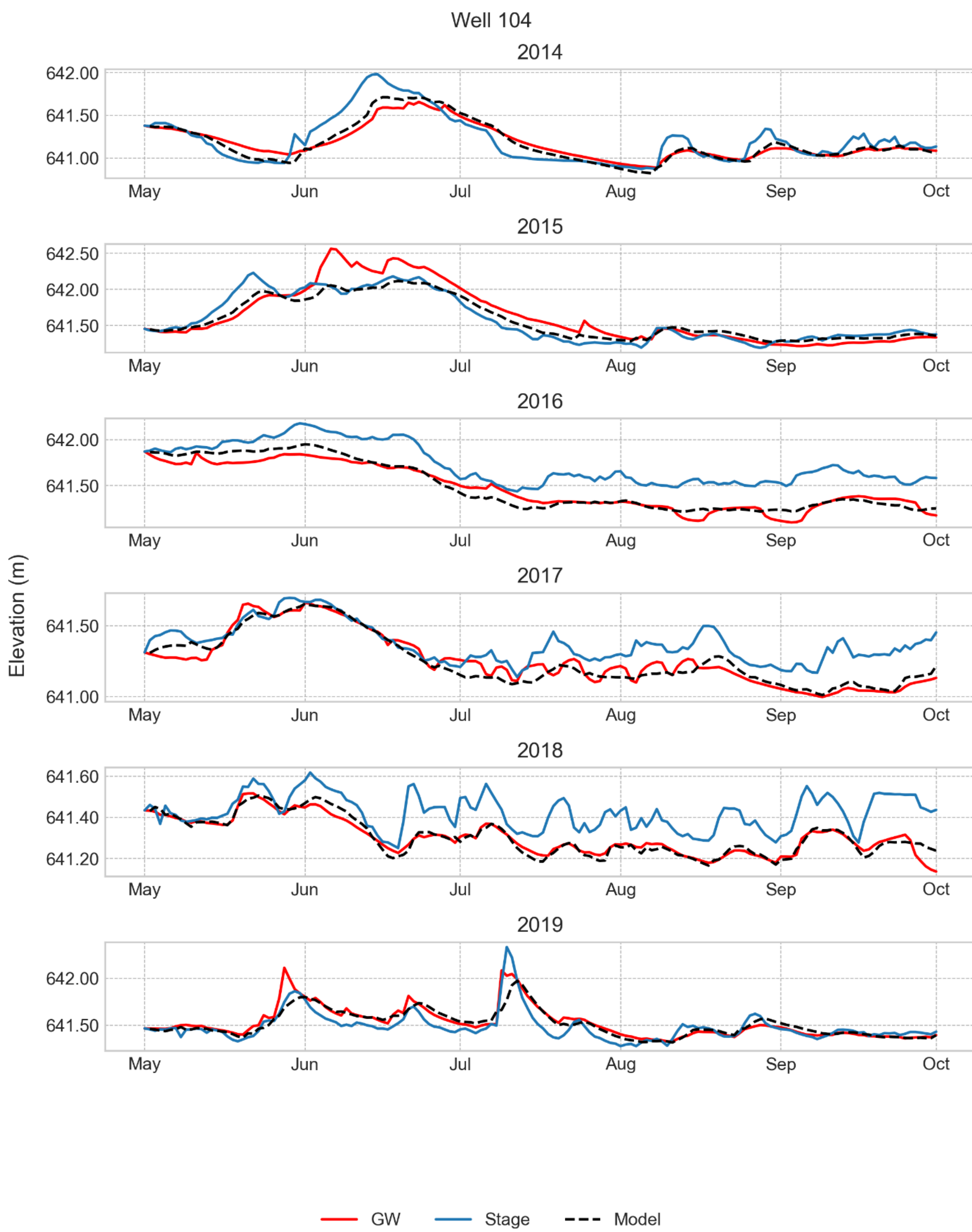
984


Well 103
2014
2015
2016
2017
2018
2019
Elevation (m)
642.00
641.50
641.00
642.50
641.75
641.25
641.60
641.40
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

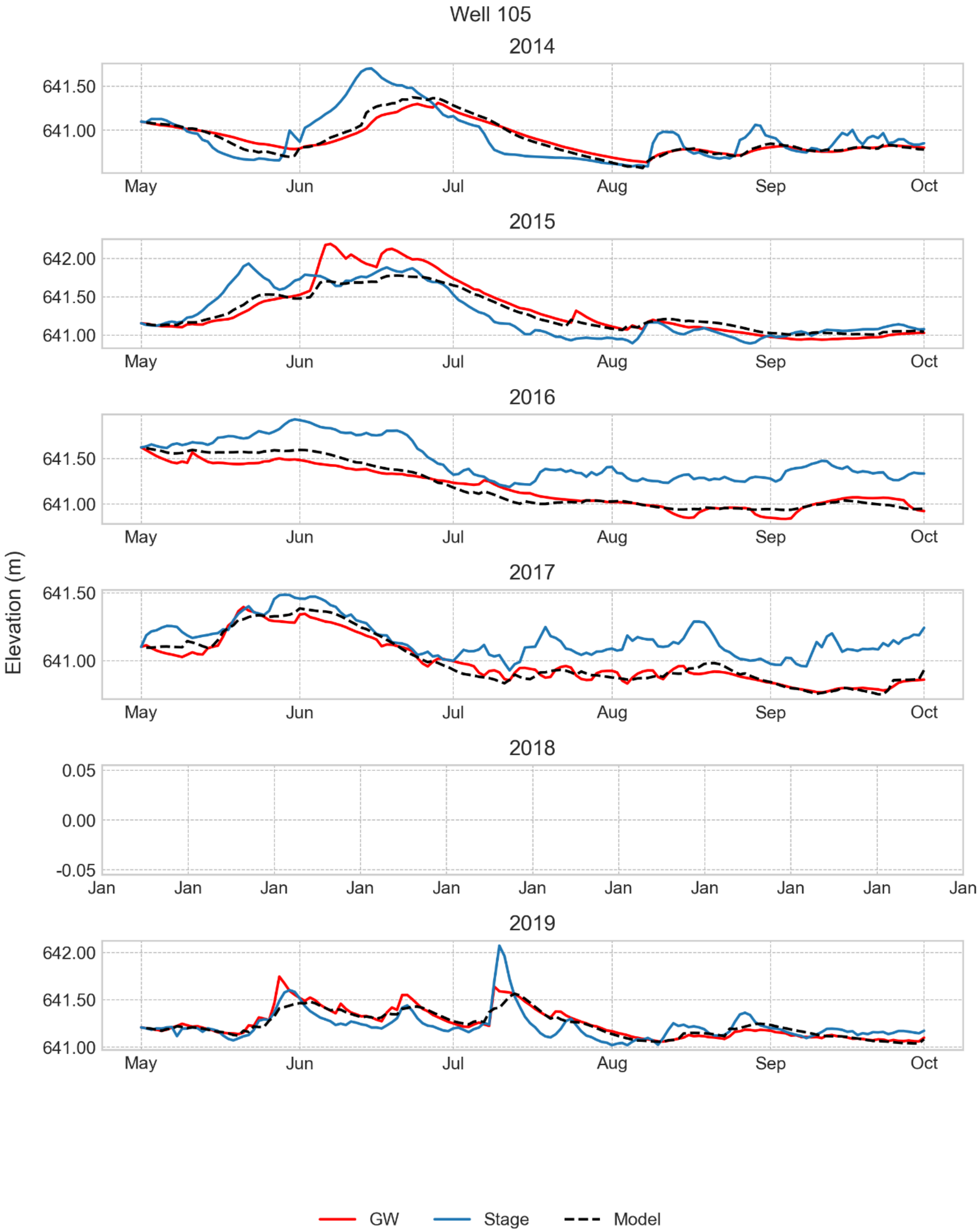
985

986

987


Well 104
2014
2015
2016
2017
2018
2019
Elevation (m)
642.00
641.50
641.00
642.50
641.60
641.40
641.20
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

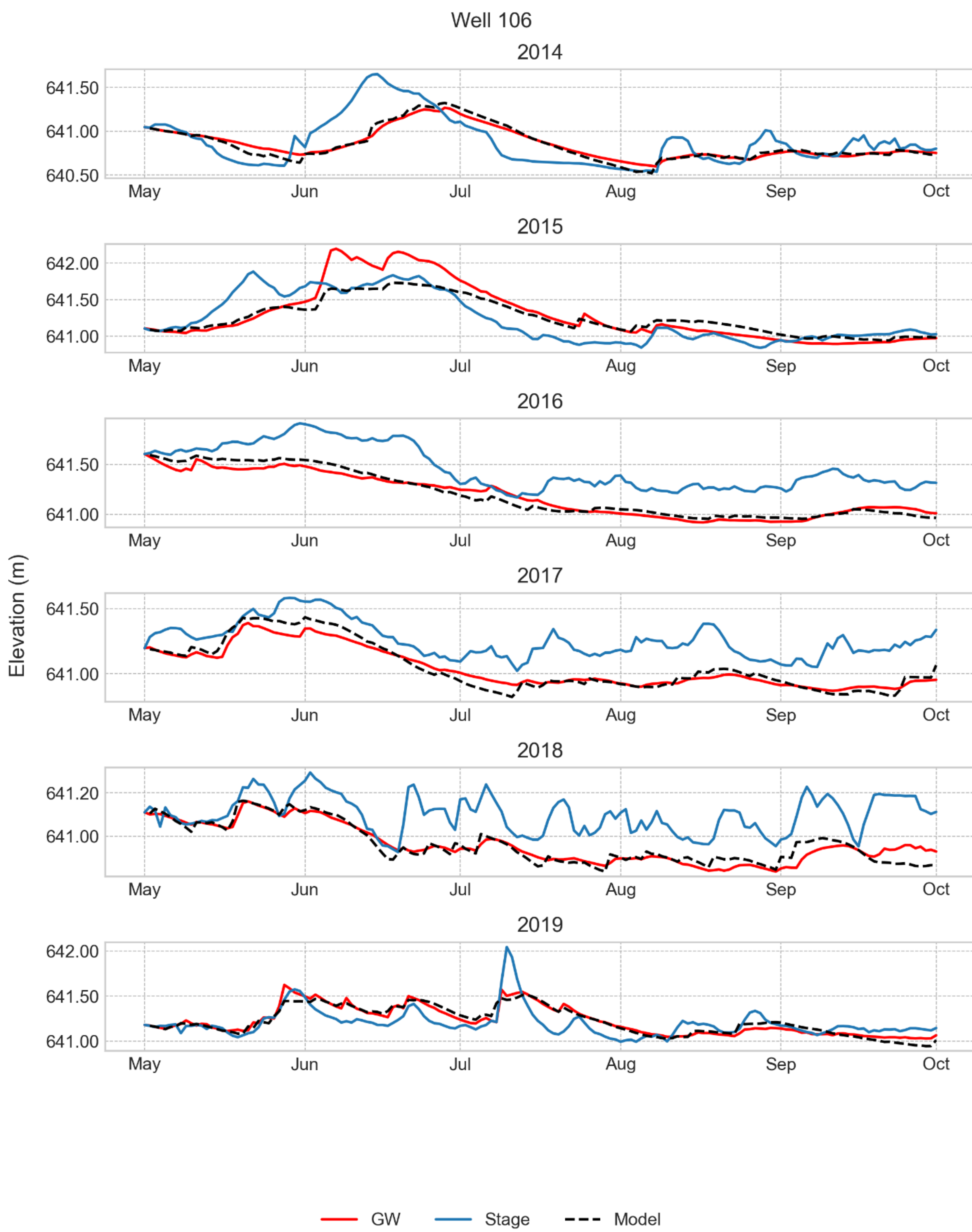
988


Well 105
2014
641.50
641.00
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.50
641.00
2016
641.50
641.00
2017
641.50
641.00
2018
0.05
0.00
-0.05
Jan
2019
642.00
641.50
641.00
Elevation (m)
GW
Stage
Model

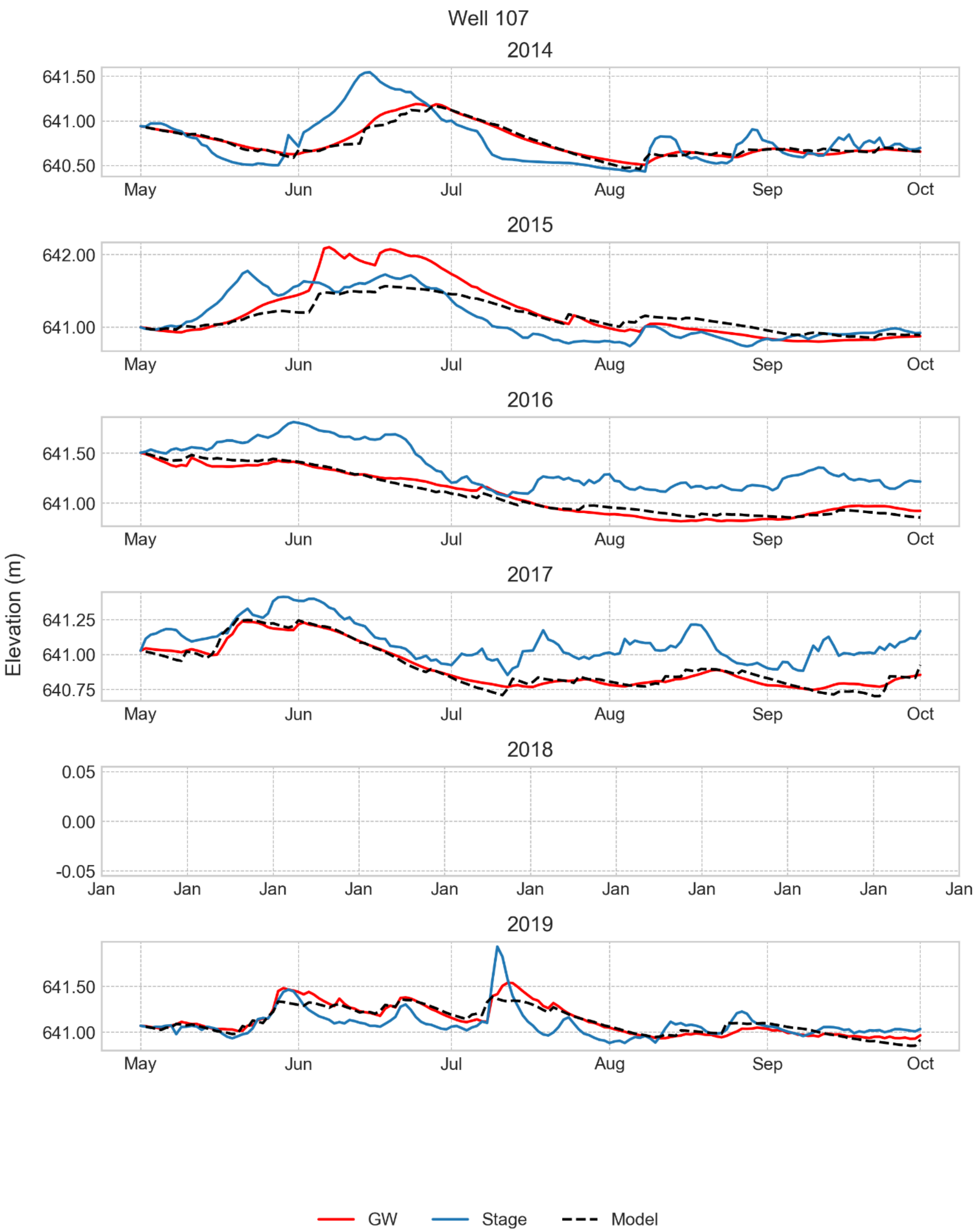
989

990

991


Well 106
2014
2015
2016
2017
2018
2019
Elevation (m)
641.50
641.00
640.50
642.00
641.20
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

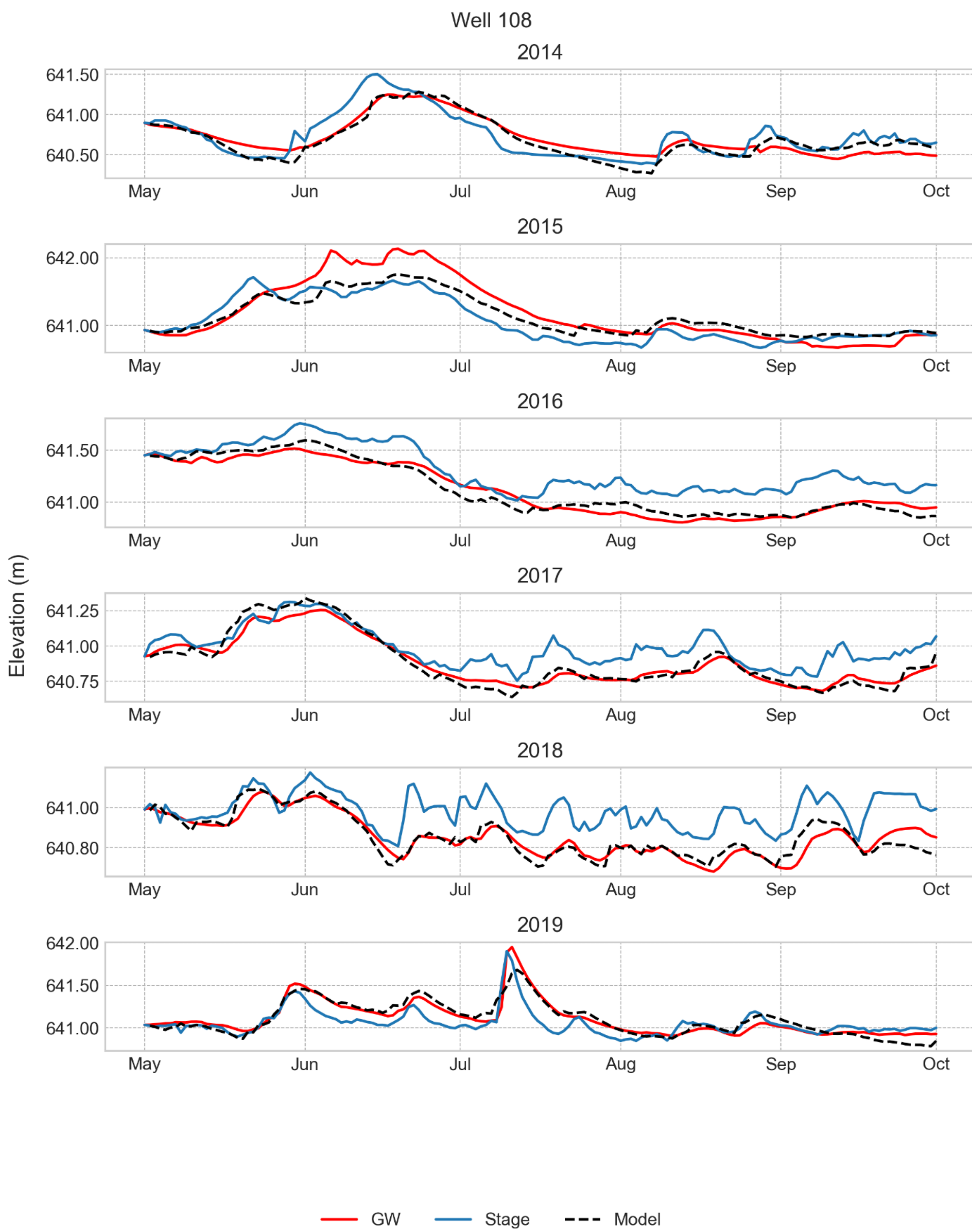
992


Well 107
2014
641.50
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.25
641.00
640.75
2018
0.05
0.00
-0.05
Jan
2019
641.50
641.00
Elevation (m)
GW
Stage
Model

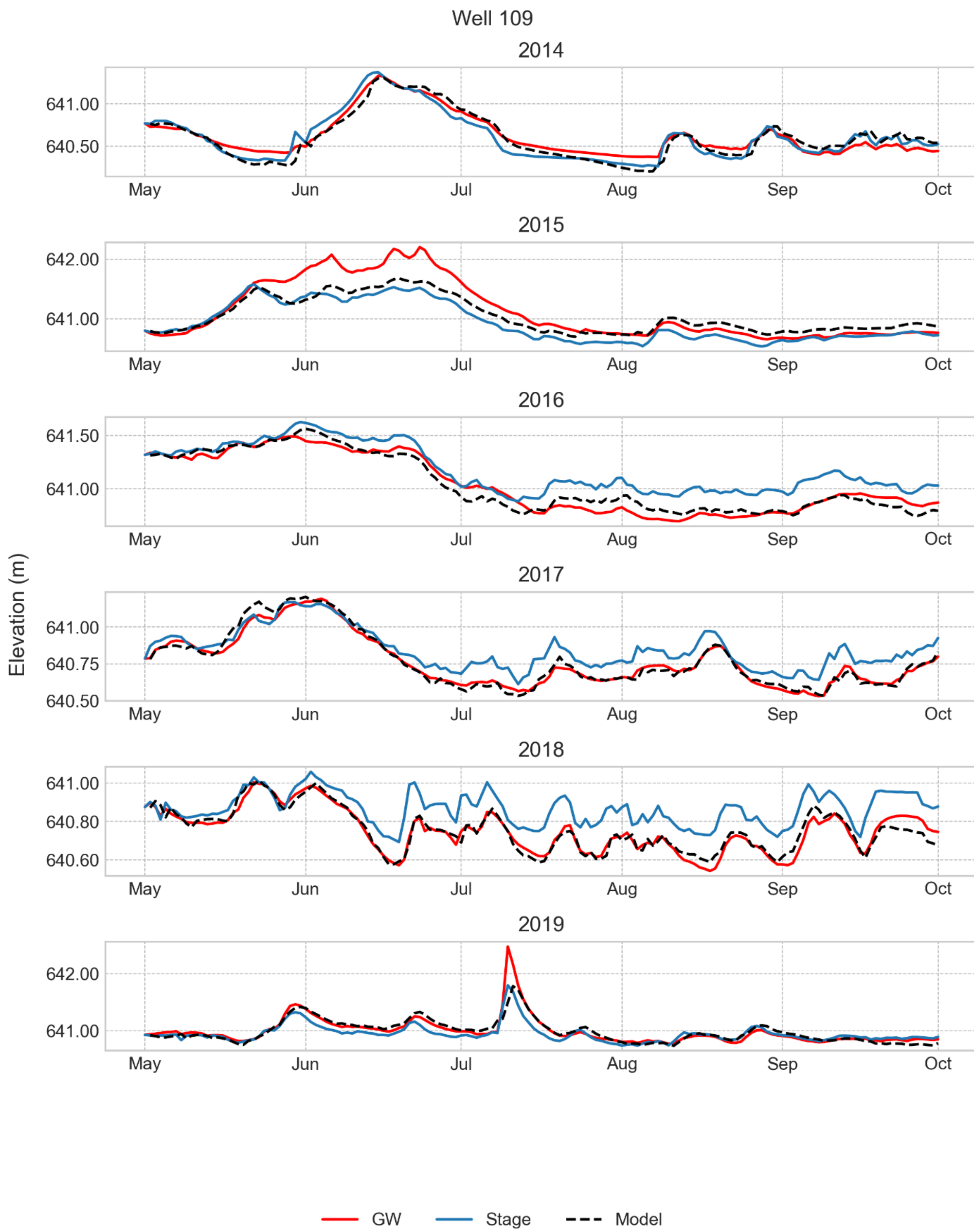
993

994

995


Well 108
2014
641.50
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.25
641.00
640.75
2018
641.00
640.80
2019
642.00
641.50
641.00
Elevation (m)
GW
Stage
Model

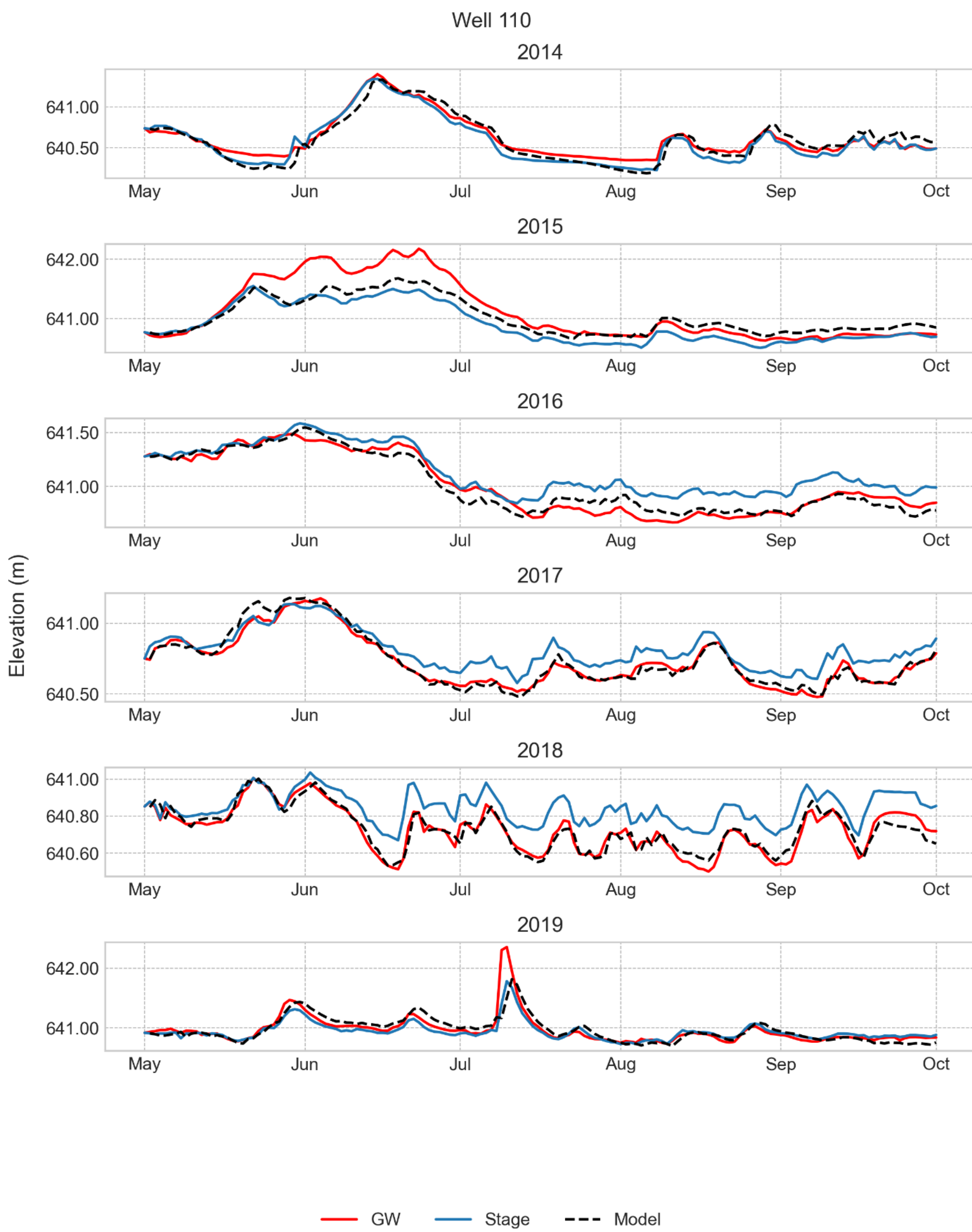
996


Well 109
2014
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.00
640.75
640.50
2018
641.00
640.80
640.60
2019
642.00
641.00
Elevation (m)
GW
Stage
Model

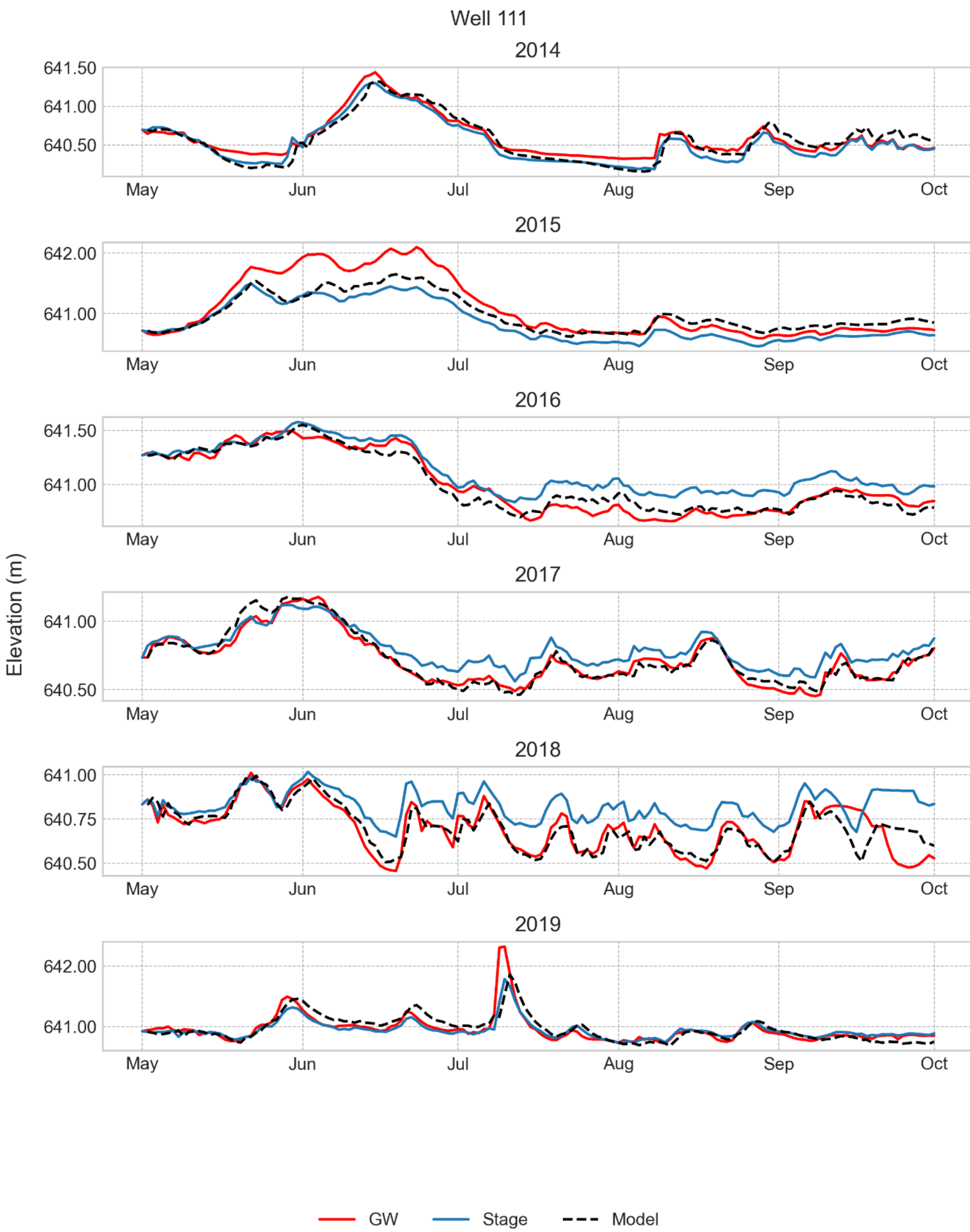
997

998

999


Well 110
2014
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.00
640.50
2018
641.00
640.80
640.60
2019
642.00
641.00
Elevation (m)
GW
Stage
Model

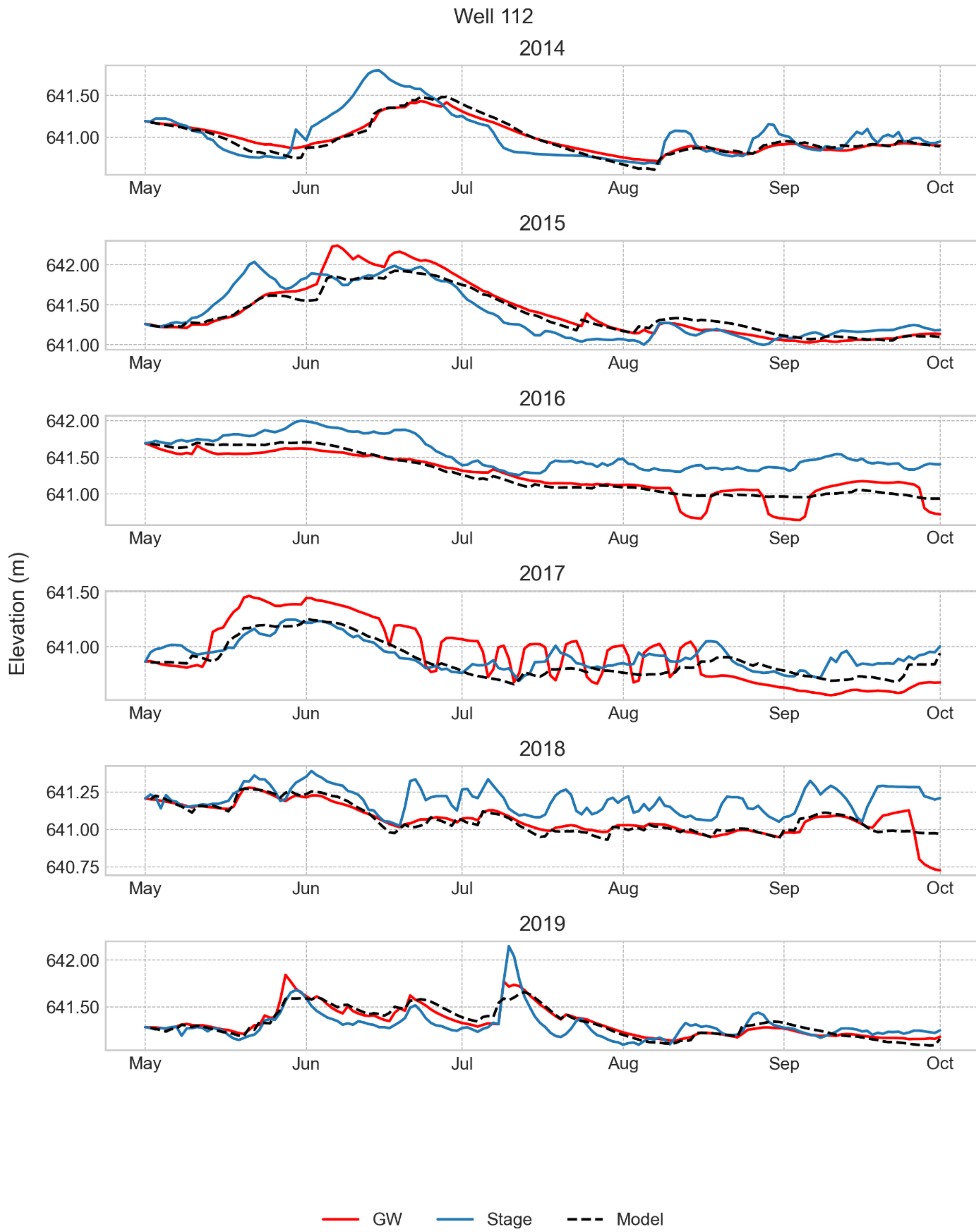
1000


Well 111
2014
641.50
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
Elevation (m)
2017
641.00
640.50
2018
641.00
640.75
640.50
2019
642.00
641.00
GW
Stage
Model

1001

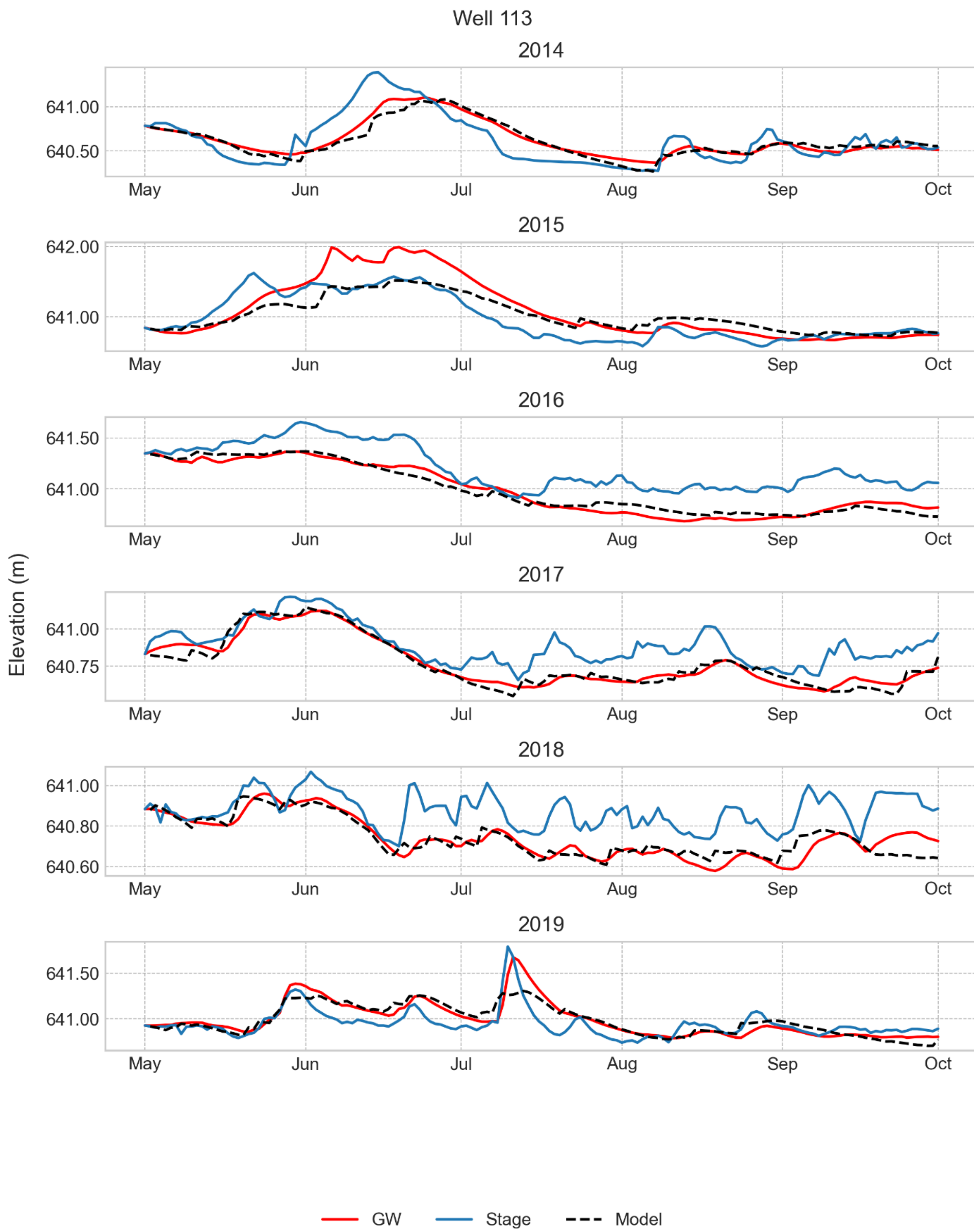
1002

1003



Well 112
2014
2015
2016
2017
2018
2019
Elevation (m)
641.50
641.00
642.00
641.25
640.75
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

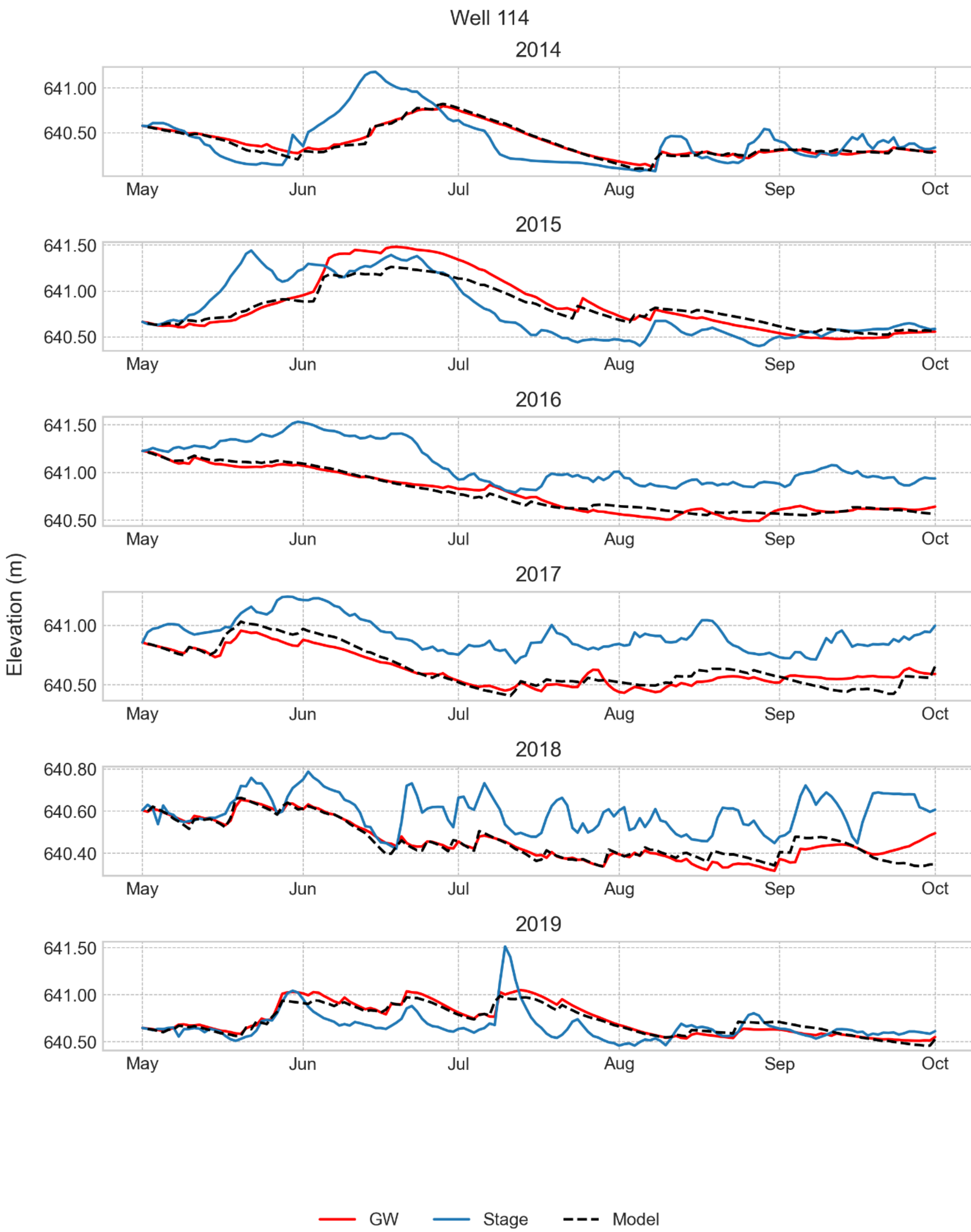
1004


Well 113
2014
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.00
640.75
2018
641.00
640.80
640.60
2019
641.50
641.00
Elevation (m)
GW
Stage
Model

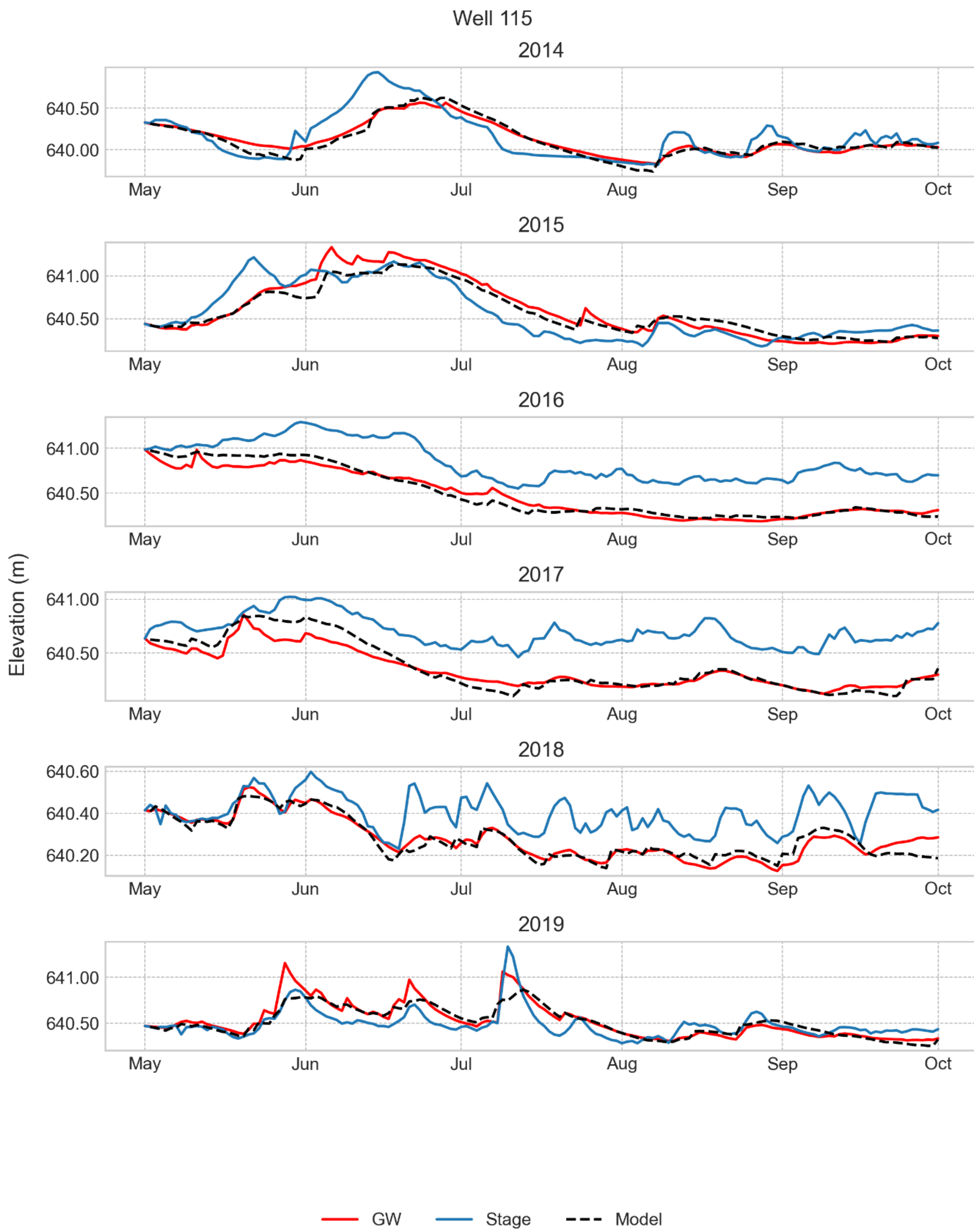
1005

1006

1007


Well 114
2014
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
641.50
641.00
640.50
2016
641.50
641.00
640.50
2017
641.00
640.50
2018
640.80
640.60
640.40
2019
641.50
641.00
640.50
Elevation (m)
GW
Stage
Model

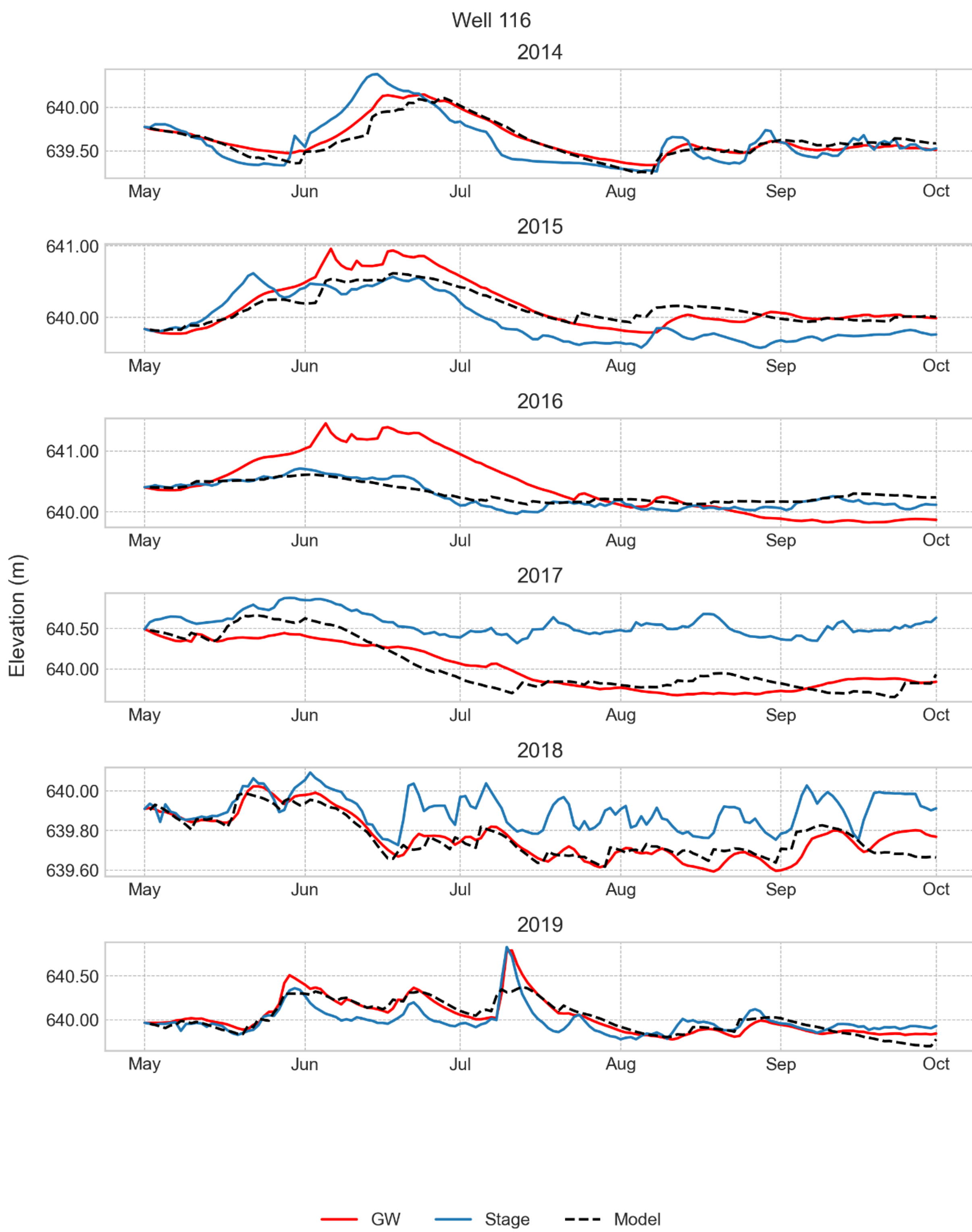
1008


Well 115
2014
2015
2016
2017
2018
2019
Elevation (m)
640.00
640.50
641.00
640.20
640.40
640.60
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

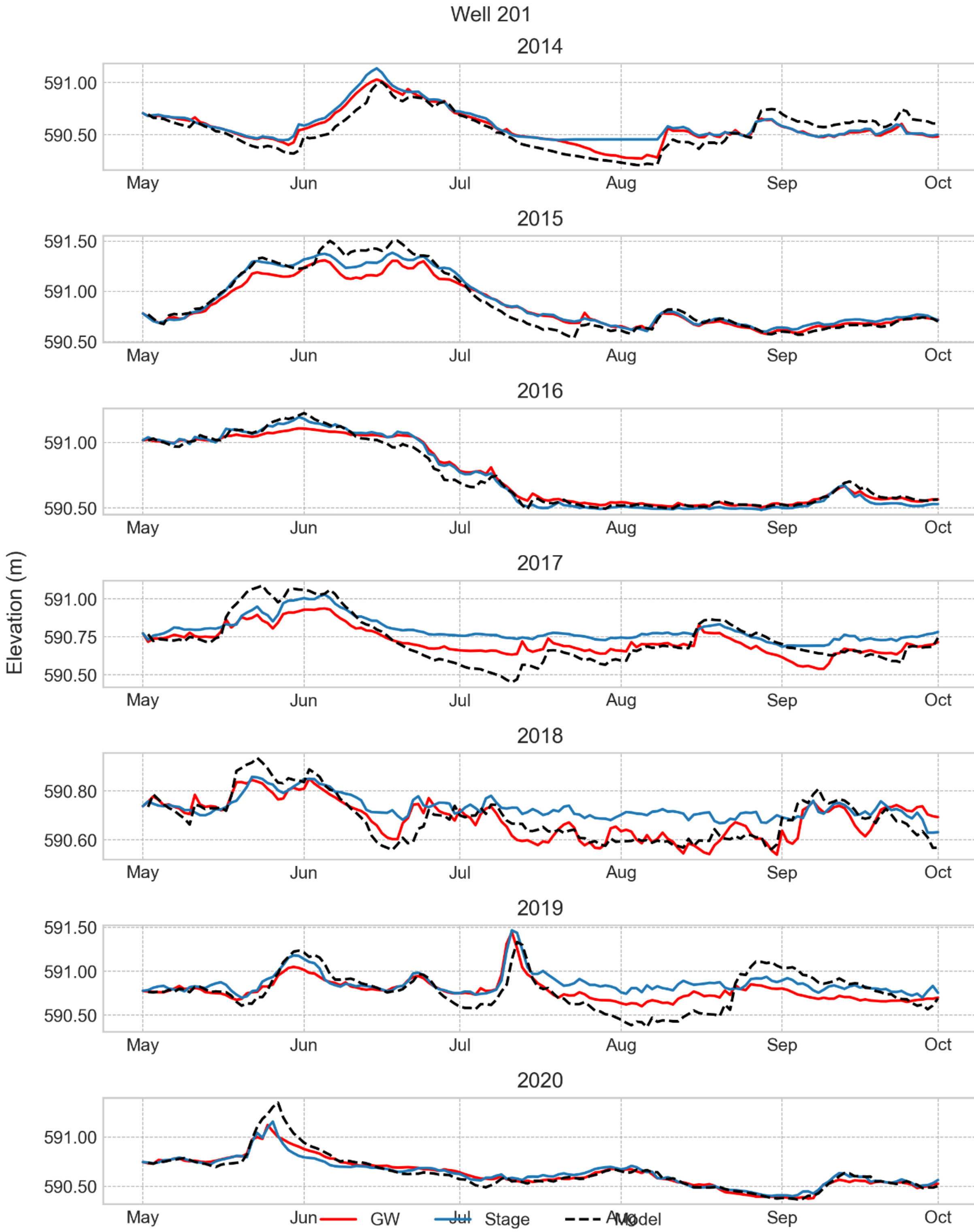
1009

1010

1011


Well 116
2014
640.00
639.50
May
Jun
Jul
Aug
Sep
Oct
2015
641.00
640.00
2016
641.00
640.00
2017
640.50
640.00
2018
640.00
639.80
639.60
2019
640.50
640.00
Elevation (m)
GW
Stage
Model

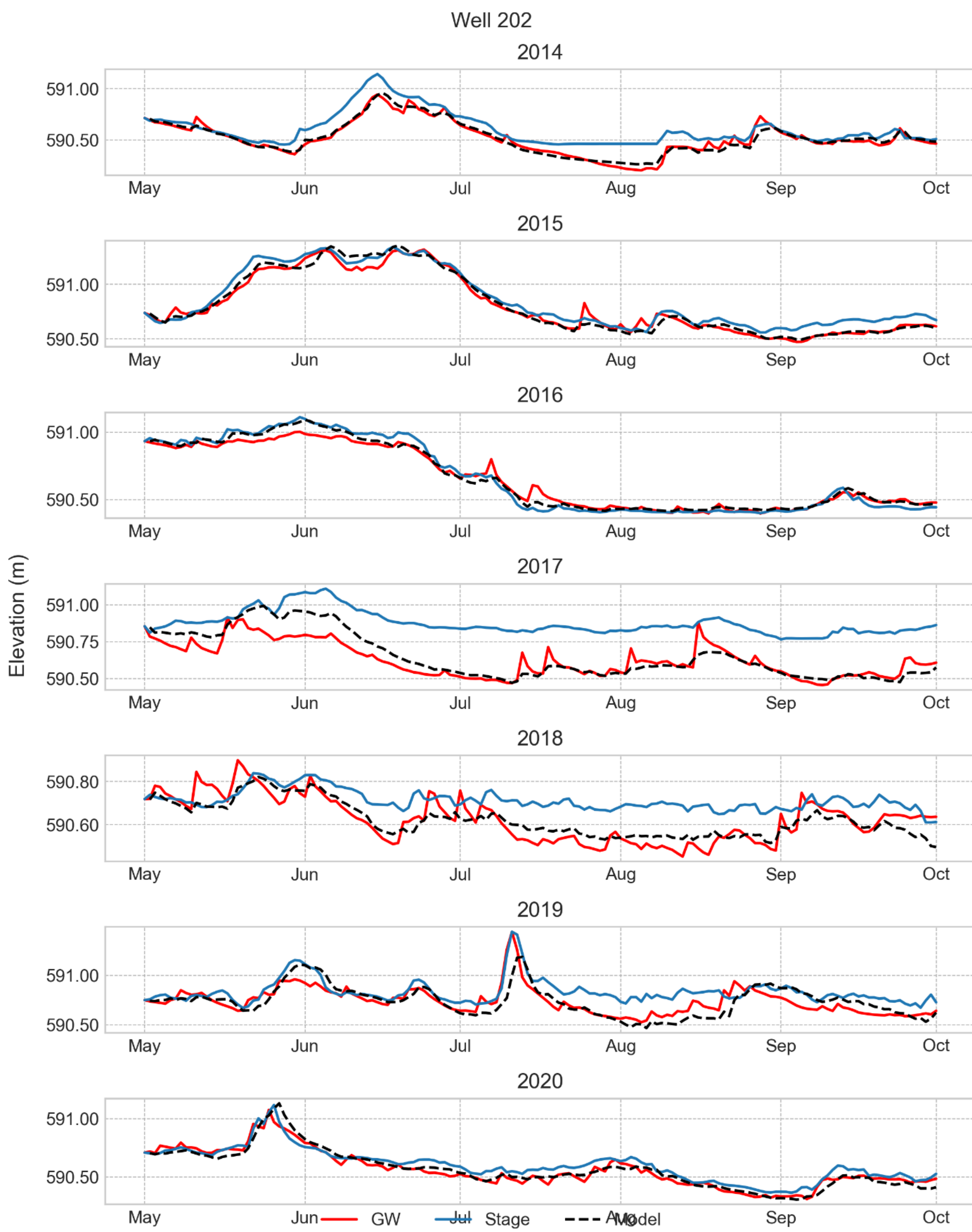
1012


Well 201
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
591.50
591.00
590.80
590.75
590.60
590.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

1013

1014

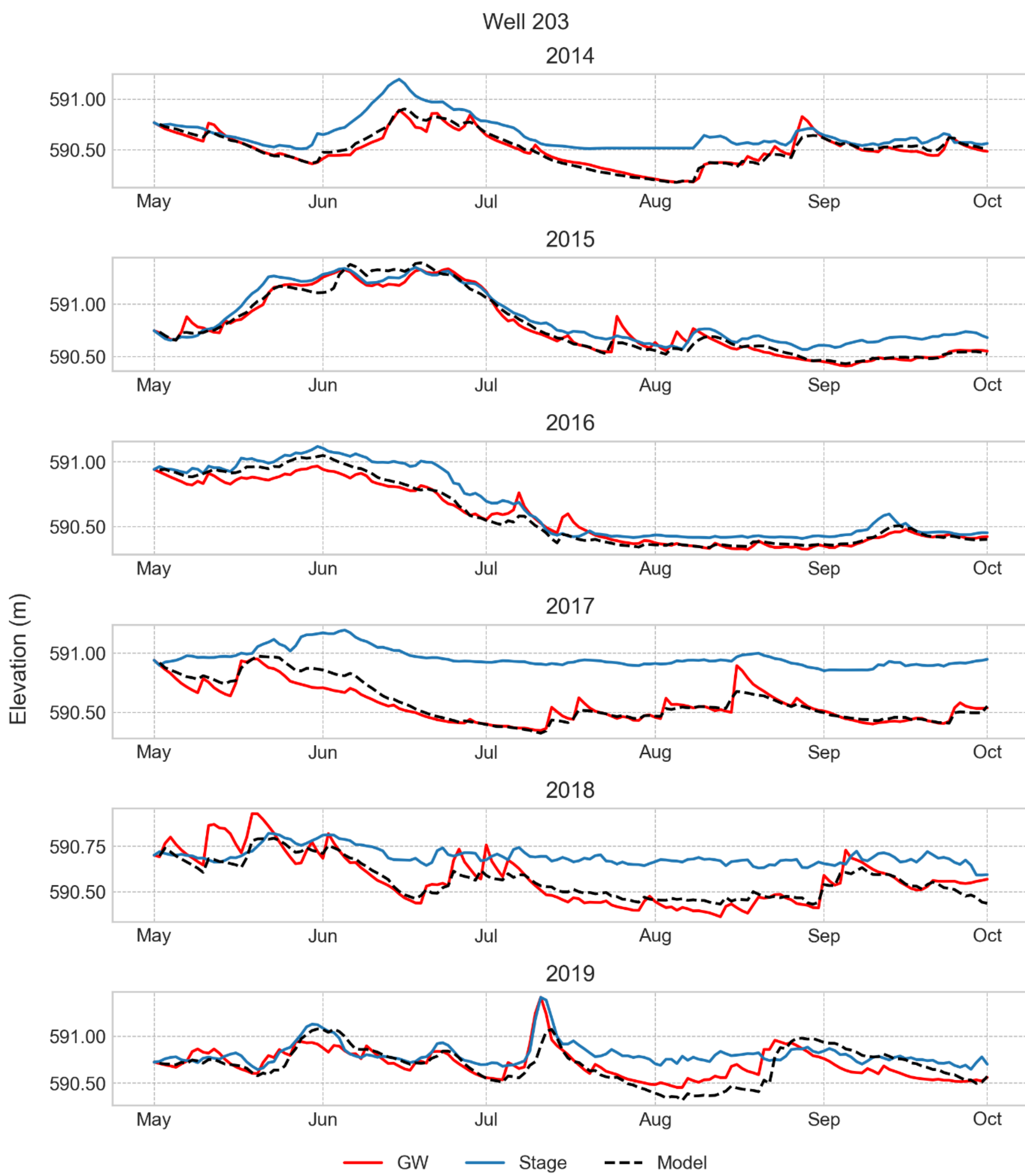
1015



Well 202
2014
2015
2016
2017
2018
2019
2020
591.00
590.50
590.75
590.80
590.60
Elevation (m)
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

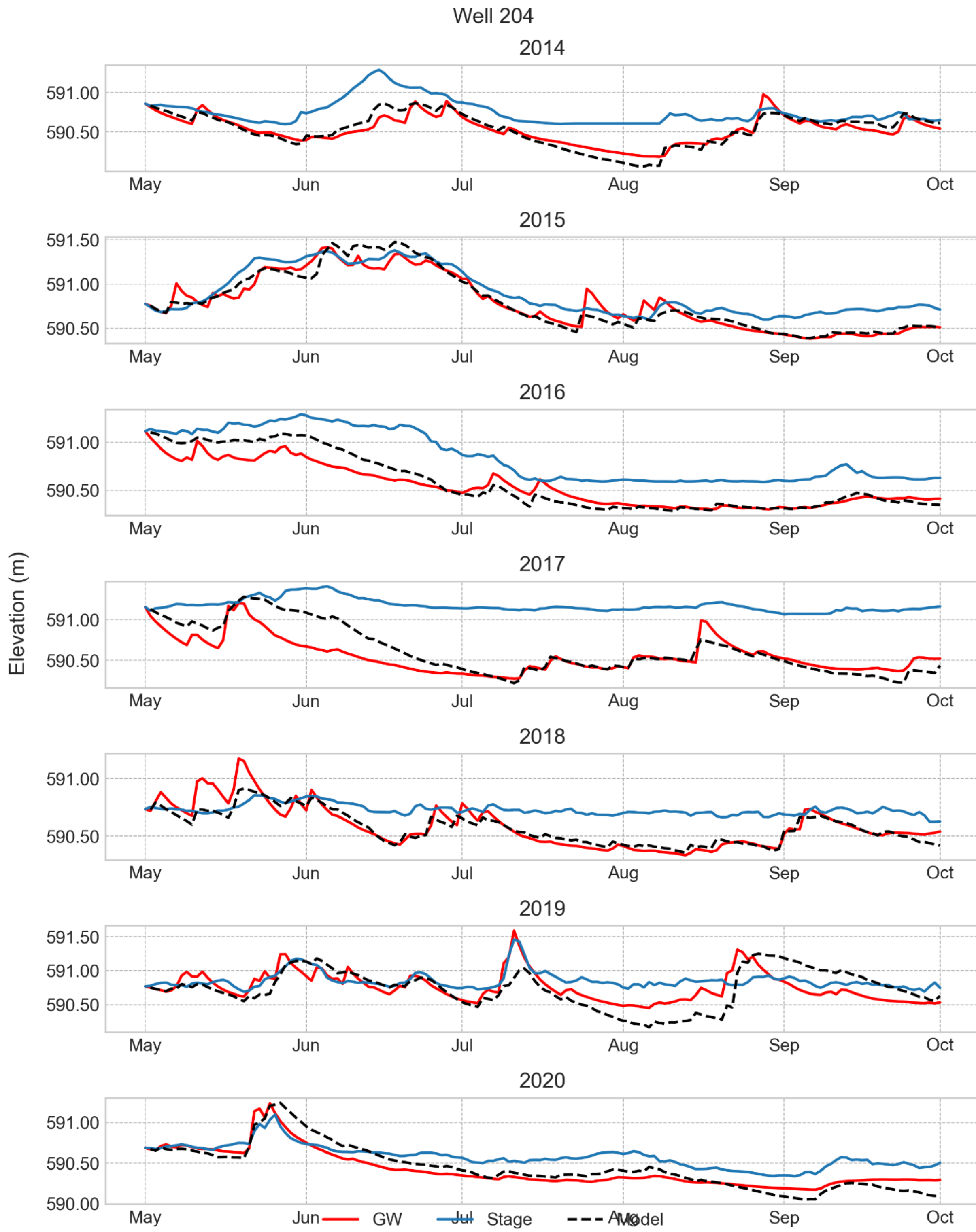
1016

1017



1018

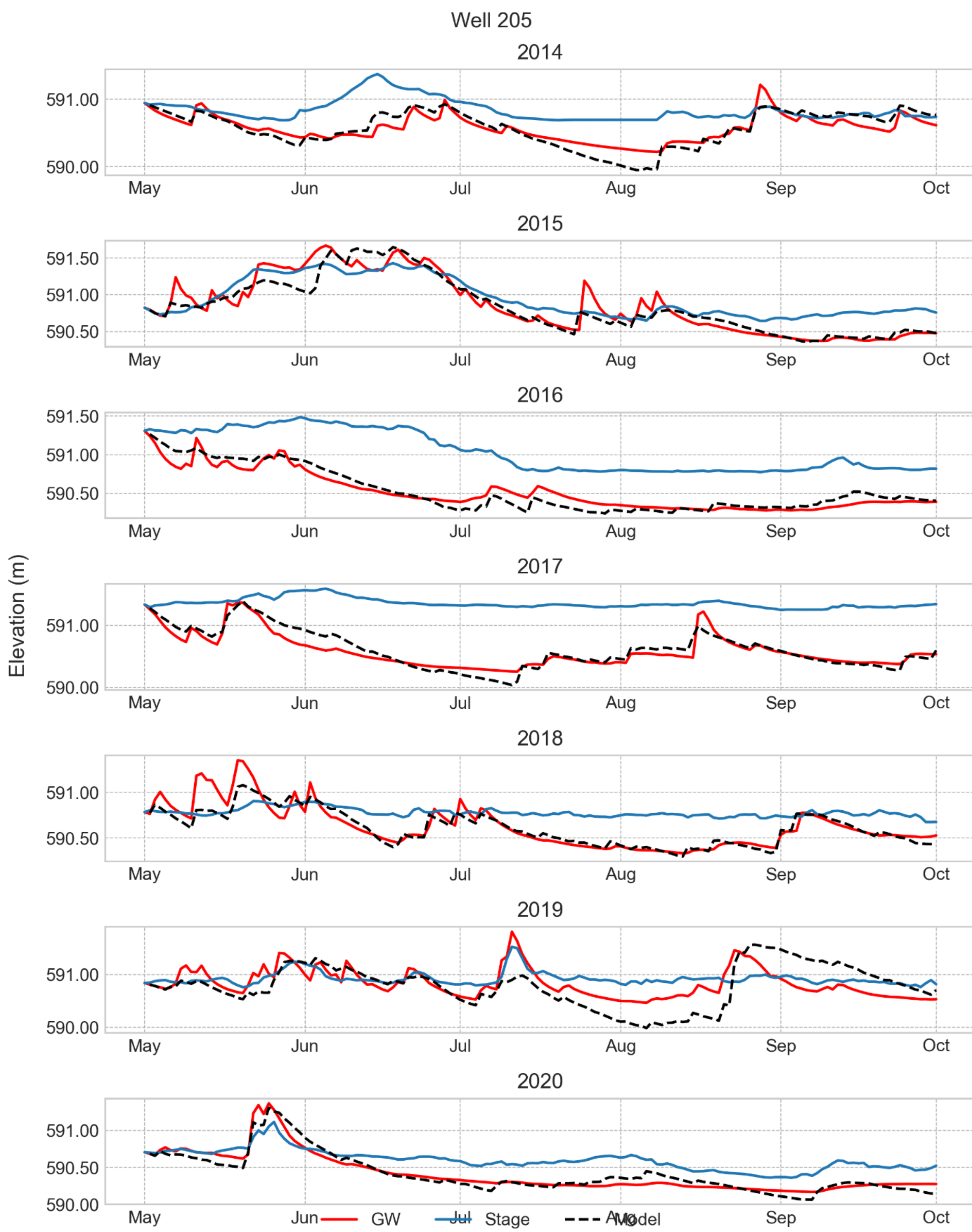
1019


Well 204
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
591.50
591.00
590.50
590.00
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

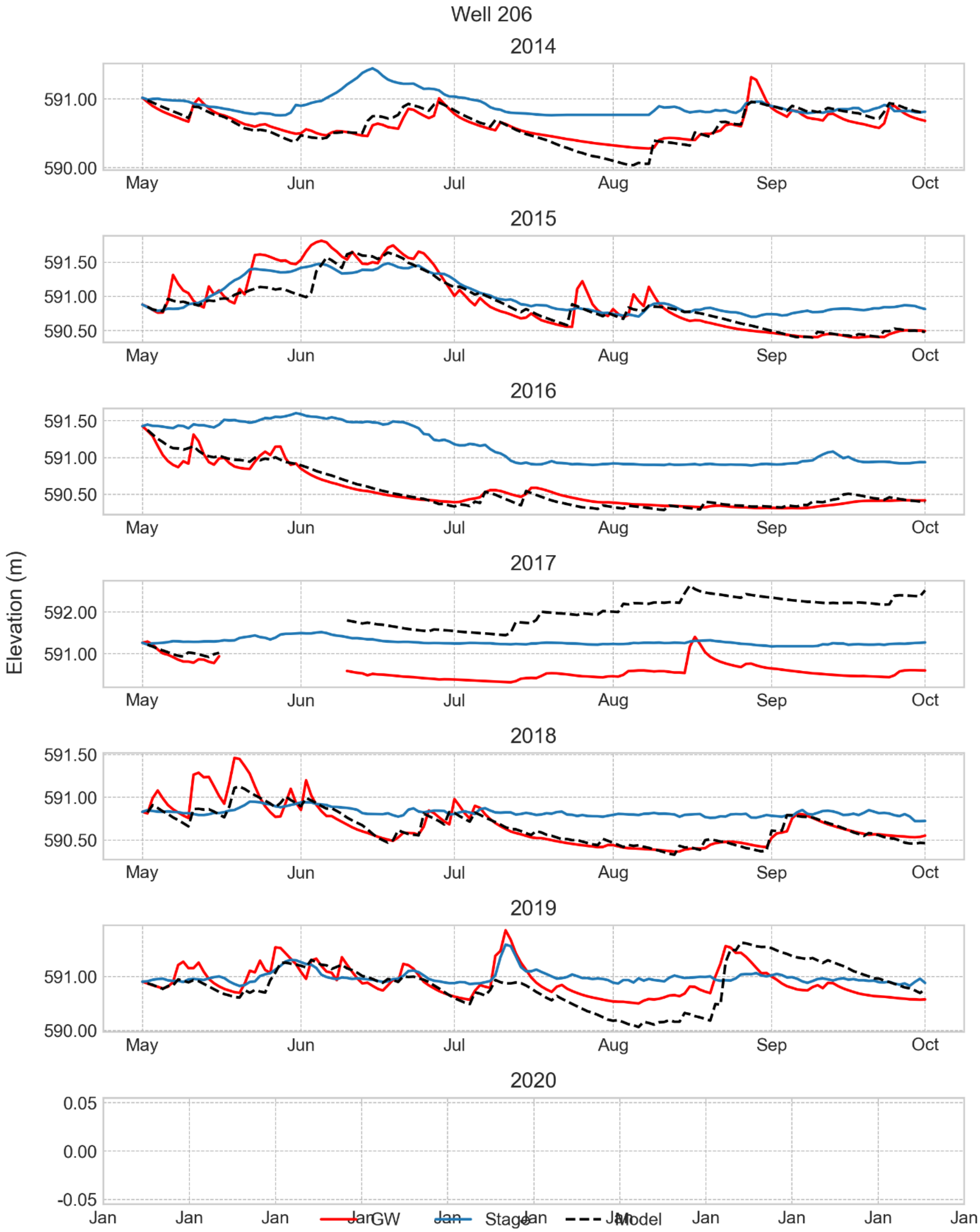
1020

1021

1022


Well 205
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
591.00
590.00
591.50
590.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

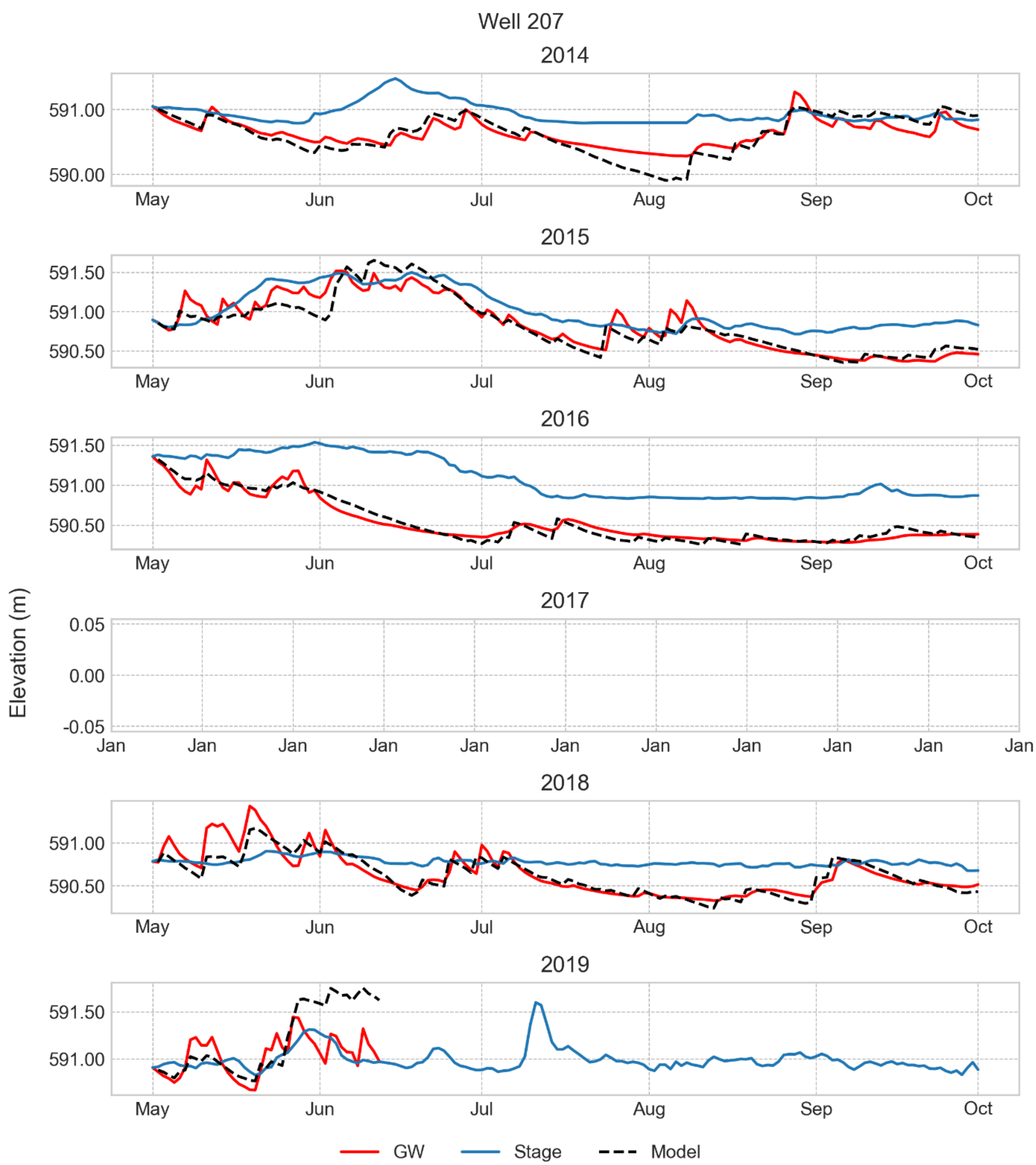
1023


Well 206
2014
591.00
590.00
May
Jun
Jul
Aug
Sep
Oct
2015
591.50
591.00
590.50
2016
591.50
591.00
590.50
2017
592.00
591.00
2018
591.50
591.00
590.50
2019
591.00
590.00
2020
0.05
0.00
-0.05
Jan
Elevation (m)
GW
Stage
Model

1024

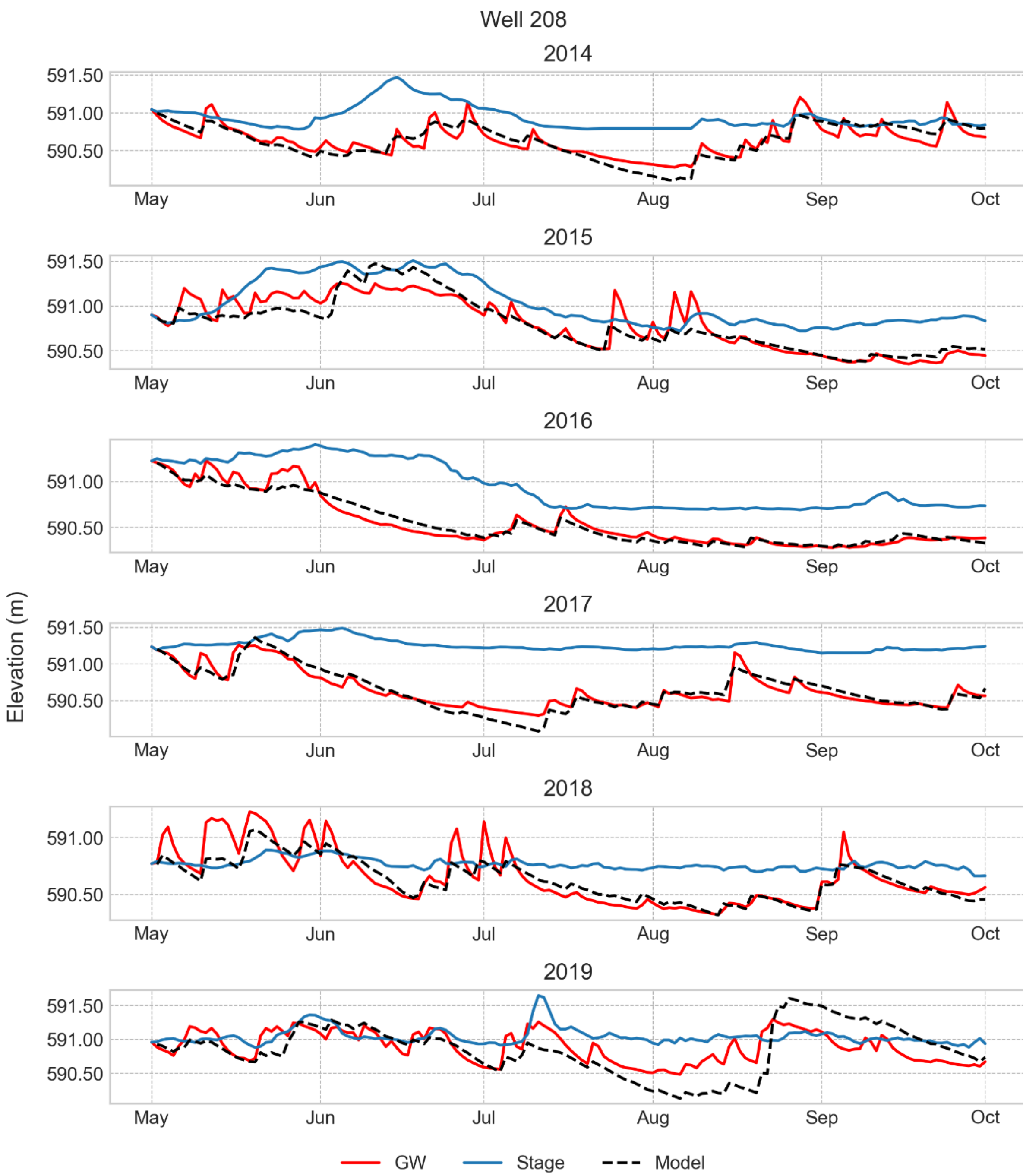
1025

1026



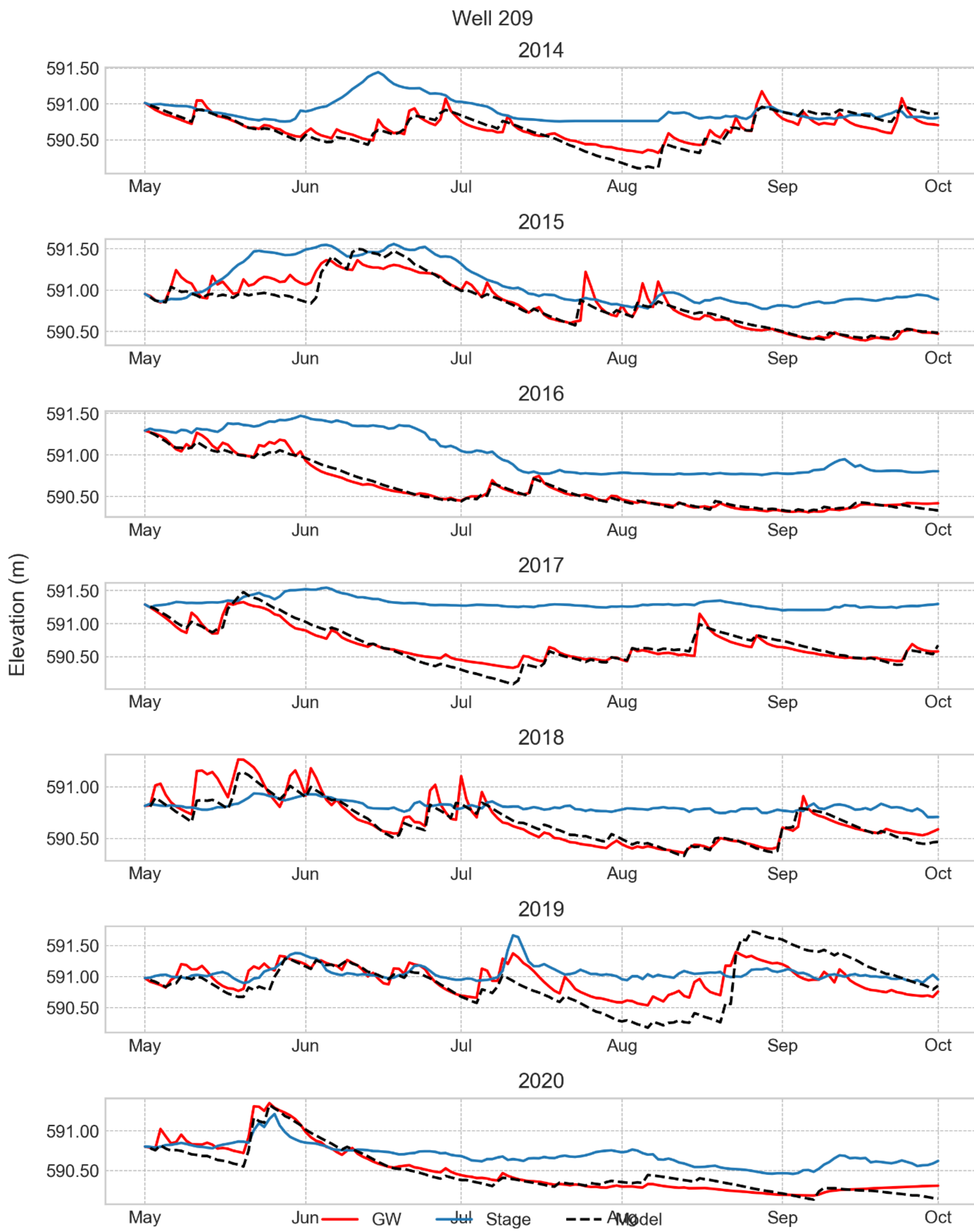
1027

1028


Well 208
2014
2015
2016
2017
2018
2019
Elevation (m)
591.50
591.00
590.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

1029

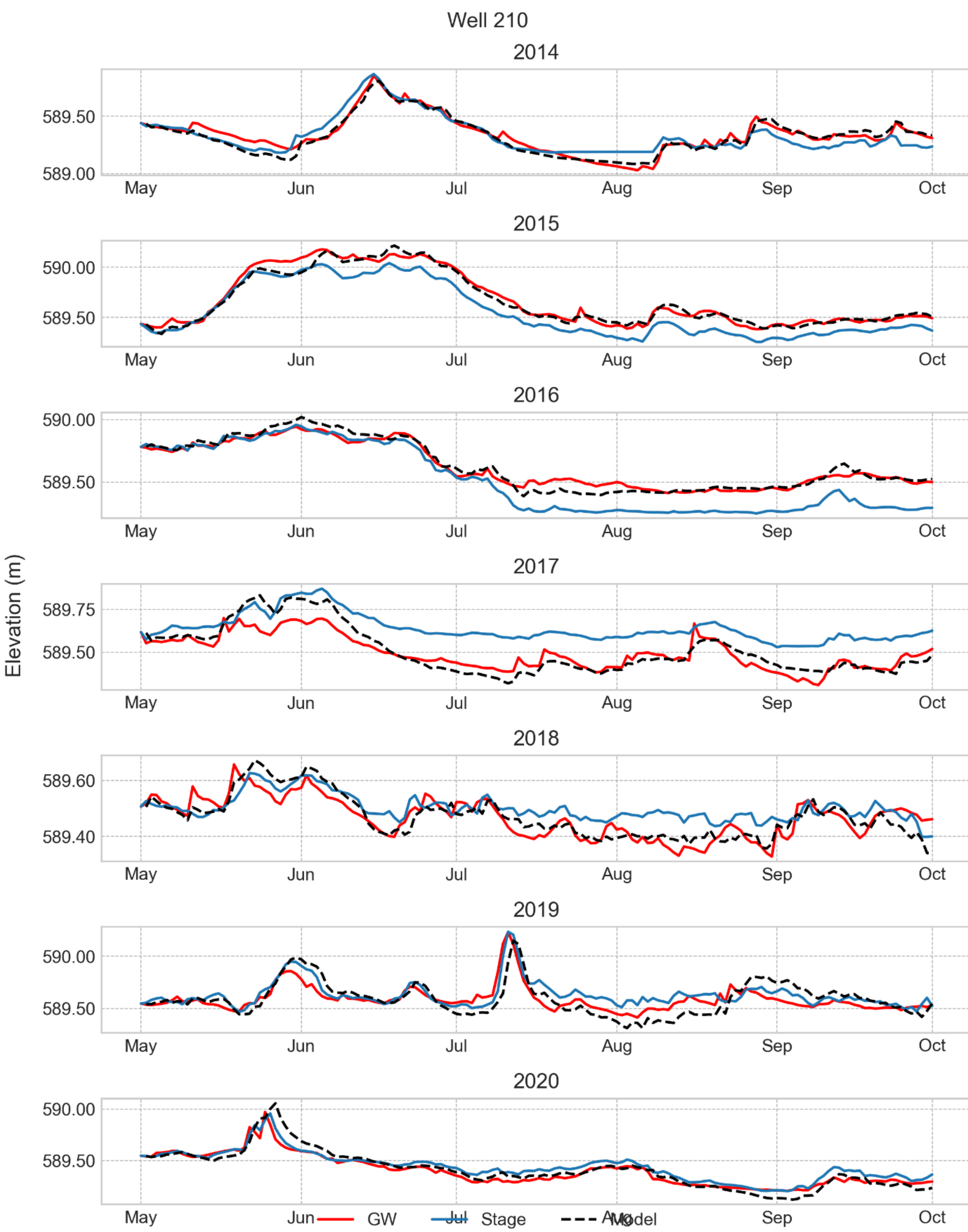
1030


Well 209
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
591.50
591.00
590.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

1031

1032

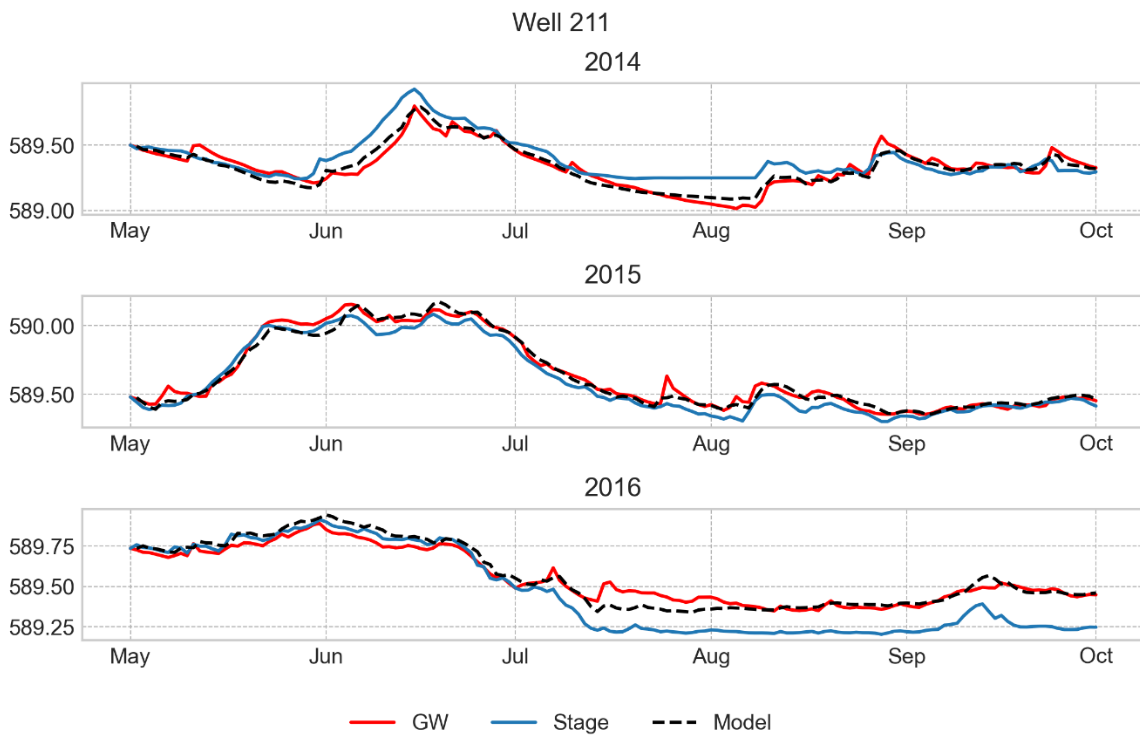
1033


Well 210
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
589.00
589.50
590.00
589.75
589.60
589.40
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

1034

1035

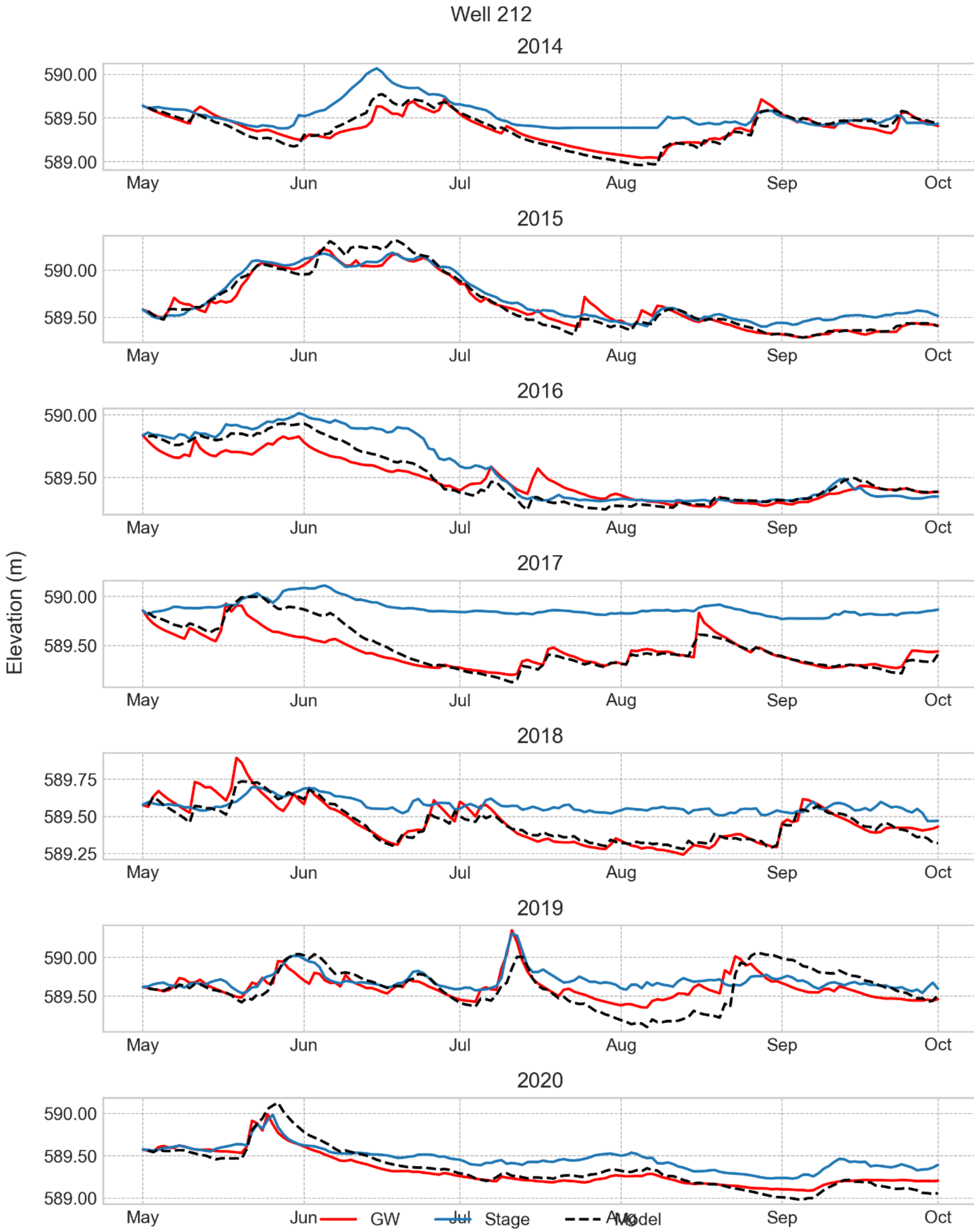
1036



1037

1038

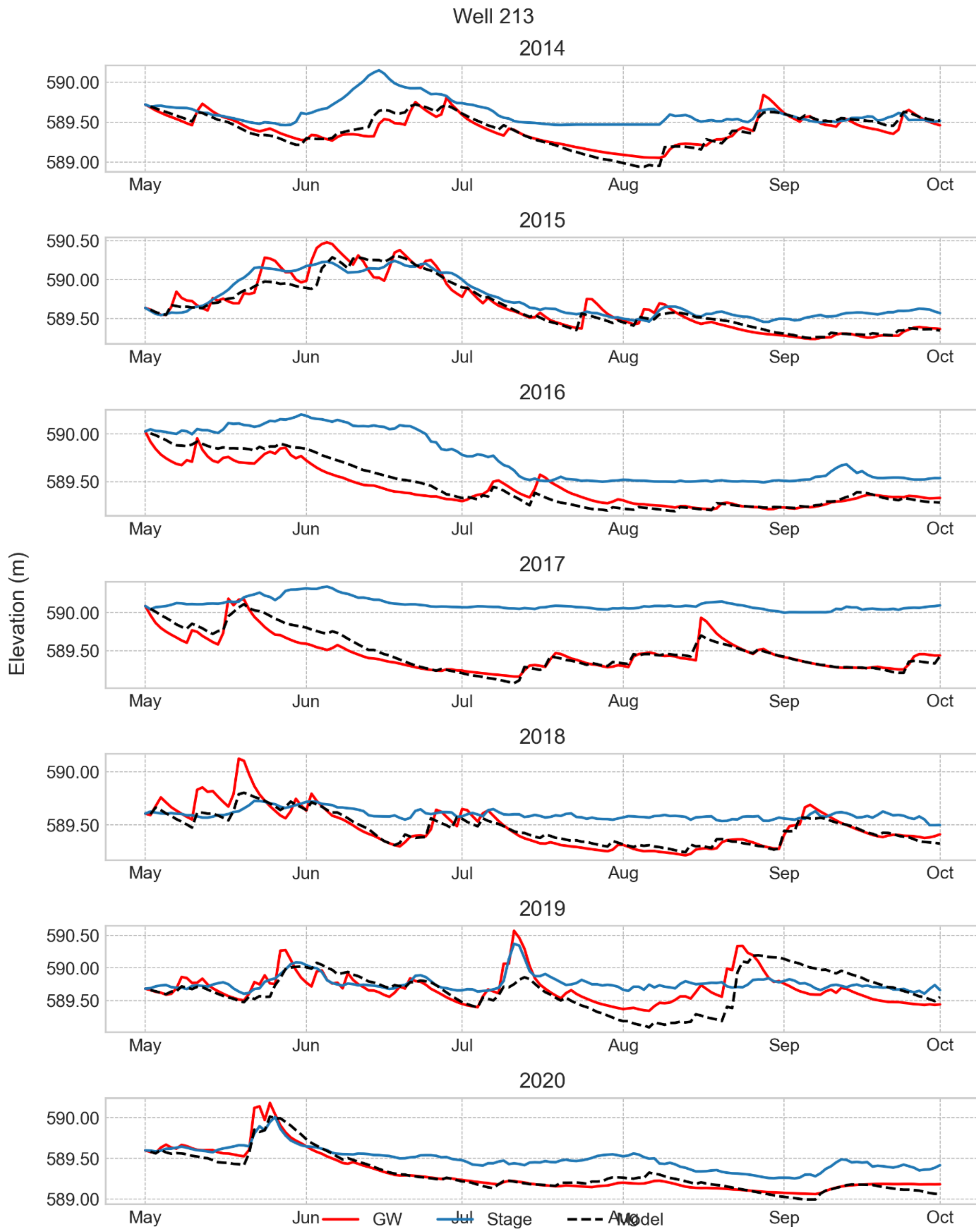
1039


Well 212
2014
590.00
589.50
589.00
May
Jun
Jul
Aug
Sep
Oct
2015
590.00
589.50
2016
590.00
589.50
2017
590.00
589.50
Elevation (m)
2018
589.75
589.50
589.25
2019
590.00
589.50
2020
590.00
589.50
589.00
GW
Stage
Model

1040

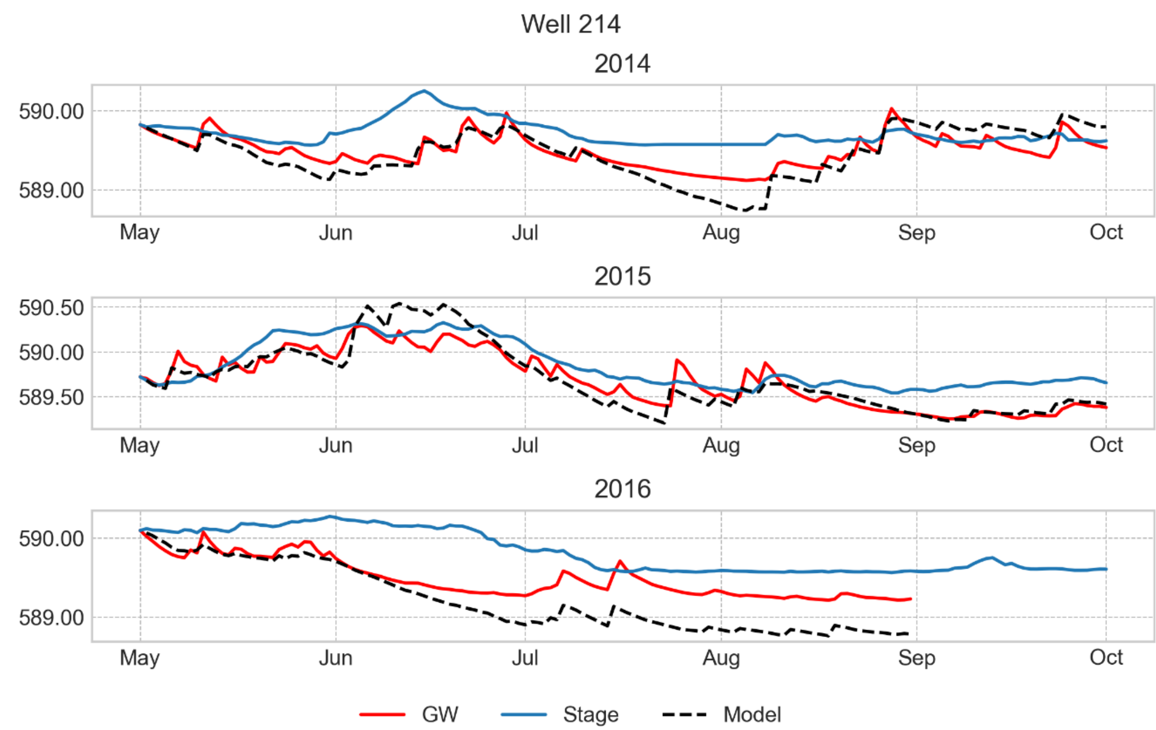
1041

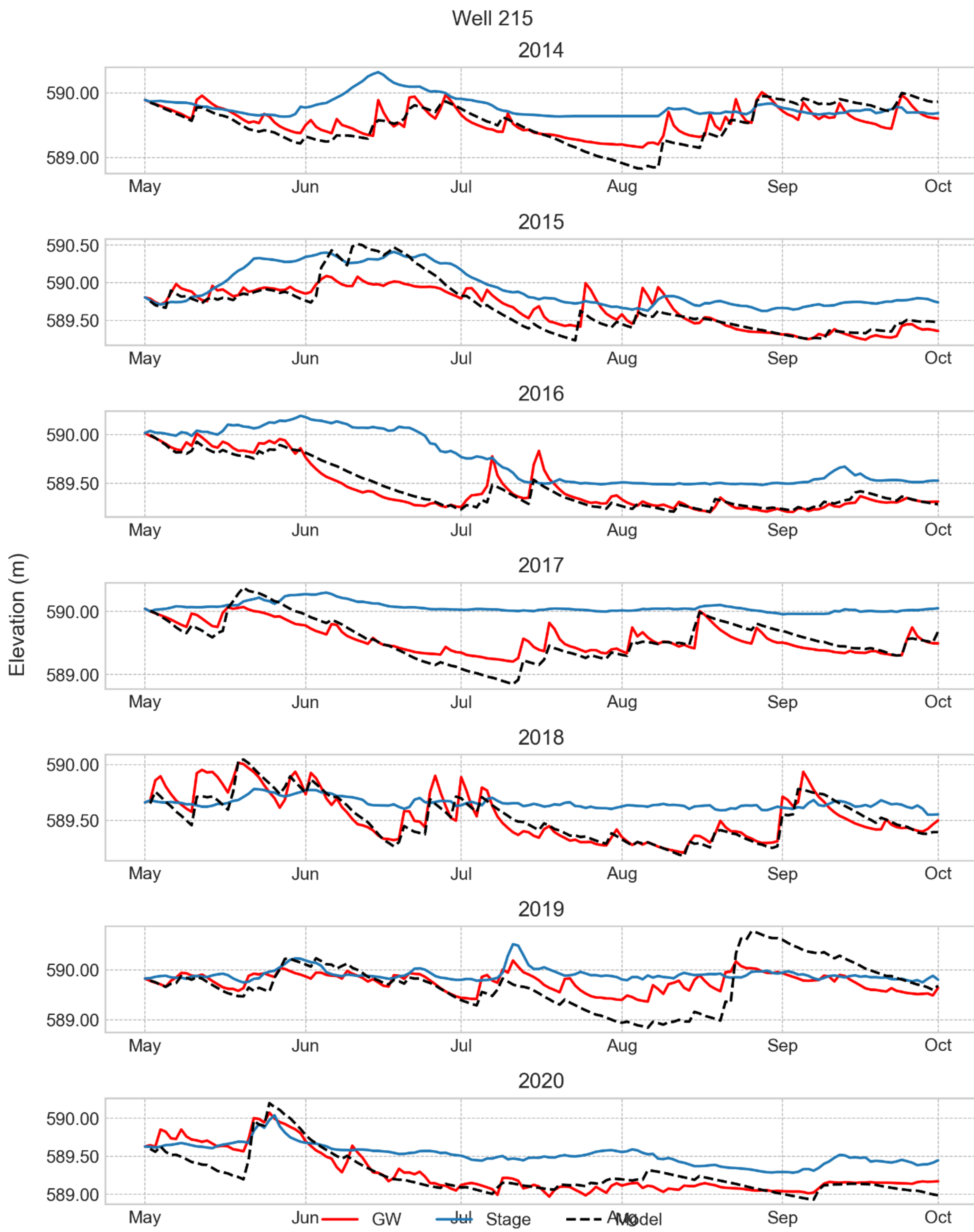
1042


Well 213
2014
590.00
589.50
589.00
May
Jun
Jul
Aug
Sep
Oct
2015
590.50
590.00
589.50
2016
590.00
589.50
2017
590.00
589.50
2018
590.00
589.50
2019
590.50
590.00
589.50
2020
590.00
589.50
589.00
Elevation (m)
GW
Stage
Model

1043

GW Stage Model

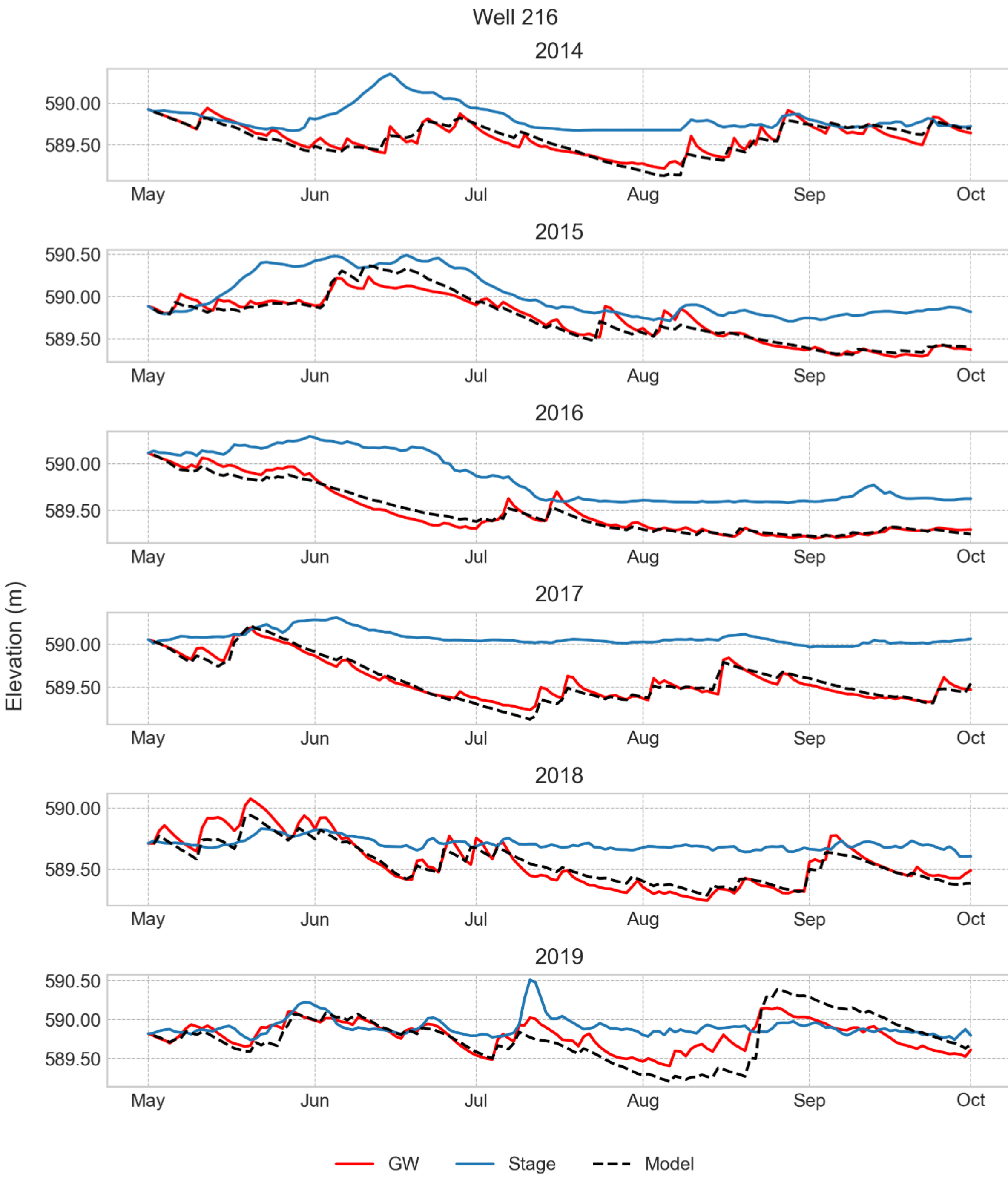



Well 215
2014
590.00
589.00
May
Jun
Jul
Aug
Sep
Oct
2015
590.50
590.00
589.50
2016
590.00
589.50
2017
590.00
589.00
2018
590.00
589.50
2019
590.00
589.00
2020
590.00
589.50
589.00
Elevation (m)
GW
Stage
Model

1049

1050

1051



1052

1053