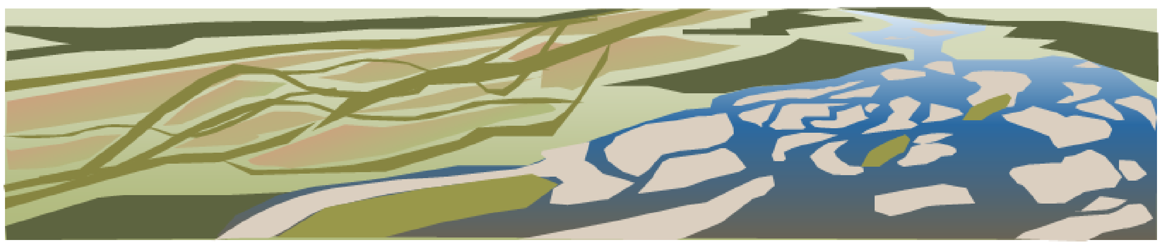
PLATTE RIVER RECOVERY IMPLEMENTATION PROGRAM
(PRRIP -or- PROGRAM)
Wet Meadow Hydrology Study (2013-2021)
DRAFT REPORT

August 2023



Prepared by Kristen Cognac

PRRP Executive Director's Office (EDO)

Preface

This document was prepared by the Executive Director's Office (EDO) of the Platte River Recovery Implementation Program ("Program" or "PRRIP"). The information and analyses presented herein are focused on informing management of wet meadows by the Program as well as other land managers along the central Platte River Valley (CPRV). The Program has invested fourteen years in implementation of an adaptive management program to reduce uncertainties about proposed management strategies and learn about on- and off-channel habitat and species responses to management actions. During that time, the Program has implemented management actions, collected a large body of physical and species response data, and developed modeling and analysis tools to aid in the interpretation and synthesis of data.

Wet meadows were a focus area for investigation during the Program's First Increment. The Program's Land Plan directs the Program to use its best efforts to acquire and manage 640 acres of wet meadow at each habitat complex. In addition, multiple priority hypotheses were developed to gather information on the importance of wet meadows for whooping cranes, the relationship between wet meadow hydrology and availability of forage resources for whooping cranes, and the relationship between river flow and wet meadow hydrology. First Increment whooping crane/wet meadow investigations culminated in a diurnal (day) use analysis conducted by WEST Inc. (Howlin and Nasman, 2017) that deemphasized the importance of wet meadows for whooping cranes. Since completion of the analysis, the Program has not focused on actively managing wet meadows as diurnal habitat for whooping cranes except for providing an area of short vegetative structure at each wet meadow site.

During the development of the First Increment Extension Science Plan, learning about the factors important for supporting wet meadow hydrology was included as Maintenance Learning for the Extension as wet meadows were still considered an important landcover type along the CPRV under Program land management. The Executive Director's Office began the analysis of wet meadow hydrology data in 2021, gathering together groundwater, surface water, climatological, river and floodplain elevation, and vegetation monitoring data from 2013 through 2021 to address the following Extension Science Plan (Big Question #10) learning objectives:

- Understand relationships between hydrological and meteorological variables and groundwater levels at natural wet meadow sites.
- Understand what constitutes a functional hydrological regime for wet meadows along the CPRV which can be used as a reference and applied to manage other sites.
- Develop a modeling tool that can be used by land managers in the CPRV to inform management decisions.

We begin with an Executive Summary that provides a consolidated summary of why this study was undertaken, what was done, and the main takeaways for decision-makers. The main text of the study is then divided into the following sections: 1) introduction and background, 2) study area and data collection, 3) quantifying the hydroregime, 4) groundwater-vegetation links, 5) model prediction of groundwater levels, and 6) river-floodplain elevation analysis. Sections 3 through 6 are structured as stand-alone scientific reports containing methods, results, and discussion. The final sections include references and appendices with supporting information.

Contents

List of Figures

List of Tables

List of Appendices

Executive Summary

i) Why did the Program initiate this study?

The importance of wet meadows for whooping cranes during migratory stopovers along the CPRV was a focus area for learning during the Program's First Increment. The Program's Land Plan directed the Program to do its best to acquire and manage 640 acres of wet meadow at each habitat complex as potential off-channel habitat for whooping cranes. Multiple priority hypotheses were developed to gather information on whooping crane use of wet meadows, the relationship between wet meadow hydrology and the availability of forage resources for whooping cranes, and the relationship between river flow and wet meadow hydrology. Several studies were contemplated including a wet meadow literature review, a study of the relative importance of wet meadows to whooping cranes, and a study of the relationship between wet meadow hydrology and wet meadow quality for whooping cranes. During the First Increment, in response to a diurnal (day) use analysis performed by WEST Inc. (Howlin and Nasman, 2017) that concluded whooping cranes did not select for wet meadows along the CPRV, the Program deemphasized wet meadow monitoring and research including investigations of methods to improve wet meadow hydrology. Regardless, the Program continues to place importance on wet meadows as an important CPRV ecotype included in Program-managed lands. As a carryover task from the First Increment, the Program's Extension Science Plan includes Big Question #10 as Maintenance Learning for better understanding the hydrological regime that supports an archetypical wet meadow along the CPRV and providing relatively simple tools that can be used by land managers to identify potential wet meadows and to evaluate options for management of wet meadow hydrology.

ii) What did we do?

The Program installed a series of groundwater monitoring wells, two surface water river stage gages, and climatological equipment at two wet meadow sites, one being a native wet meadow (Binfield) and the other former crop ground restored as a wet meadow by the Program (Fox). We paired the data collected between 2013 and 2021from these sources with elevation data collected from LiDAR to quantify depth to groundwater, hydroperiod (number of days when groundwater was above the ground's surface, or surface inundation), and the variability in these metrics over both time and space within each site. We compared across sites to reveal patterns that differed between the native Binfield site and the restored Fox site. We also compared results to published data from other CPRV wet meadow sites to reveal a broad spectrum of hydrologic conditions that have been documented at wet meadows.

We used the quantitative links between hydrology and typical wet meadow vegetation established by Henszey et al (2004) to predict the proportion of each site covered by typical wet meadow vegetation based upon our groundwater depth data. We then calculated the changes in hydrology that would be required to achieve a desired minimum cover of wet meadow vegetation. Through this, we provide a basis and method for developing hydrologic management targets likely to support typical wet meadow vegetation at wet meadow sites.

We developed an analytical model that can be used to predict groundwater levels at wet meadow sites based on site-specific data collected on stage, precipitation, evapotranspiration (ET), and calibrated hydraulic parameters. To provide an alternative for land managers that does not require extensive field data collection, we used publicly available river and ground surface elevation differences to identify areas within the Platte River floodplain more likely to have the shallow groundwater necessary to support a wet meadow.

iii) What did we find?

No two CPRV wet meadows are alike. The ridge and swale topography typical of CPRV wet meadow sites coupled with site-specific ground elevation in relation to river channel elevation results in groundwater depths that are highly dynamic within and between sites and years. Hydroperiod durations are also highly variable even within single wet meadow tracts. The result is a mosaic of vegetation types within and among wet meadow sites which closely follow topographic variations that are associated with relic braided river deposits. The inherent variability within and across sites makes it challenging to define ideal hydrologic, vegetation, and biotic conditions. Realistic hydrologic targets for restoration will likely fall within the limits of conditions reported in this study and will certainly require considerations for site specific constraints related to topography and hydrology.

Binfield has shallower groundwater overall supporting a typical wet meadow vegetation community over a larger proportion of the site. The Fox site has deeper groundwater levels overall with concentrated areas of shallow groundwater constrained to excavations that were performed to improve the site hydrology. Thus, the Program-restored Fox site is able to support wet meadow vegetation over an area limited to the edges of excavations where groundwater is shallow or comes to the surface periodically but does not create ponding. Substantial excavation across the Fox site would be required to reduce depth to groundwater enough to support wet meadow vegetation over 50% of the site. Using the analytical model developed in this study, a comparison of alternatives including surface water recharge and increasing river stage to raise groundwater levels enough to support wet meadow vegetation over 50% of the site demonstrated greater viability for the surface recharge scenario.

The river-ground surface elevation differencing method produced similar maps of hydrologic conditions to those produced with extensive groundwater datasets yet relied only on remotely sensed data, providing a useful screening tool for comparing and assessing hydrology at potential wet meadow sites.

iv) What is next?

The recent publication by Baasch et al. (2022), indicating that whooping cranes occur more frequently as wetland areas within grasslands and agricultural fields increases, has increased uncertainty around the importance of wet meadows for whooping cranes. With refocused interest in wet meadows, the findings in this study can be used by CPR land managers to guide management and expectations of targets, particularly for dryer sites. We provide a model that requires data from at least one groundwater monitoring well to test how alternative management actions including stage change and surface water application will impact site hydrology. By identifying a suite of hydrological variables that are linked to the support of a typical wet meadow plant community, we provide a means to compare across other CPRV wet meadow sites to provide information on the likelihood that a given site will have the appropriate hydrology or could be effectively managed to improve hydrology enough to support a wet meadow community over some pre-determined portion of a site. In addition, we provide a means of using publicly available remotely sensed elevation data to identify CPRV sites that may have better hydrology for supporting a typical wet meadow community.

1. Introduction and background

Wet meadows are a rich and diverse component of the Platte River ecosystem that support a range of wetland functions and habitat types (Davis et al., 2006; LaGrange, 2022). Nearly 150 species of migratory birds, including sandhill cranes, use wet meadows for nesting, feeding, and other activities (Currier et al., 1985; Varner et al., 2020).

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 landcover 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; USBR, 2004; Davis et al., 2006). 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).

It is well documented that wet meadows constitute high-quality foraging sites for Whooping Cranes (Lingle et al. 1991, Austin and Richert 2005, Davis et al., 2006; Geluso et al. 2013, Baasch et al. 2019; Caven et al., 2021). Whooping Crane use of wet meadows and wetlands has been documented throughout the migratory corridor (Austin and Richert, 2005; Jorgensen and Dinan, 2016; Pickens et al., 2016; Austin et al., 2019). Within the central Platte River valley, Whooping Cranes have been observed foraging in wet meadows (Caven et al., 2021). Some research suggests that family groups may be more likely to forage in wet meadows than birds without young (Howe, 1987; Caven et al., 2021). Hydrology, particularly shallow groundwater, provides a critical support for vegetation and biota at CPRV wet meadows. For the PRRIP, understanding the role of hydrology at wet meadows in the CPRV has been a long-term goal under broader objectives to manage land and water resources to benefit four threatened and endangered species[1] (Target Species) (Cooperative Agreement,1997).

However, the selection and preference for wet meadows by Whooping Cranes within the CPRV has been challenging to document, and recent studies have reported inconsistent conclusions. In a study designed specifically to address Whooping Crane use of the CPRV, Howlin and Nasman (2017) found that Whooping Cranes along the CPRV select in-channel riverine habitat over corn, but select corn over grasslands, soybeans, and wet meadows for diurnal use. Examining Whooping Crane selection of diurnal habitat over the wider Great Plains, Baasch et al. (2019) found that Whooping Cranes are at least three times as likely to select rivers, open water, and semi-permanent wetlands, and twice as likely to select lowland grasslands for diurnal use, over all other landcover types, including cornfields. More recently, in a study again focused on Whooping Crane use of the CPRV, Baasch et al. (2022) found areas with increased wetland components located near the Platte River and decreased densities of roads and development had a higher likelihood of occupancy for diurnal use by Whooping Cranes than drier components on the landscape. However, due to crop rotation practices in the CPRV and the inability to

[1] The interior least tern, one of the Program's four target species, was federally delisted on February 12, 2021.

distinguish between those landcover classes over the study period, they also combined corn and soy crops into a lumped agricultural category which may have diminished the relative importance of corn. Their 2022 study combined National Wetlands Inventory with an existing fine scale landcover product developed for the CPRV (Brei and Bishop, 2008) to separate the wetter and dryer portions of grasslands and agricultural fields. For example, their 'meadow-marsh' class incorporated hydrological and vegetation landcovers that were intended to represent herbaceous wetlands thought to be preferred by Whooping Cranes, including wetland components of wet meadows (i.e., swales) and excluding xeric components (i.e., ridges). Baasch et al. (2022) hypothesized that results from their study differed from previous Program publications due to the scale and aggregation methods for landcover classes utilized across analyses.

Differing conclusions may be related to substantive inconsistencies in defining and delineating wet meadows (e.g., see Appendix A and Chavez Ramirez and Weir, 2010). A notable point of discussion has been whether relatively upland, xeric areas (i.e., the ridges of ridge-swale topography) should be included in wet meadow designations (e.g., Currier 1995; Whiles and Goldowitz, 1998; Henszey et al., 2004; Meyer et al., 2008; Baasch et al., 2022). In 2005, when the Land Plan was developed for the Platte River Recovery Implementation Program (PRRIP or Program), wet meadows were defined in terms of habitat characteristics that are important for Whooping Cranes. This definition incorporated proximity to channels and disturbance, vegetation, hydrology, topography, and food sources, and importantly, included upland components of meadows. More recent studies have defined wet meadows in relation to national wetland-based designations (e.g., Tiner, 2016; Baasch et al. 2019; 2022) that incorporate soil, vegetation, and hydrologic conditions; excluding upland (xeric) areas. While further research is likely needed to clarify this point, wet meadows continue to be recognized as valuable habitat within the CPRV.

For this report, we retain the original PRRIP definition for wet meadows, while acknowledging that within the CPRV, the wetland portions of wet meadows occur embedded within lowland grasslands forming mosaics of emergent, sedge meadow, mesic prairie, and xeric vegetation accompanied by a wide range of moisture, soil, and biotic conditions. As discussed herein, highly variable groundwater levels likely cause boundaries between landcover types to shift over time.

During the First Increment, the Program's Land Plan set the goal of acquiring and restoring 640 acres of wet meadow habitat at each Program habitat complex with the objective of providing additional wetland diurnal habitat for the endangered Whooping Crane. However, in practice the amount of wet meadow habitat that could be acquired and restored has often been limited based on 1) the fact that most riparian grasslands have been converted to irrigated cropland, 2) lack of willing sellers, or 3) the prohibitively high cost of purchasing irrigated cropland and attempting to convert it back to riparian grasslands. Early in the First Increment, it was also thought that flow releases to benefit wet meadows could consume a major fraction of the Environmental Account[2] (EA) water budget (PRRIP, 2012). However, as water and land priorities adapted in response to learning about Whooping Crane use of wet meadows, water-use to support wet meadows decreased in priority.

[2] The EA is an annual allotment of water set aside in an upstream reservoir that is managed and released by the USFWS and PRRIP for the benefit of the Target Species.

Despite the shift in priorities, the Program continues to recognize wet meadows as vital components of the CPRV. Today, the Program manages over 5,000 acres of lowland grassland, with up to 41% containing wet meadow habitat (Appendix A). Management actions at wet meadow sites include rotations of seeding, livestock grazing, haying, mowing, and prescribed fires. At least one Program-managed site (Fox) has undergone topographic recontouring and discharge of groundwater at the land surface to decrease groundwater depths and 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.

Understanding the hydrology of wet meadows is critical for maintaining and restoring wet meadow sites within the CPRV. Wet meadows are sub-irrigated wetlands and grasslands that lack standing water for most of the year but become inundated during periods of seasonal high water, typically during winter and spring months (Mitsch and Gosselink, 2015; Tiner, 2016; Baasch et al., 2022). Within the central Platte River Valley (CPRV), wet meadow hydrology is controlled by dynamic linkages between river stage, precipitation, and evapotranspiration (ET), wherein fluctuating shallow groundwater tables temporarily extend above the land surface, particularly in low-lying swales (Brinley Buckley et al., 2021). Shallow groundwater is an essential component of CPRV wet meadows; supporting saturated soils during low-water periods, sustained surface inundation, and conditions for a variety of hydrophytic grasses, sedges, rushes, and wetland wildflowers (Henszey et al., 2004; Tiner, 2016, Brinley Buckley et al., 2021; LaGrange, 2022).

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 for 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 to improve the characterization and understanding of CPRV wet meadow hydrology. Methods presented herein are intended to provide utility for managers of other wet meadow sites throughout the CPRV.

This study was designed to utilize an existing hydrological and climatological dataset that was collected at two Program managed wet meadow study sites (Fox and Binfield) between 2013 and 2021. Objectives for the study are to improve the understanding and characterization of hydrological variability that characterizes CPRV wet meadows, quantify relationships between hydrology and vegetation to develop integrated management targets, and to develop tools and methods that inform management and restoration for wet meadows throughout the central Platte River Valley.

2. Study Sites and Monitoring Networks

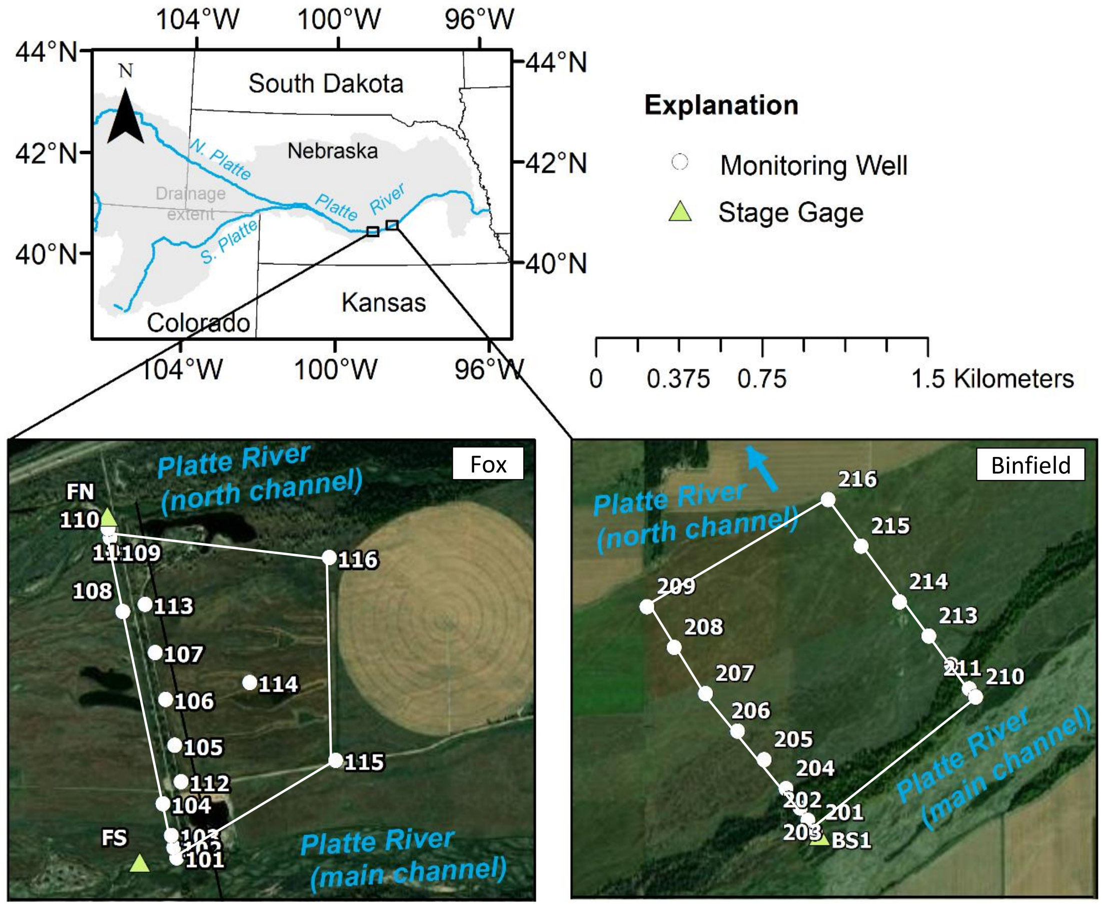
The study area for this project includes a 145-km stretch of the CPRV in Nebraska that serves as critical stopover habitat for migrating Whooping Cranes. Within the study area, the Platte River forms a wide, braided channel with shallow depths (typically < 1m), extensive sand bars, and grassland floodplains. Within the CPRV, wet meadows commonly occur in historic braided river deposits within a few kilometers of the active channel (Currier and Henszey, 1996).

The majority of data used in this study was obtained from two wet meadow tracts that are located on vegetated islands bounded by primary (south) and secondary (north) channels of the Platte (Figure 1). Sites were selected based on site history, management status, and data availability and both are managed by the Program as part of a broader effort to restore and protect habitat for Program Target Species. 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. Fox is managed through seeding (2012 and 2017), grazing, haying, controlled burns, and rest periods. Topographic re-contouring was also performed at Fox in 2014 to improve hydrologic conditions and likelihood of inundation at the site. This involved excavating five elongated depressions to depths that were determined to produce seasonal standing water. Additionally, groundwater was pumped and discharged within depressions for approximately one month during spring and fall migration periods (typically Feb-March and October-November) to supplement ponded water. Details about excavations at Fox can be found on the PRRIP website (PRRIP, 2012). 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. 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 consistently less and in some years no obligate or facultative wetland species (PRRIP 2016, 2019, and 2022). Monitoring networks at each study site were installed in 2013 and include 16 shallow observation wells (3-meter depths), two 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). For a more detailed summary of data collection methods and instruments deployed at the two wet meadow study sites, please see PRRIP (2015).

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. 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. Stage gages were anchored to the streambed using existing structures or steel rods that were manually driven into the streambed. Data gaps in river stage were previously filled through linear regression models developed for each gage site. Quality control methods, including regression information, is included in the 2021 Wet Meadows Data QC Report (PRRIP 2021).

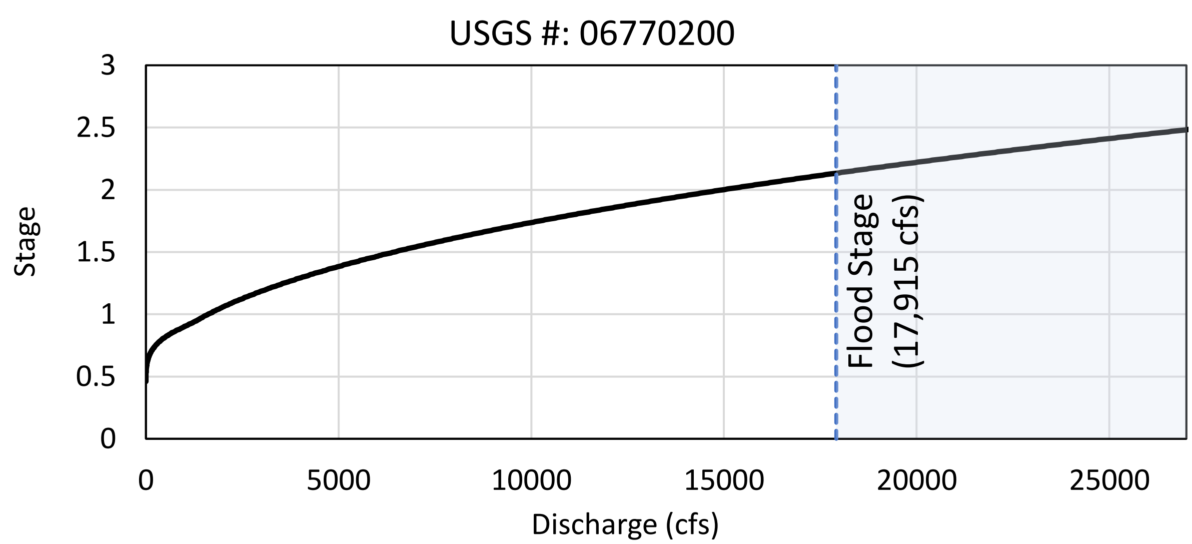
Weather stations were maintained by the High Plains Regional Climate Center and meteorological data and method details are publicly available (HPRCC 2021). HPRCC provides daily potential evapotranspiration (ET_0) values estimated using the daily Penman equation with the wind function for reference crop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

River stage and groundwater levels are tightly linked within Platte River wet meadows, particularly in wet meadow areas that are close to the active river channel (Wesche et al., 1994; Chen, 2007; Brinley Buckley et al., 2021). The Platte River is highly modified by human activities that include numerous dams, diversions, and hydropower plants. Residential and agricultural parcels occur throughout the Platte River floodplain. These factors, along with the regulatory flood stage (Figure 2), pose important considerations and constraints for potential management actions that require altering river stage and discharge.



62

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



65

66 *Figure 2 – Stage-discharge relationship for the USGS Kearney gage on the Platte River.*

3. Characterizing the Hydroregime

Wet meadows are characterized by significant hydrological variability that includes spatial and temporal variations in groundwater depth and inundation (Tiner, 2016; Brinley-Buckley et al., 2021). Since groundwater depth and inundation are primary controls for soil moisture, vegetation, and biota, characterizations of groundwater depth and inundation are important for understanding key aspects of habitat and ecological function at CPRV wet meadow sites (Henszey et al., 2004; Davis, 2006).

Within the CPRV, spatiotemporal variations in groundwater level occur due to an interplay of dynamic drivers that include river stage, precipitation, evapotranspiration (ET), and overland flooding (Wesche et al., 1994; Chen, 2007; Brinley Buckley et al., 2021). Areas close to the channel fill and drain quickly with changes in river stage with high-hydraulic connectivity between the river and floodplain (Chen, 2007). Undulating ridge-swale topography promotes spatial variations wherein upland ridges are further from the water table (and thus dryer) while lowland swales are closer to the water table (wetter). Evapotranspiration varies not only due to climactic changes but also vegetation type and groundwater depth (Yue et al., 2016). Because of these dynamic factors, wet meadows occur as a heterogeneous patchwork of vegetation and biota that include local adaptations to variable hydrology and soil conditions which vary across seasons and years (Currier, 1995; Whiles and Goldowitz, 2001, 2005; Henszey et al., 2004; Davis et al., 2006).

Methods which evaluate hydrology over a range of spatial and temporal scales are therefore essential for capturing hydrologic variability at wet meadow sites. Previously, authors have sought to quantify the hydroregime (i.e., hydrologic regime) at wet meadow sites. Wesche et al. (1994) quantified groundwater depths and inundation dynamics at three CPRV wet meadows, however, manual collection methods limited the analysis to relatively low spatial and temporal resolution. Brinley-Buckley et al. (2021) used timelapse imagery and wavelet coherence to quantify the hydroregime at a CPRV wet meadow site on Mormon Island at a high-temporal resolution (<daily). They evaluated inundation frequency and its relationship to hydrologic variables and concluded that stage was the dominant control on groundwater levels. However, the time-lapse imagery captured only a small area fraction of a wet meadow site, limiting the spatial coverage of results. As wet meadows in the CPRV become increasingly sparse and degraded, it is ever more important to understand and characterize the hydrology at remaining wet meadow sites to identify reference conditions and maintenance requirements.

The objective for the current study is to characterize the hydroregime at CPRV wet meadow sites using a method that captures a broad range of spatial and temporal variations. To do this, we quantify key hydrologic metrics, including depth to groundwater (DTGW) and hydroperiod using an eight-plus year dataset of hourly groundwater measurements and interpolated groundwater elevation surfaces. We calculate exhaustive statistics for groundwater depths across space and time to compare and contrast hydrology across wet and dry portions of sites. Groundwater and inundation summary statistics provide a basis for understanding hydrologic variability at CPRV wet meadows and may be used to define hydrologic targets for management.

3.1. Methods

3.1.1. Depth-to-Groundwater (DTGW) Statistics

Groundwater depths are a key aspect of the hydroregime at wet meadow sites. Herein, we characterize groundwater depths at two wet meadow sites using summary statistics from point- and area-based datasets. Point-based data includes hourly DTGW values from wells located at each study site. Area-based data includes interpolated groundwater elevation surfaces used to calculate site-wide DTGW values. We compare statistics from the two datasets and demonstrate how site-wide (area-based) data results in better characterizations of variability.

Point-statistics were calculated using hourly DTGW values from monitoring wells, where DTGW was calculated by differencing the measured hourly groundwater elevation (GWE) from the surveyed ground surface elevation (GSE) at each well location (Figure 1). By convention, DTGW is reported as negative for groundwater elevations below ground surface. Point-based statistics include: the minimum, maximum, mean, median, range, and standard deviation of DTWG for each year and total for the study period.

Area-based statistics were calculated from gridded DTGW surfaces that were generated from well and LIDAR data. LIDAR data were 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 a final bare earth elevation gridded areal resolution of 0.91m (grid size = 0.91 by 0.91 m). For each day during the study period, a groundwater elevation surface was created by linearly interpolating between daily mean groundwater elevations at well points within a regularly spaced grid (e.g., Figure 3). The cell size and boundary of the grid were set to match the LIDAR ground surface raster. Potentiometric contours from existing numerical groundwater flow models show smooth and evenly distributed hydraulic heads between wells which suggest linear interpolation is appropriate (See Appendix C for supporting figures). Each daily groundwater elevation surface was then subtracted from the corresponding LIDAR ground surface elevation to produce a daily DTGW gridded raster (Figure 3). Resulting surfaces contained 1,341,543 grid cells for Binfield and 1,255,258 grid cells for Fox with corresponding total areas of 1.23 km^2 and 1.15 $km^{2,}$ respectively. Area-based statistics were then calculated for each grid cell using yearly sets of DTGW rasters, including the minimum, maximum, mean, median, range, and standard deviation. The result is a spatial distribution of DTGW statistics across each site.

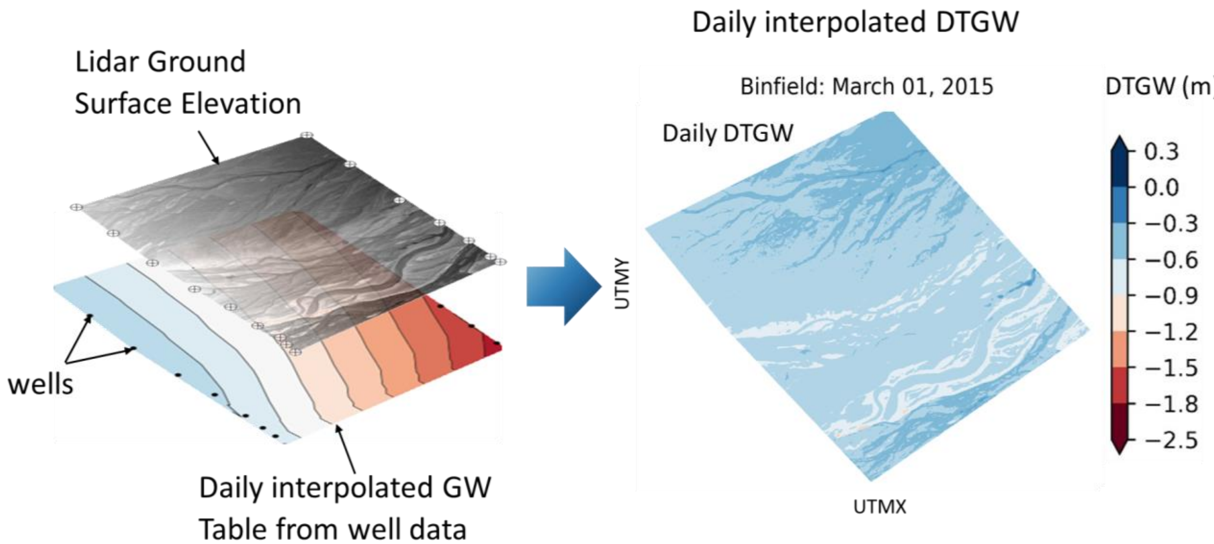


Figure 3 - Illustration of method for generating spatial depth to groundwater statistics.

3.1.2.Depth-to-Groundwater area percentages

Since wet meadow groundwater depths vary significantly across space and time, statistics which integrate spatial and temporal dynamics are necessary. The area percentage of a site over which a given range of groundwater levels occur during a set period of time is one such statistics that has been used to describe highly variable groundwater depths (Wesche et al., 1994). We replicate the methodology of Wesch et al. (1994) but with higher-frequency and higher spatial resolution datasets to calculate DTGW proportions. Previously described DTGW surfaces were used to calculate the area proportion of each site for which a range of groundwater depths occurred. The minimum, maximum, spring (Mar-May) median, and summer (Jun-Aug) median area percentage was calculated for each DTGW range for the period between March 2013 to September 2020.

We replicate 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. Discharge data from nearby Platte River gages (USGS 06768000 near Overton and USGS 06770500 near Grand Island) record slightly lower annual discharge for the Wesche study period compared to the period of record in this study. Therefore, water levels may be skewed towards wetter conditions for the Fox and Binfield datasets. Regardless, we expect comparisons to be informative. A map with locations of study sites relative to Wesche et al. (1994) sites is presented in Figure 4. A major objective of the area proportion exercise was to compare hydrology at the two study sites to other wet meadows in the CPRV.

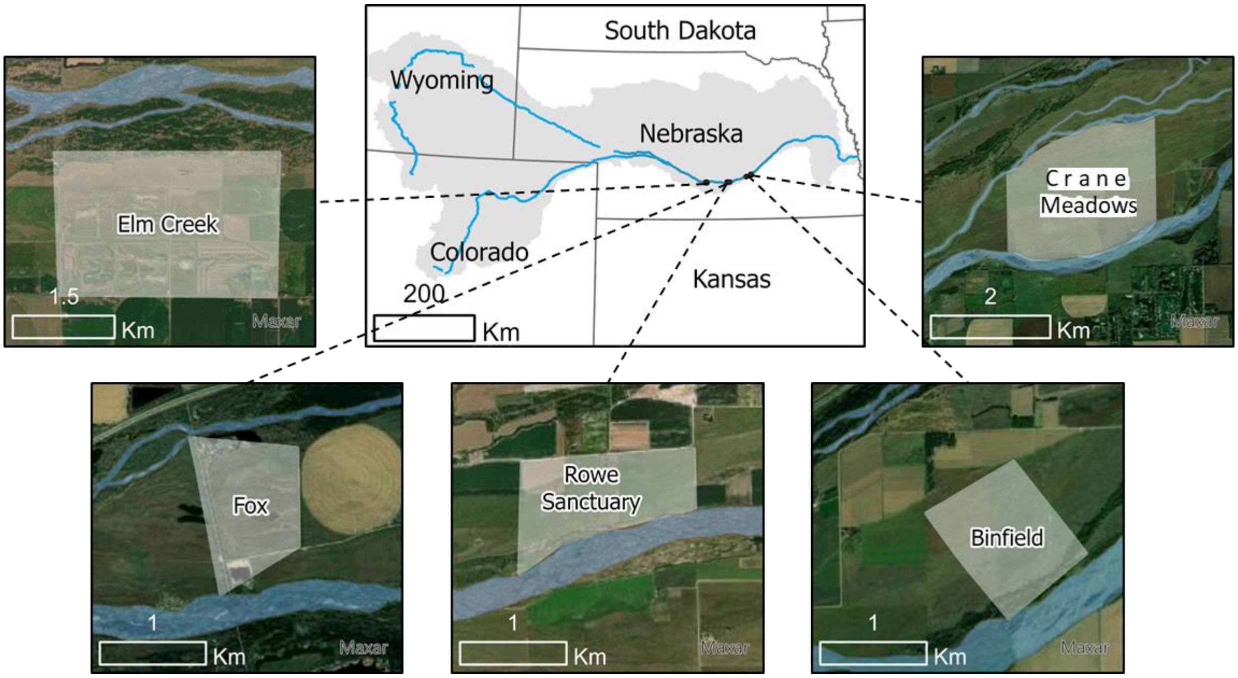


Figure 4 - Map showing the location of Binfield and Fox sites relative to the three Wesche et al. (1994) study sites.

3.1.3.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, both CPRV wet meadows, using manual measurements during a 1-year study period. Brinley-Buckley et al. (2021) reported mean hydroperiod duration of 141 days, typically spanning from 10 December to 1 May, and generally peaking in the early spring at Crane Meadows over a 6-year period. They used timelapse imagery positioned to record water extent at a single location. 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 the current report, groundwater elevation and inundation dynamics vary significantly across wet meadow sites. The hydroperiod for two swales at a single wet meadow tract is expected to vary due to a range of factors that include the distance from the river and ground surface elevation. Therefore, to account for spatiotemporal variations in hydroperiod, we quantify both the classic hydroperiod, calculated as the number of days that a site records standing water, as well as the proportion of each study site that has water at or above the surface for each day during the study period. To calculate the latter, we use the methodology described previously for generating gridded DTGW surfaces, and then calculate the percentage of the total site area with DTGW values at or above the ground surface for each day. Daily proportions from each site were then used to calculate the annual mean, median, minimum, and maximum proportion of each study site that records groundwater at or above the surface for each year.

3.2. Results

3.2.1. Depth-to-Groundwater (DTGW) Statistics

Depth-to-groundwater statistics for point locations (i.e., wells) are summarized in Table 1. During the study period, well DTGW values ranged from -2.96 to 0.35 m at the Fox site and from -1.72 to 0.83 at the Binfield site, with median values for all wells of -1.55 and -0.76 m at each site, respectively. The median range of DTGW values for all years was 1.90 m at the Fox site and 1.79 m at the Binfield site. Three wells at Fox, and thirteen at Binfield, recorded minimum (i.e., shallowest) DTGW values that were above ground surface (i.e., positive values). The Fox site records well DTGW values that are on average 0.79 m deeper than the Binfield site.

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

	Loc	Min (m)	Max (m)	Mean (m)	Median (m)	Range (m)
Fox	101	-2.60	-0.73	-1.70	-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	-1.90	-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	-1.50	-1.50	1.92
	116	-2.42	-0.03	-1.68	-1.71	2.39
	Site Median	**-2.33**	**-0.25**	**-1.55**	**-1.55**	**1.90**
Binfield	201	-0.99	0.42	-0.36	-0.35	1.40
	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	-0.80	1.99
	207	-1.72	0.43	-1.10	-1.10	2.15
	208	-1.31	0.56	-0.68	-0.68	1.87
	209	-1.38	0.40	-0.73	-0.73	1.78
	210	-1.53	-0.23	-0.99	-0.99	1.30
	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	1.80
	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 Median	**-1.39**	**0.41**	**-0.76**	**-0.76**	**1.79**

Depth-to-groundwater statistics for area surfaces (i.e., DTGW surfaces) are summarized in Table 2. Because statistics for individual grid cells cannot be reasonably tabulated (i.e., >1,000,000 grid cells for each site), the median value for all grid cells at each site is reported in Table 2. These values are comparable to site-median values reported for well-based statistics in Table 1. DTGW statistic values vary substantially across sites and contain a broader range than is reported in Table 2. Statistics for interpolated DTGW surfaces were calculated for each grid cell. Groundwater depths from gridded surfaces ranged from -2.06 to -0.32 m at the Fox site and -1.21 and 0.52 m at the Binfield site, with corresponding median values of -1.36 m and -0.63 m and ranges of 0.81 m and 1.02 m at Fox and Binfield respectively during the study period.

DTGW surfaces reveal significant spatial variations across sites in all reported statistics (minimum, maximum, mean, median, range, and standard deviation). Since each grid cell is comparable to an individual well, these figures highlight how data from wells are biased by location selection. At the Binfield site, the shallowest annual mean and median DTGW values occur in low-lying swales, with patterns that mimic braided channels. Conversely, at the Fox site, shallowest mean and median DTGW values are found within and at the perimeter of excavated wetlands. Interestingly, areas further from the channel (north and south channels) tend to have greater annual range and standard deviations for DTGW. Variability across years resulted in a range of median DTGW values of 0.36 m and 0.3 m at the Fox and Binfield sites respectively, highlighting an approximately 25% inter-annual variability that is attributed to wet and dry years. Gridded statistics for all years provided in Appendix C. Additional plots exploring relationships between groundwater levels and distance from the channel are included in Appendix D.

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

	Year	Min (m)	Max (m)	Mean (m)	Median (m)	Range (m)	Sigma
Fox	**Site Median**	-1.61	-0.81	-1.36	-1.36	0.81	0.18
Binfield	**Site Median**	-1.03	-0.08	-0.63	-0.63	1.02	0.24

In comparing the well-based and area-based sets of summary statistics, site mean and median DTGW values are similar across datasets. Both well-based and area based DTGW statistics record deeper DTGW values at the Fox site that are on average 0.72 m deeper than the Binfield site. In general, area-based statistics record shallower mean and median DTGW values than well statistics. This indicates potential for bias in well placement locations, specifically that wells may be placed on higher elevation portions of the site. While not reported in Table 2, the absolute range of DTGW values (i.e., minimum to maximum for all grid cells) was significantly greater in results from area-based methods. The absolute range in DTGW values at Fox was -9.1m to 4.8 m and at Binfield was -2.7 to 1.4 m. While the Binfield absolute values are reasonable, extreme DTGW values at the Fox site are unlikely to be found at wet meadows. A close inspection of spatial statistics for the Fox site shows how an existing bike trail, road, and sand pit significantly influence topography and in turn, DTGW values. Since no wells are located on bike trails, roads, or sand pits, areas influenced by these features are expected to result in outliers DTGW values that skew area-based results towards deeper values.

In summary, a wider range of DTGW values were represented in surfaces, indicating that area-based methods capture a broader range of groundwater depth conditions at a site than were measured at wells. However, calculating summary statistics on the exhaustive area-based set of DTGW values yields a narrower range of minimum and maximum statistics. Both methods indicate that Fox had statistically lower mean, median, and annual ranges of DTGW values, and site wide DTGW values that are typically 0.72-0.79m deeper than the Binfield site during the study period. By comparing resulting statistics from the two methods, we demonstrate how sparse point-based measurements (i.e., wells) may contain bias based on well placement locations.

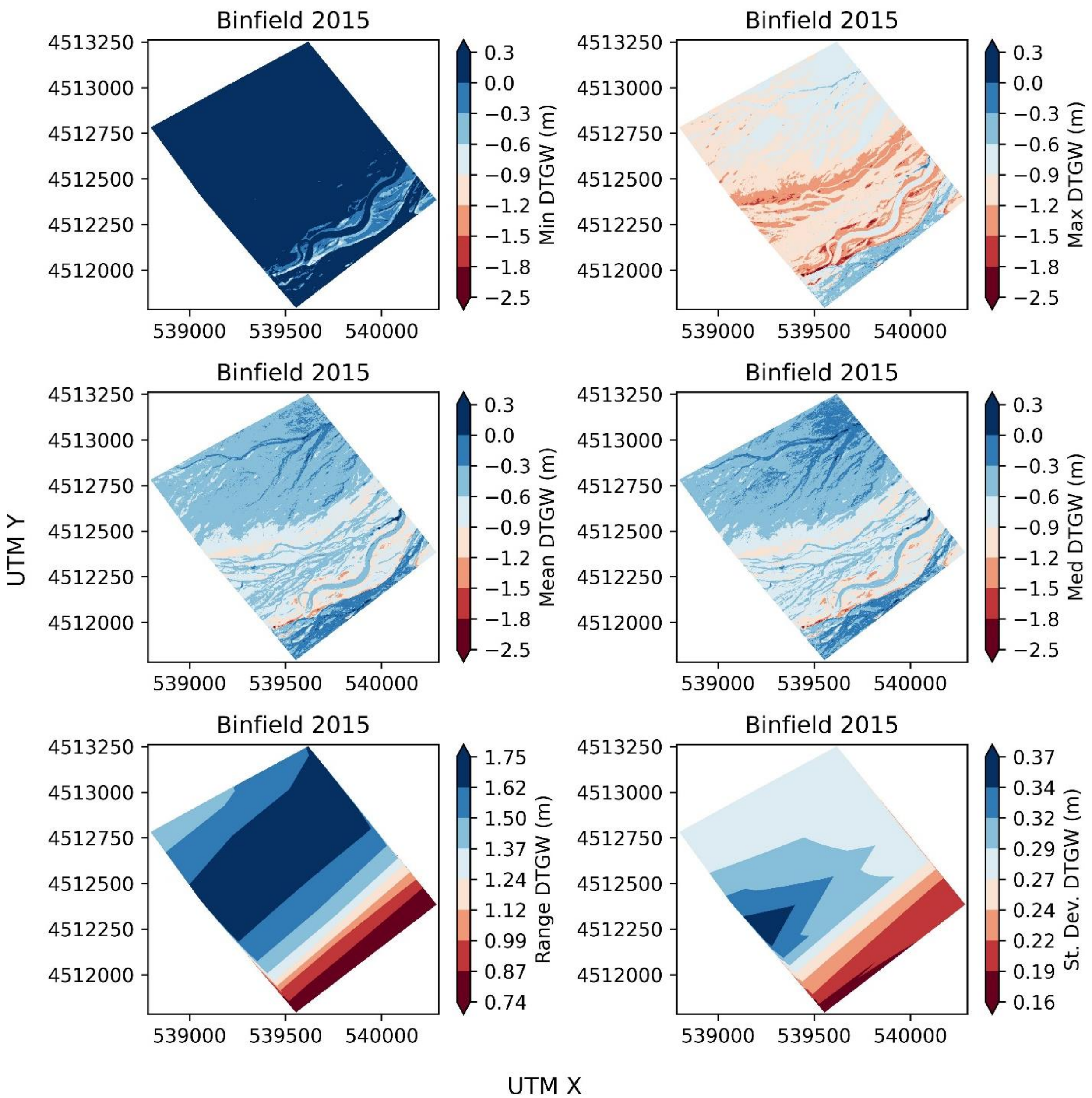


Figure 5 - Area-based depth to groundwater (DTGW) gridded annual statistics for 2015 at the Binfield Site. Negative and positive values indicate groundwater levels below and above ground surface, respectively.

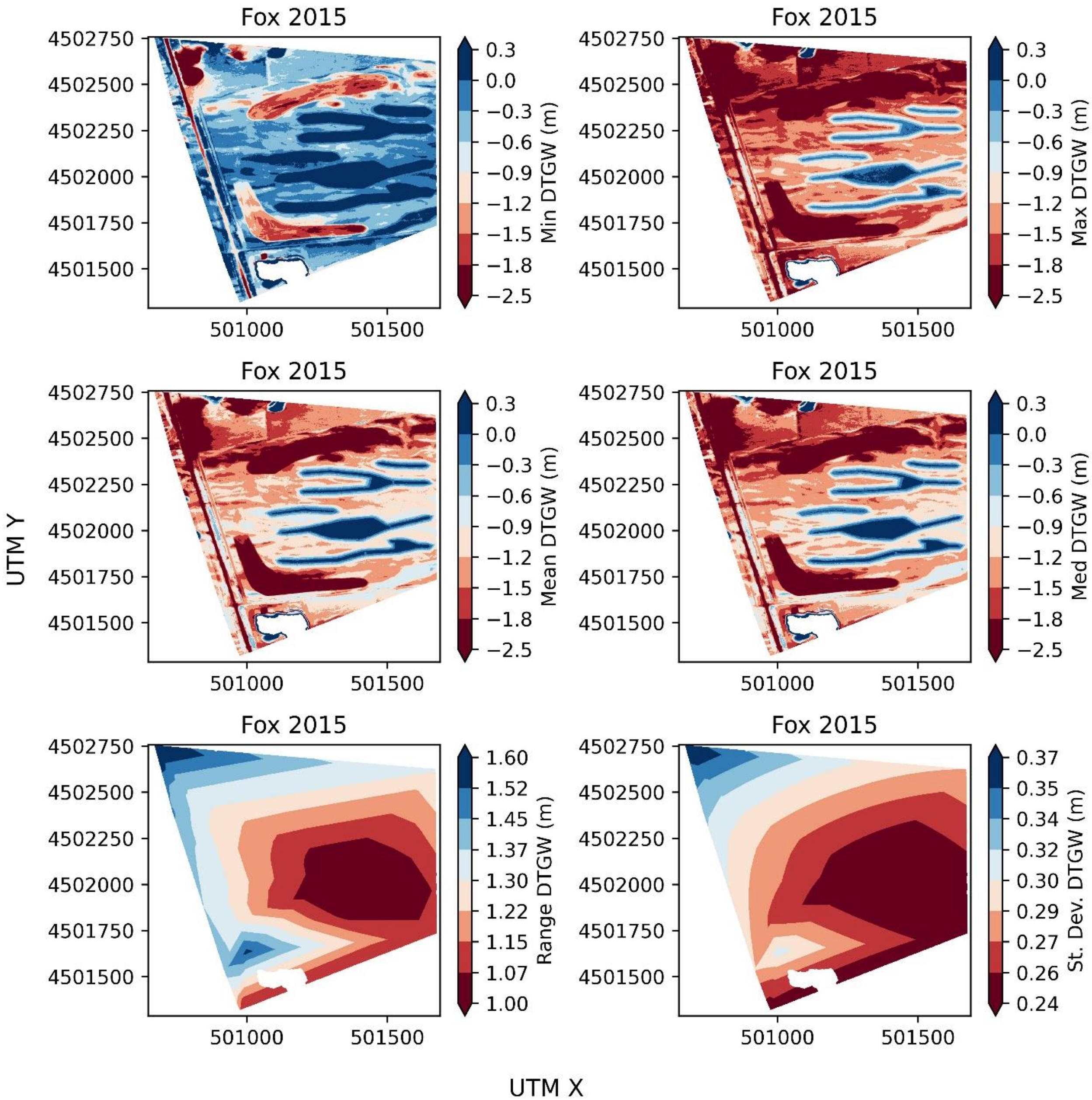


Figure 6 - Area-based depth to groundwater (DTGW) gridded annual statistics for 2015 at the Fox Site. Negative and positive values indicate groundwater levels below and above ground surface, respectively.

3.2.2. Depth-to-Groundwater area Percentages

Area percentages for the Binfield, Fox, and Wesche et al. (1994) study sites (Elm Creek, Rowe Sanctuary, and Crane Meadows), are presented in Table 3. The datasets reveal the broad range of groundwater depths within and across wet meadows. The wettest site, Crane Meadows, records 94% of the site area having median spring groundwater depths within 0.6m of the surface. Sites become increasingly dryer towards the west, with Binfield recording 73% of site area with median spring groundwater depths within 0.6m of the surface, Rowe Sanctuary recording 31%, Fox recording 15%, and Elm Creek recording less than 0.5%. Across all sites, spring groundwater depths were shallower than summer values. All sites recorded at least 1% of the site area having groundwater levels above the ground surface (i.e., DTGW>0) for the maximum groundwater surface. Figure 7 shows the spatial distribution of tabulated statistics for the Fox and Binfield sites in 2015, with similar patterns to those previously described.

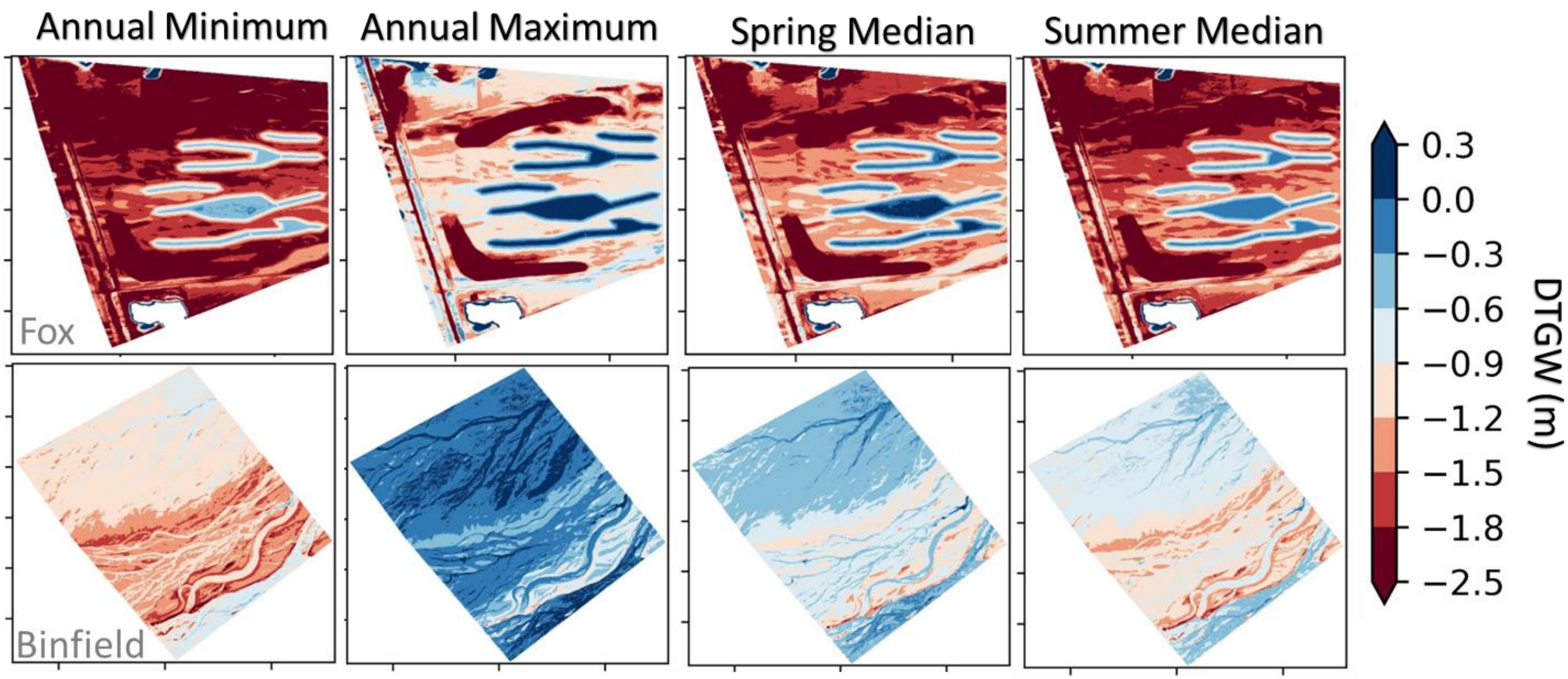


Figure 7 - Plot showing spatial distribution of depth to groundwater (DTGW) statistics at Fox and Binfield sites. Note, positive and negative values indicate groundwater depths above and below ground surface, respectively.

Table 3 - Area-based groundwater depth statistics for Binfield and Fox (March 2013 to June 2021) and Elm Creek, Rowe Sanctuary, and Crane Meadows (July 1988 to September 1992).

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

	DTGW (m)			min	max	Spring median	Summer median
Binfield			> 0	0.0	89	3.7	2.0
	0.0	to	-0.3	0.0	6.0	23	5.0
	-0.3	to	-0.6	0.4	3.0	46	10
	-0.6	to	-0.9	6.5	1.0	24	57
	-0.9	to	-1.2	44	0.0	4.0	23
	-1.2	to	-1.5	36	0.0	0.0	4.0
	-1.5	to	-1.8	12	0.0	0.0	0.0
		<	-1.8	1.2	0.0	0.0	0.0
Fox			> 0	0.35	21	5.7	4.0
	0.0	to	-0.3	0.1	21	4.0	4.0
	-0.3	to	-0.6	2.0	28	5.0	4.0
	-0.6	to	-0.9	4.3	10	11	6.0
	-0.9	to	-1.2	3.9	5.0	23	17
	-1.2	to	-1.5	4.9	5.0	21	28
	-1.5	to	-1.8	13	6.0	11	15
		<	-1.8	72	3.0	20	23
Elm Creek			> 0	0.0	1.0	0.0	0.0
	0.0	to	-0.3	0.0	9.0	0.0	0.0
	-0.3	to	-0.6	0.0	38	<0.5	0.0
	-0.6	to	-0.9	<0.5	39	6.0	2.0
	-0.9	to	-1.2	10	11	36	21
	-1.2	to	-1.5	46	2.0	50	55
	-1.5	to	-1.8	37	0.0	8.0	20
		<	-1.8	7.0	0.0	<0.5	2.0
Rowe Sanctuary			> 0	<0.5	4.0	<0.5	<0.5
	0.0	to	-0.3	<0.5	46	<0.5	<0.5
	-0.3	to	-0.6	<0.5	37	31	<0.5
	-0.6	to	-0.9	8.0	12	58	33
	-0.9	to	-1.2	68	1.0	11	56
	-1.2	to	-1.5	33	0.0	<.5	11
	-1.5	to	-1.8	1.0	0.0	0.0	<0.5
		<	-1.8	0.0	0.0	0.0	0.0
Crane Meadows			> 0	0.0	46	24	0.0
	0.0	to	-0.3	0.0	40	53	<0.5
	-0.3	to	-0.6	6.0	11	17	38
	-0.6	to	-0.9	55	2.0	5.0	45
	-0.9	to	-1.2	30	1.0	1.0	14
	-1.2	to	-1.5	8.0	<0.5	<0.5	2.0
	-1.5	to	-1.8	1.0	0.0	0.0	1.0
		<	-1.8	<0.5	0.0	0.0	<0.5

3.2.3.Hydroperiod

Resulting hydroperiod durations were 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 (# Days in Table 4 and Table 5). Note, 2013 data are reported but not included in summary metrics because data from part of the year were missing. These hydroperiod duration results are not typical. Whiles and Goldowitz (2001) reported a 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.

A likely reason for these differences is the broad scale at which inundation is recorded in this study as compared to previous studies. Previously, hydroperiod has been reported for individual swales at wet meadow sites, whereas we report site-wide inundation that spans multiple swales. It is not surprising that the site-wide analysis yielded longer inundation periods. Interestingly results suggest that both sites contain regions that remain perennially inundated during most years. At Fox, areas with long hydroperiod duration are concentrated at excavations that were created as part of restoration efforts described previously. At Binfield, long hydroperiods characterize swales near the channel and at remnant channels further inland (Figure *8*).

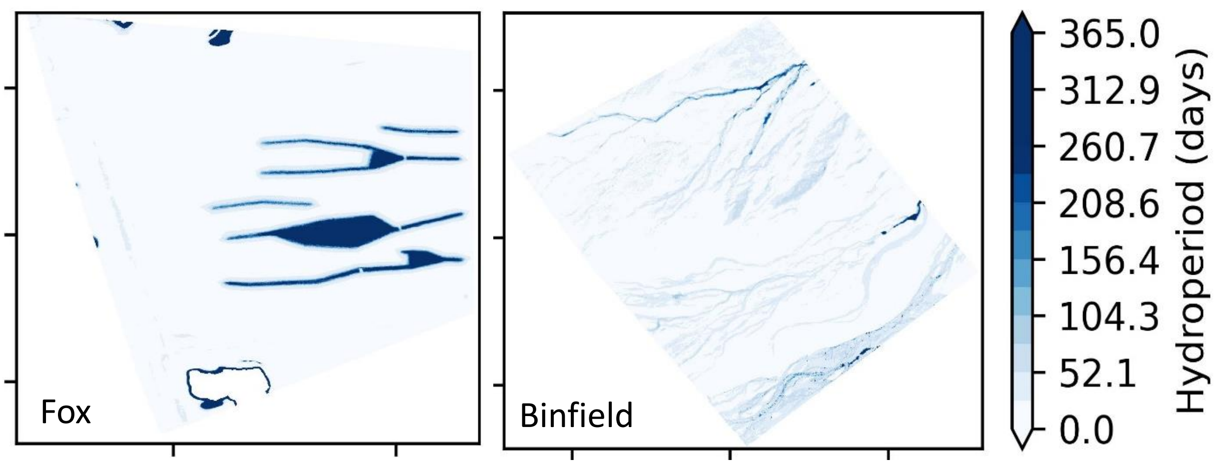


Figure 8 - 2015 hydroperiod across the Fox and Binfield wet meadow study sites.

To account for the influence of perennially inundated depressions on hydroperiod results, a threshold area was determined for each site that represents the theoretical lower limit separating ponded water and temporary standing water. This threshold was calculated as the 60th percentile of daily area inundation and approximately corresponds to the area of permanent inundation at each site (4.7% and 0.32% at Fox and Binfield respectively). An updated hydroperiod was calculated using the threshold (P_{60}) that excludes permanently inundated areas at each site. P_{60} hydroperiod durations range from 19 to 239 days at Fox and from 52 to 233 days at Binfield and serve as more representative hydroperiods for the average swale at each site (e.g., Figure 7). P_{60} hydroperiod results are also more similar to the ranges reported in other studies. The selected threshold impacts the resulting hydroperiod and may require careful consideration during future applications. However, this analysis still lends insight into wet meadow hydrology. Since the applied threshold is consistent across years, the P60 hydroperiod is comparable across years. Hydroperiod durations vary across years, demonstrating variability in inundation dynamics 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 – Annual hydroperiod statistics for the Fox site. P_{60} indicates 60th percentile threshold for daily inundated area percentage.

		Hydroperiod (Days)		Annual percentage of site area with standing water			
	Year	All	P_{60}	Mean (%)	Median (%)	Min (%)	Max (%)
Fox	2013[1]	295	19	1.45	0.62	0.05	6
	2014	365	40	2.15	2.05	0.47	7
	2015	365	168	5.91	4.59	2.40	22
	2016	366[2]	239	6.45	6.10	2.25	13
	2017	365	172	4.28	4.23	1.09	9
	2018	365	104	3.48	3.93	0.56	7
	2019	365	229	5.83	5.34	2.99	13
	2020	366[2]	170	3.69	3.16	0.52	15

Note: [1]2013 data span a partial year; [2]366 days indicates leap year

Table 5 - Annual hydroperiod statistics for the Binfield site. P_{60} indicates 60th percentile threshold for daily inundated area percentage.

				Annual percentage of site area with standing water			
	Year	# Days	# Days (P_{60})	Mean (%)	Median (%)	Min (%)	Max (%)
Binfield	2013[1]	180	62	0.45	0.05	0	18
	2014	324	52	0.28	0.09	0	17
	2015	364	194	5.30	0.39	0	89
	2016	365	164	1.6	0.25	0	40
	2017	349	154	0.96	0.15	0	28
	2018	350	142	0.92	0.20	0	22
	2019	364	233	3.1	0.60	0	79
	2020	315	133	0.81	0.04	0	35

Note: [1]2013 data span a partial year.

Figure 9 and Figure 10 plot the inundated proportion of each site along with groundwater elevation at an example well and river stage. Days when the area percentage was above the determined P_{60} threshold are shaded in gray and represent days that are counted towards the P_{60} hydroperiod. Notably, inundated area percentage of the site peaks during winter, spring, or early summer depending on the year and is highly variable, showing interspersed wetting and drying cycles rather than a slow rise and fall marking the start and end of the hydroperiod duration. Like Brinley Buckley et al. (2021), our data record a bimodal distribution during wet years (e.g., 2015 and 2016) that includes a second peak in late spring. They identified a relationship between hydroperiod and streamflow during these periods, which can also be observed in our data (e.g., Figure 9 and Figure 10). The variability in the area coverage of inundation reveals a dynamic hydroperiod that varies between across swales at individual wet meadow tracts and changes year-to-year.

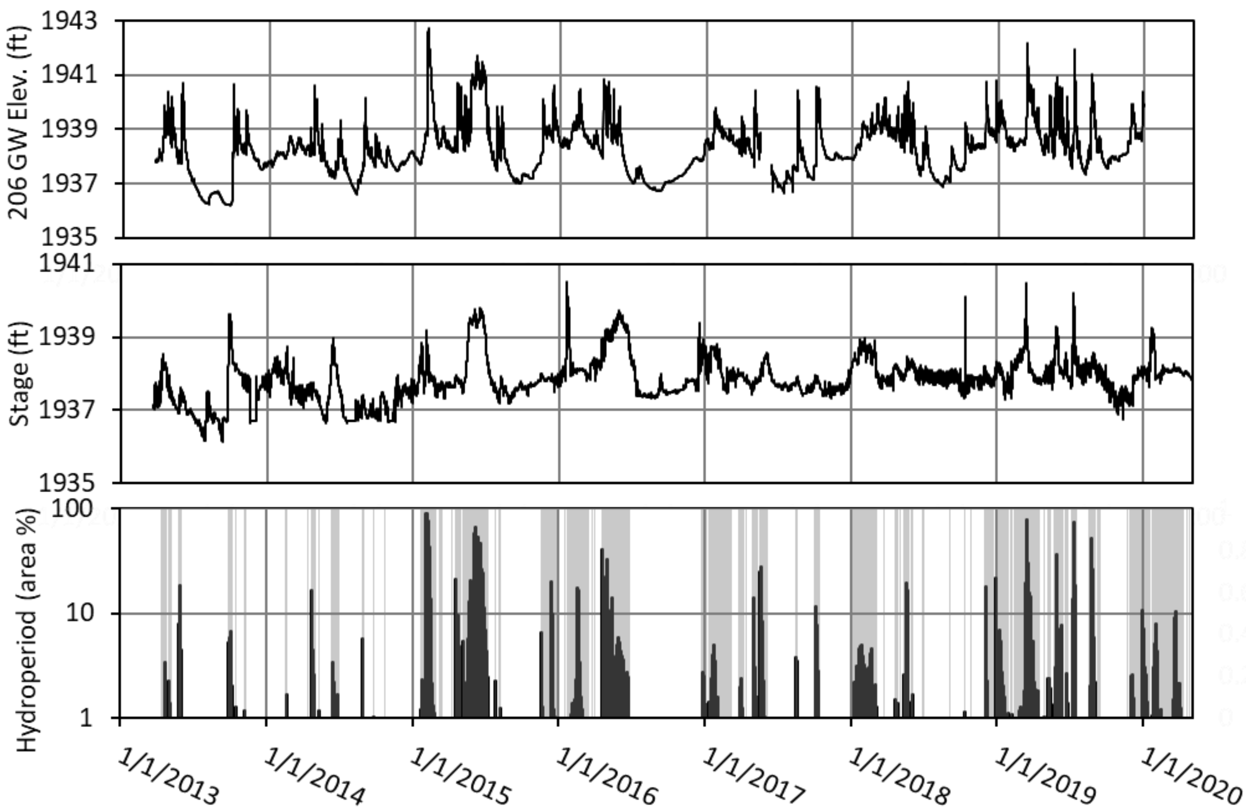


Figure 9 - Groundwater elevation (Well 206), river stage, and inundated area percent for the Binfield site. Gray shading indicates the area percentage was above the P_{60} threshold value.

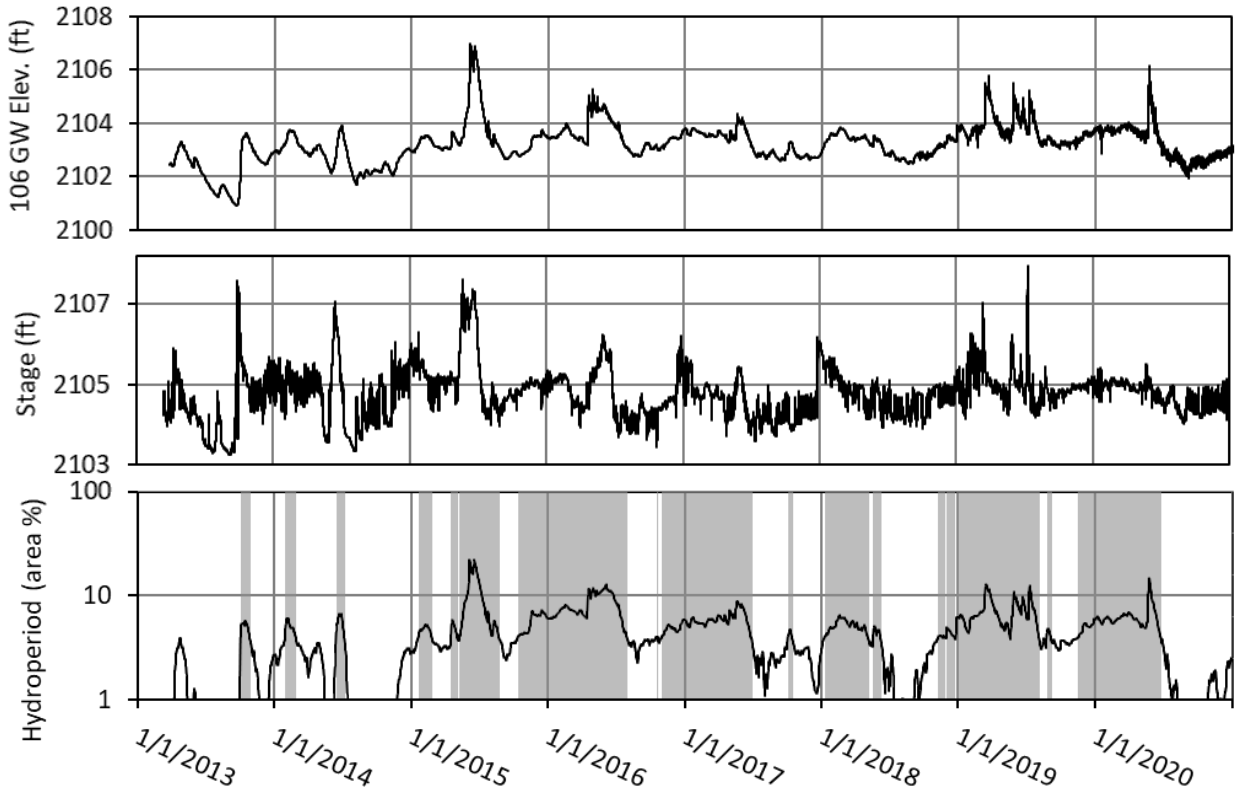


Figure 10 - Groundwater elevation (Well 106), river stage, and inundated area percent for the Fox site. Gray shading in the indicates the area percentage was above the P_{60} threshold value.

3.3. Discussion / implications

What emerges from this dataset is a broad spectrum of wet meadow hydrologic conditions with respect to groundwater depth and inundation at the two study sites. The Binfield site has shallower groundwater overall and dispersed zones of especially shallow groundwater (median <0.5m) that follow relic braided river channels. Conversely, the Fox site has deeper groundwater levels overall with concentrated areas of shallow groundwater constrained to excavations that were performed to improve the site hydrology. Areas where the topography wasn't modified record the deepest groundwater levels at the site.

Interestingly, site-wide hydroperiod duration is similar at both sites, and despite being dryer overall, the Fox site records greater inundated area proportions than the Binfield site. However, the distribution and area percentage of each site that records longer duration hydroperiods (>50 days) differs greatly. The Binfield site records a range of hydroperiod durations that vary across the site and follow patterns associated with relic braided channels. Shallower swales have hydroperiod durations that range from 10-100 days, while deeper swales record hydroperiod durations that span 100-365 days. Conversely, Fox records hydroperiod durations of 0 days across much of the site and hydroperiod durations of 10-365 days within excavated areas. Since inundated areas are concentrated around excavations, large portions of the site are far from the nearest inundated area and there are less gradations in hydroperiod duration than are recorded at the Binfield site. Results demonstrate how hydroperiod duration reported at a single swale is uncharacteristic of site-wide conditions, particularly at the natural wet meadow site which maintains the ridge-swale topography associated with relic braided river deposits. The inundated proportion of each site is highly dynamic within and between years.

When compared to other wet meadows in the CPRV, the two Program study sites (Fox and Binfield) fall somewhere in the middle of the recorded range of hydrologic conditions as reported in previous studies (i.e., 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. However, the current analysis shows that Crane Meadows is at the wettest edge along the spectrum of central Platte River wet meadow hydrology.

This analysis highlights uncertainty surrounding hydrology at wet meadow sites in the CPRV. We identify significant spatiotemporal variability within and across sites in metrics that are commonly used to quantify wet meadow hydrology. Wetter sites tend to have spring and summer median groundwater depths that are within a meter of the surface, whereas dryer sites tend to have median groundwater depths greater than one meter. However, variability across wet and dry years shifted median values up to 0.36 m, or by approximately 25%. Hydroperiod duration was similar across sites with vastly different groundwater depths due to a few low-elevation swales or depressions. Hydroperiod also varied significantly across individual sites, with some areas recording 365 days of standing water and others recording zero.

This study does not address causes for the variable hydrology that is identified across CPRV wet meadow study sites. Further work is needed to determine whether differences are natural or related to human activities. The finding of wetter sites to the east could be coincidental, or it could be related to a number of proposed factors. Upstream reaches of the study area along the Platte River have experienced incision due to return flows from hydropower plants that are absent of sediment. A lowering of the river channel would likely impact (i.e., lower) groundwater levels in the adjacent floodplain, which could result in dryer wet meadows. The loss of wet meadows across the CPRV has been attributed to changes in hydrology across the CPRV related to draining and ditching for agriculture and more broadly, water development throughout the region. Particularly restored wet meadow sites may have undergone historic recontouring to support agriculture that modified the topography and in turn groundwater. Conversely, Crane Meadows and Binfield are located within conservation areas that may have less human-impacts to hydrology.

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 sites like Crane Meadows or Binfield can be used as realistic targets for wet meadow restoration elsewhere. When tasked with managing CPRV wet meadow sites, it is important to recognize the spectrum of hydrologic conditions that exist. Realistic hydrologic targets for restoration will likely fall within the limits of conditions reported in this study and will certainly require considerations for site specific constraints related to topography and hydrology.

4. Groundwater-vegetation links

Groundwater and vegetation are tightly linked at wet meadow sites. 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 percentage of time) that groundwater was at or above a particular level. Wesche et al. (1994) recommended 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, or dry ridge) are likely to be present based on groundwater depth statistics for a given location.

Although the utility of the Henszey et al (2004) study was presciently recognized by Wesche et al. in 1994 for management applications and by Weir and Chaves Ramirez (2011) for defining wet meadows, to our knowledge, few if any studies have applied their methodology to manage or evaluate wet meadows in the CPRV. One of the remaining challenges for managing wet meadows 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 a 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 and amount 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 water and 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 the prevalence of wetland vegetation 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 a groundwater depth statistic that was the strongest predictor for the type of vegetation found at wet meadow sites. They grouped vegetation into four categories, emergent, sedge meadow, mesic prairie, or dry ridge. They tested a range of groundwater depth statistics and using the Akaike information criteria and identified the growing-season maximum 7-day moving average groundwater-level to be the strongest predictor. Note, the L7th is calculated by taking a subset of DTGW values from the growing season, calculating a 7-day moving average, and then taking the maximum value from that set. Their results included a series of non-linear models that use the L7th as the dependent variable to predict which of four key wet meadow vegetation groups is likely to be present. The models also assign groundwater depth ranges associated with each vegetation category. Their reported vegetation categories and associated L7th ranges are, 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 (see Figure 11). Positive L7th values indicate maximum growing-season groundwater levels that are above the 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).

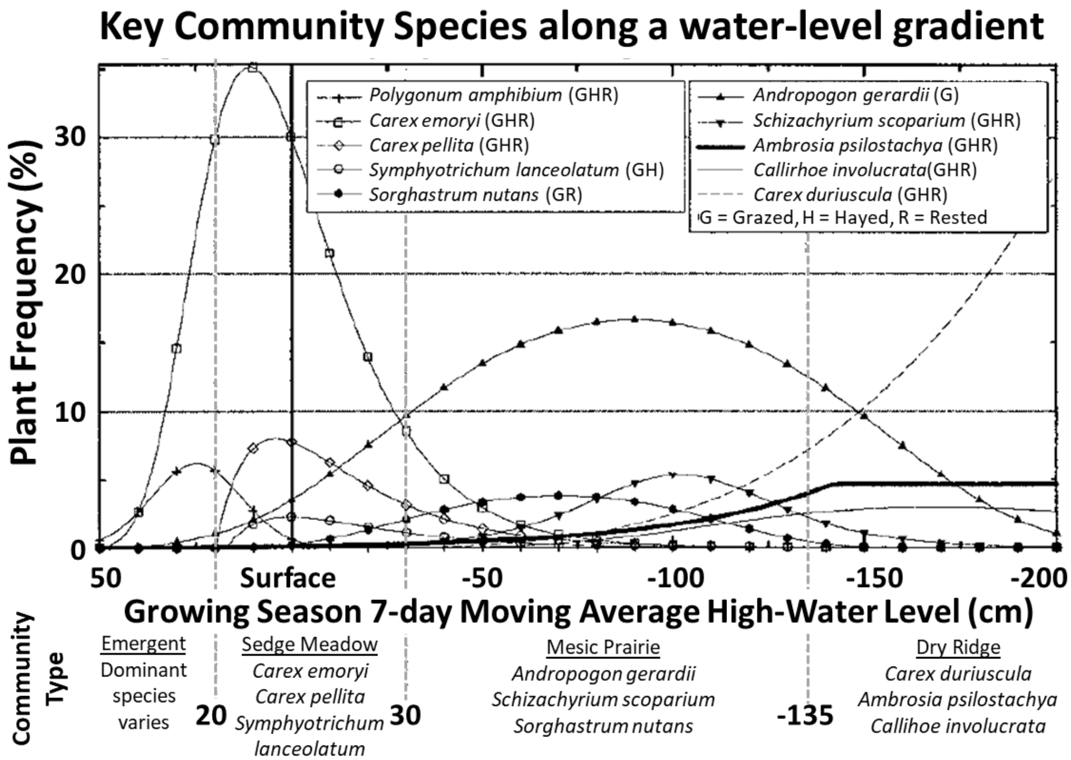


Figure 11 – Nonlinear models and groundwater depth categories linking plant frequency to the growing season 7-day moving average high-water level (L7th) (Henszey et al., 2004).

We calculated the L7ths for a series of groundwater depth surfaces to develop spatially continuous L7th statistics and corresponding 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, the 7-day moving average groundwater elevation was substituted for the mean daily groundwater elevation used during interpolation. This effectively produced a 7-day moving average groundwater elevation surface which was subtracted from the ground surface elevation to convert to 7-day moving average DTGW. Following Henszey et al. (2004), groundwater elevation timeseries were limited to the growing season period (15 March to 15 October), rather than the entire year. For each yearly set of 7-day moving average DTGW surfaces, the maximum DTGW value was calculated for each grid cell. Note, by convention, DTGW is negative for groundwater depths below ground surface. Therefore, the maximum represents the shallowest groundwater depth. The result was a single gridded raster surface for each growing season where each grid cell contains 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.

L7th predicted vegetation groups were compared to landcover databases and vegetation data from grassland surveys to evaluate the accuracy of results. Grassland surveys were conducted by Prairie Legacy Inc. (https://prairielegacyinc.com/) at the Fox site in 2019 and 2022 and at the Binfield site in 2016, 2019, and 2022. Surveys, which typically occurred in late June or July, involved collecting plant cover data within microplots along a series of standardized transects throughout the central Platte River valley. Transects and microplots were consistent across years, though some transects were omitted during 2019 and 2022 due to flooding. Reports indicate that the Fox site was surveyed in 2016, but data were not available in the accessory files. Survey protocols are documented within PRRIP Grassland Vegetation Assessment Reports for associated years (PRRIP 2016, 2019, and 2022). Figure 12 and Figure 13 include transect start and end points for 2016 grassland surveys at the Fox and Binfield sites to provide an example of sample distributions.

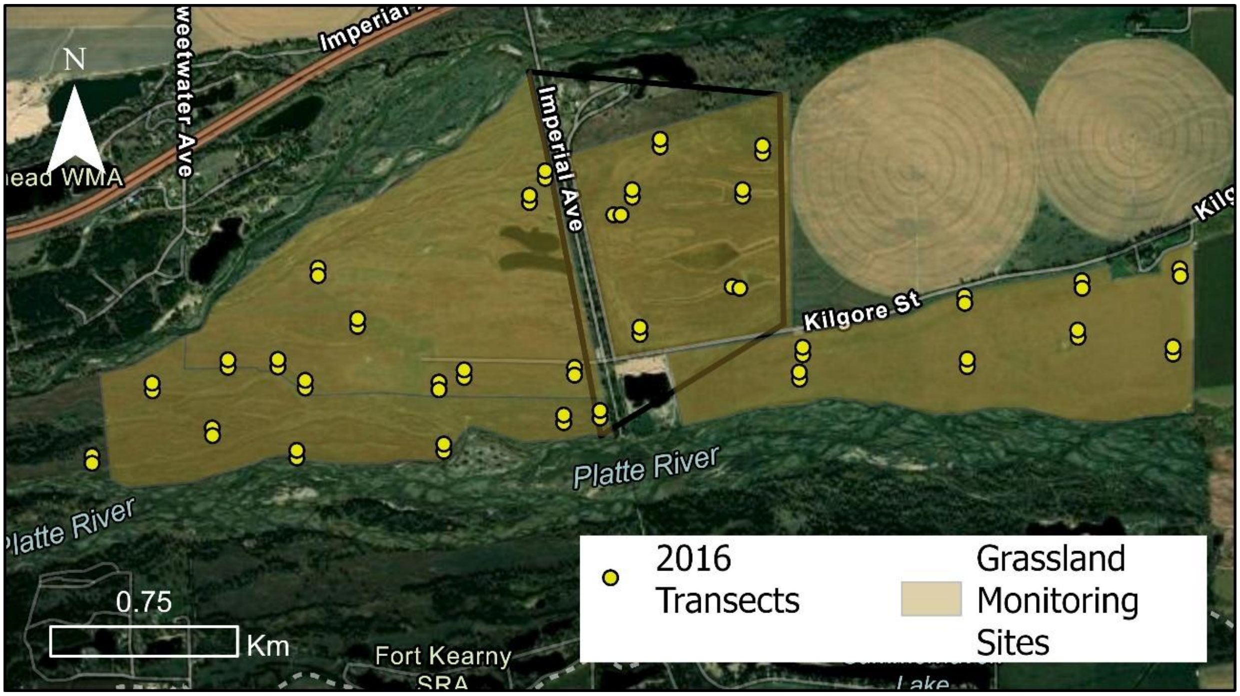


Figure 12 – Grassland vegetation transect points at the Fox site.

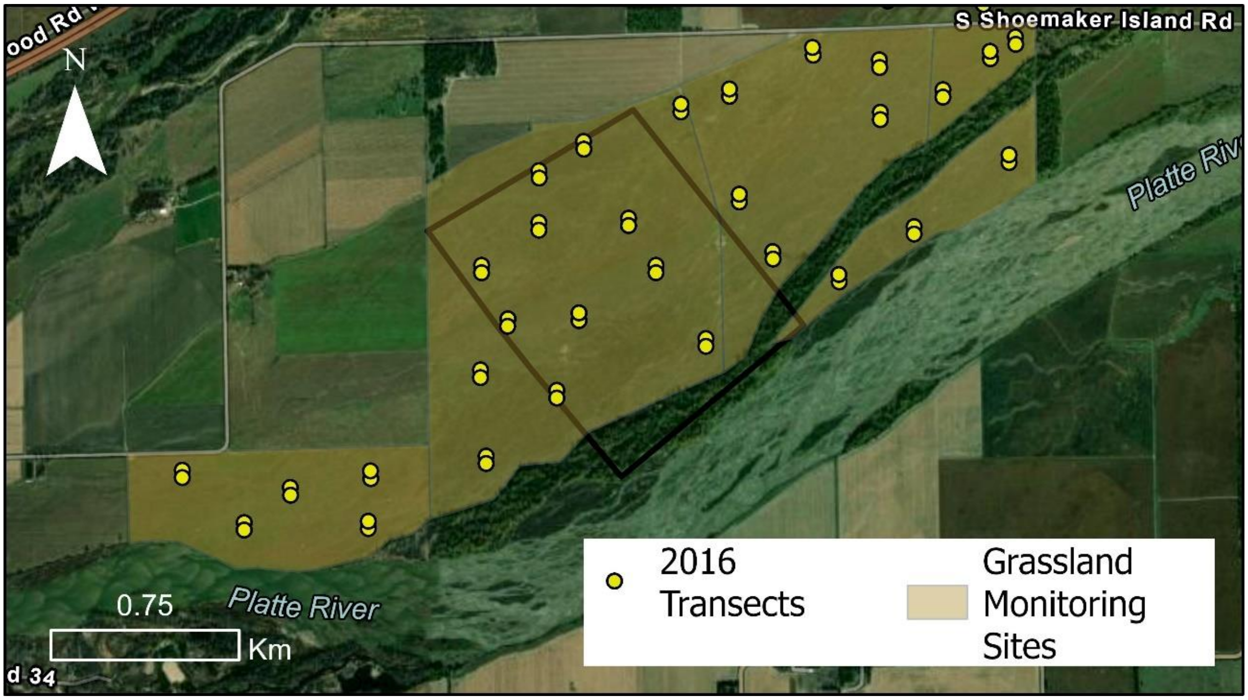


Figure 13 – Grassland vegetation transect points at the Binfield Site.

To enable comparison, L7th and surveyed vegetation proportions were extracted along transects and converted to equivalent metrics. L7th vegetation proportions were assigned to each transect as the category with the greatest proportion along the transect. Most transects spanned a single L7th category, but for transects where more than 30% spanned another vegetation group, both groups were assigned, again using the format 'A/B'. A confusion matrix was generated to compare results for each category.

Surveyed vegetation species were converted to equivalent L7th categories (i.e., emergent, sedge meadow, mesic prairie, and dry ridge) by recording the number of species in each wetland indicator status category (WISC) (see Table 6) and calculating an L7th category proportion along each transect where: OBL and FACW = Emergent; FACW and FAC = Sedge Meadow; FAC and FACU = Mesic Prairie; FACU and UPL = Dry Ridge. Since WISCs overlapped in L7th categories, percentages totaled greater than 100%. The L7th category with the greatest proportion was then assigned to each transect. For sites where the second most prevalent L7th category dominated a significant portion of the transect (>50%), two vegetation groups were assigned using the format 'A/B' (e.g., Emergent/Sedge Meadow). It's worth noting that L7th predictions often spanned multiple vegetation groups for a given transect as well.

Table 6 - Wetland indicator status categories as described in the National List of Plant Species that Occur in Wetlands (Reed 1988); From the National Wetland Plant List Indicator Rating Definitions (USACE, 2012).

Wetland Indicator Status (abbreviation)	% Occurrence in wetlands
Obligate (OBL). Occur almost always under natural conditions in wetlands.	99
Facultative Wetland (FACW). Usually occur in wetlands but occasionally found in non-wetlands.	67-99
Facultative (FAC). Equally likely to occur in wetlands and non-wetlands.	34-66
Facultative Upland (FACU). Usually occur in non-wetlands but occasionally found in wetlands.	1-33
Upland (UPL). Occur in wetlands in another region but occur almost always under natural conditions in non-wetlands in the region specified.	1

4.2. Results

Vegetation categories were predicted using the L7th hydrology statistic. L7th DTGW and corresponding vegetation category are presented as raster surfaces in Figure 14. Results for each site are summarized in Table 7 as the percentage of each vegetation category that occurs at the site. Groundwater depths are reported as the L7th statistic with positive values indicating groundwater above ground surface. 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 recorded considerable area proportions within the dry ridge category (11 to 34%) during all years.

In comparing the two sites, Binfield exhibited 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 results 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 suggests that recontouring efforts improved conditions at the Fox site. However, it is worth noting that despite a greater percentage of emergent vegetation, overall, the Fox site overall has significantly less sedge meadow than Binfield, and a majority of landcover in the mesic prairie and dry ridge groups. This suggests that larger scale recontouring efforts would be needed to significantly alter vegetation classes and associated hydrology at the site. Further, recontouring efforts would likely need to address the transitional zones and distributed high- and low- areas to mimic natural wet meadow vegetation assemblages.

During most study years (all but 2013), at least 50% of the Binfield site recorded groundwater depths that could support wetland (i.e., sedge meadow and emergent) vegetation (Table 7). Conversely, during all study years, only 9.4% to 42% of the Fox site records groundwater depths that would support wetland vegetation. Results suggest that wetland areas at Fox are, during most years, limited to <20% of the site, even with excavations and pumping to promote wetter conditions.

Table 7 – Percentage of total site area corresponding to L7th depth and vegetation category (D=dry ridge, M=mesic prairie, S=sedge meadow, E=emergent) by growing season.

	Binfield				Fox			
Veg. Group	D	M	S	E	D	M	S	E
L7th (m)	(-5 to -1.35)	(-1.35 to -0.3)	(-0.3 to 0.2)	(0.2 to 2)	(-5 to -1.35)	(-1.35 to -0.3)	(-0.3 to 0.2)	(0.2 to 2)
2013	<0.1	52	47	1.1	32	58	7.0	2.4
2014	<0.1	41	59	0.31	29	60	6.8	3.4
2015	<0.1	8.4	81	11	11	46	28	14
2016	<0.1	25	74	1.0	17	61	11	9.7
2017	<0.1	26	73	0.71	25	62	6.6	5.8
2018	<0.1	39	60	0.25	34	55	6.6	3.3
2019	<0.1	14	81	4.4	17	62	11	9.3
2020[1]	<0.1	31	68	0.53	18	63	9.7	9.0
AVERAGE	<0.1	29.6	68.0	2.4	22.8	58.3	10.9	7.1

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

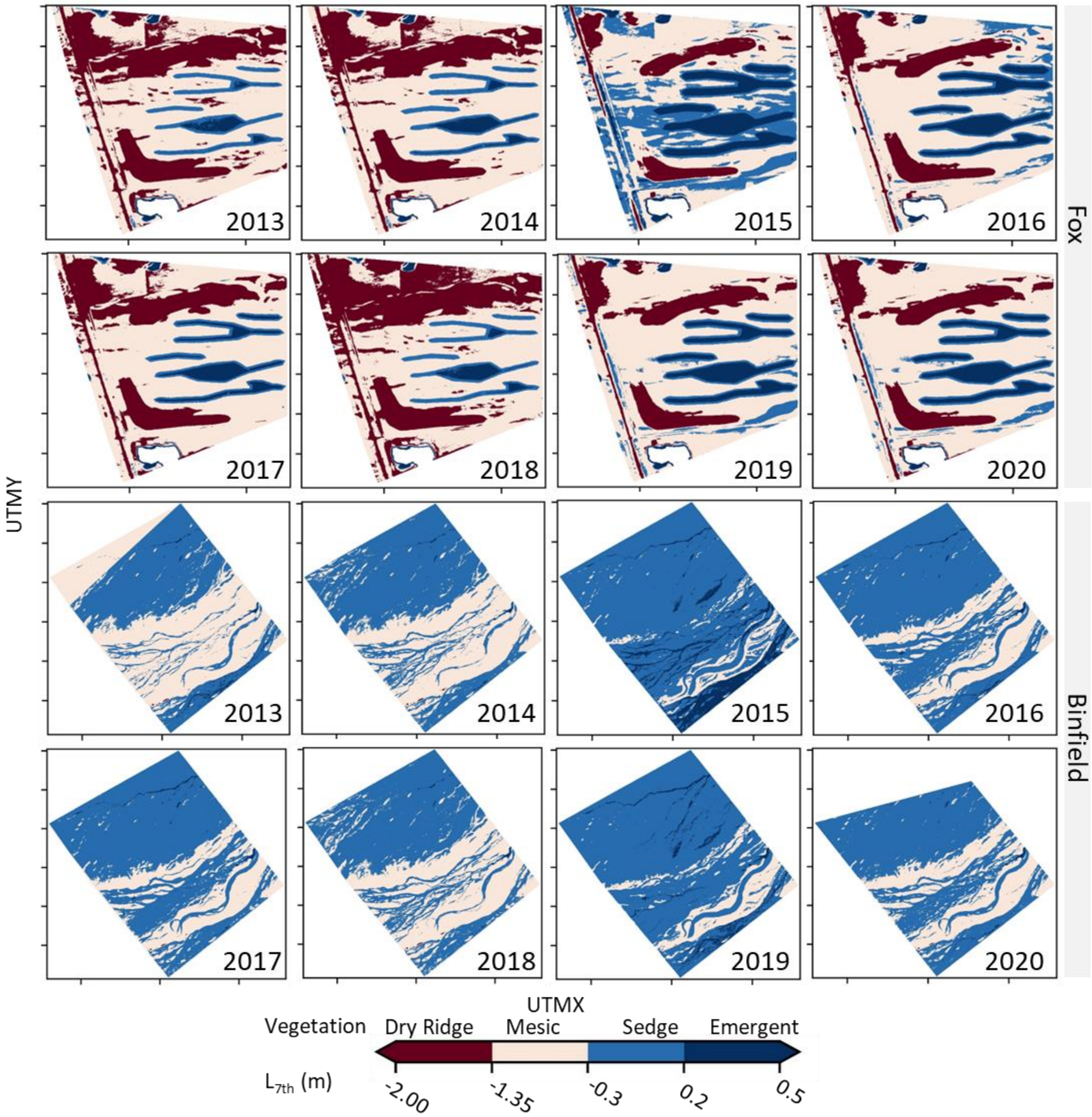


Figure 14 – L7th depth to water surfaces for Fox (top) and Binfield (bottom) sites. Color indices correspond to Henszey et al. (2004) vegetation categories. Well 216 DTGW data was unavailable for 2020, hence truncated surface.

Vegetative composition measured during grassland surveys was in agreement with L7th predicted vegetation category. Tables of extracted L7th and grassland survey vegetation groups are provided in Appendix E. A confusion matrix was created to compare L7th predicted vegetation groups with grassland survey vegetation groups (Table 8). A confusion matrix represents the prediction summary in matrix form. It shows how many predictions are correct and incorrect per class and helps in understanding the classes that are being confused by model as another class. Within the confusion matrix, diagonal components sum the number of correctly predicted instances. Off-diagonal entries total incorrectly predicted instances, with the column category indicated what it was incorrectly predicted as. The right-most column calculates the accuracy as 100 x # correct / total number of samples in category.

Prediction accuracy ranged from 0% to 100% across vegetation groups. However, low-accuracy occurrences (<50%) were associated with co-dominating groups (e.g., EM/SM) in which the predicted L7th value was accurate for at least one of the dominant groups. For example, the EM/SM category was incorrectly predicted as SM by the L7th method. There was one occurrence where the grassland survey indicated mesic prairie conditions and the L7th method predicted dry ridge. None of the sites, grassland survey or L7th, were within the emergent category, so method accuracy for that group is not available. Results for comparisons to landcover classes are presented and discussed in Section 7.

Table 8 - Confusion matrix for L7th predicted and grassland survey vegetation groups for Fox and Binfield study sites.

		L7th Predicted							
	Group	EM	EM/SM	SM	SM/MP	MP	MP/DR	DR	Accuracy
Grassland Survey	EM	0	0	0	0	0	0	0	-
	EM/SM	0	0	3	0	0	0	0	0%
	SM	0	0	7	1	0	0	0	88%
	SM/MP	0	0	0	11	1	0	0	92%
	MP	0	0	0	0	10	0	1	91%
	MP/DR	0	0	0	0	1	1	1	33%
	DR	0	0	0	0	0	0	2	100%

4.3. Discussion

Significant differences in hydrology and predicted vegetation were recorded within and between two wet meadow study sites. The 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.

The presence of seasonally inundated wetlands is a key component of CPRV wet meadows, regardless of the definition. Achieving wetland conditions within some proportion of wet meadow sites is therefore a reasonable objective for restoration. Herein, we demonstrate how the method described herein can be applied to generate a hydrologic target at wet meadow sites. We set an arbitrary vegetation target of 50% cover of hydric vegetation (i.e., emergent and sedge meadow vegetation groups) and calculate the change in hydrology (ΔL7th) that would be required to theoretically support hydric vegetation across 50% of the site. Note, the purpose was not to delineate wetlands for alternative applications. For all years but 2013, the Binfield site met 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 did not meet criteria for wetland designation, with site area percentages ranging from 9.4% to 42%.

The required change in groundwater level (ΔL7th) that would be necessary to achieve at the vegetation target of 50% wetland coverage at the Fox Site for years 2013-2020 are presented in Table 9. Note, the ΔL7th describes a change in the 7-day moving average groundwater level during the highest groundwater level during the growing season. Translating to an objective for managing groundwater levels could include something similar to a 7-day sustained increase in groundwater level equal to the magnitude of the L7th. Changes in L7th ranged 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 the peak sustained (~7 days) groundwater levels would need to rise in order to support wetland vegetation across 50% of the site.

Table 9 - 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 that resulted in the excavation of five elongate depressions on the site. However, herein we demonstrate how this recontouring effort produced ponds that are inundated year-round and are associated with limited areas along pond margins that support key wet meadow emergent and sedge meadow vegetation. The majority of the Fox site remains as mesic prairie and dryland with proportionally less emergent and sedge meadow vegetation than the Binfield site. To successfully improve hydrology through recontouring would require substantial excavation across the site. Further, excavations would need to incorporate gradational elevation that supports vegetation which are found across moisture gradients described by Henszey et al. (2004).

An alternative approach could 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.

Although L7th comparisons to field vegetation data indicate generally good agreement, future applications of this method would benefit from field data collection efforts that were specifically designed to evaluate method accuracy. As indicated in the confusion matrix, the sample size for emergent and dry ridge vegetation groups was limited to non-existent. Stratified sampling across vegetation groups, and increased sample sizes would improve method validation.

5. Modeling

A shallow, fluctuating groundwater table is necessary for supporting sustained periods of inundation (Brinley-Buckley et al., 2021), wetland vegetation, and habitat associated with diverse species at wet meadow sites (Henszey et al., 2004; Davis et al., 2006). Previous studies have showed that river stage exerts the predominant influence on wet meadow groundwater levels in the CPRV, followed by precipitation, and to a lesser degree, evapotranspiration (ET) (Whiles and Goldowitz, 1993; Wu, 2003; Wesche et al., 1994; Chen, 2007).

Hydrologic models can be used to predict how groundwater levels will respond to a given hydrologic stress, and therefore may be used as tools to guide and test management actions at wet meadow sites. In this section, we describe an expanded version of the Glover bank storage equation that can be calibrated to match field data, and then used to predict how changes in Platte River stage and surface additions of water would modify groundwater levels at wet meadow sites.

Previously, studies have used models to evaluate hydrology at wet meadows. Chen (2007) utilized a three-dimensional numerical model to evaluate river-aquifer-vegetation hydraulic connections in the CPRV (approximately 5 miles west of the Fox site). They identified river stage as having a predominant influence on groundwater levels, and described the river-groundwater system as highly connected with rapid response in groundwater levels to fluctuations in stage. Notably, the study also suggested potential for hydraulic lift at wet meadow sites caused by vegetative water use. However, the ET rates they predicted as capable of generating lift (1.5 cm/day) were substantially greater than typical rates estimated for CPRV wet meadow sites (<0.4 cm/day) (USGS, 2013). Loheide II and Gorelick (2007) developed a finite element model of variably saturated groundwater flow to assess hydroecologic functions of a wet meadow system 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 based on Glover's bank storage equation to test how changes in stage affect groundwater levels at wet meadow sites (PRRIP, 2012). However, the model lacked a method of accounting for precipitation and ET, and therefore could not be calibrated to match observations at wet meadow sites, limiting the confidence in results.

Here, we expand the PRRIP (2012) analytical model to include added functions to account for effects of precipitation and ET on groundwater levels. 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. Effectively reproducing groundwater flow dynamics would require a 3D groundwater flow model; requiring significant site-specific data collection and development and limiting utility at other sites where this information is more limited. This analytical model can be readily applied to different sites with relatively minimal parameter inputs and data inputs that are often publicly available (i.e., through the High Plains Regional Climate Center at https://hprcc.unl.edu/).

The resulting model provides a useful tool for testing water management scenarios, with improved predictions as compared to previous models, indicated by 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 15). 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, in this case, 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).

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

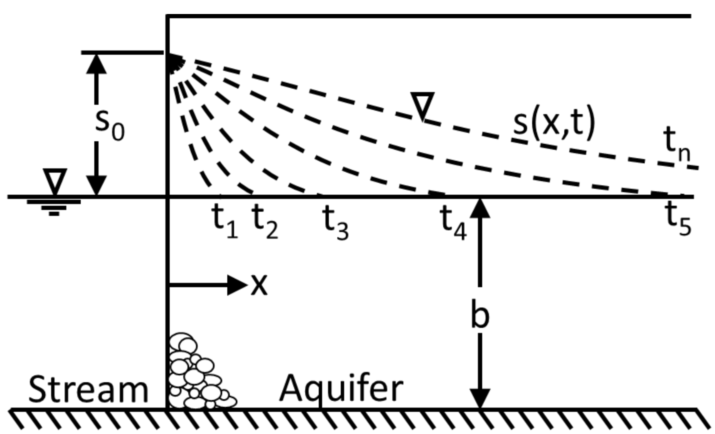


Figure 15 - Schematic illustrating variables in the Glover Bank Storage equation (Sanders, 2001).

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 (i.e., does not runoff or evaporate). This fraction is referred to as the recharge or precipitation infiltration coefficient (f). Groundwater level changes due to precipitation were calculated by multiplying a recharge coefficient (f) by the daily total precipitation as recorded at the on-site weather station and then 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). Specific yield was estimated through manual model calibration by varying the specific yield within a range of expected values (0.08 to 1.5) until the observed and predicted change in groundwater level matched. Calibration periods were preferentially selected for the period immediately following precipitation events which generated short-term spikes in groundwater levels. This method is appropriate for sites with shallow water tables and high soil water content such as riparian wet meadows (Healey and Cook, 2002). Since the change in water table also depends on the fraction of recharge that reaches the water table, calibration also involved testing a range of recharge coefficients. Recharge coefficients (f) typically range from 0.1 to 0.6 (Andualem 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:

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

To test the model at wells, calibration was performed for each well location to achieve a best-fit between modeled and observed groundwater levels for each growing season during the study period. Calibration involved varying hydraulic parameters (K, Sy, f, ε and λ) between a plausible range of values until a visual best-fit was achieved. Notably, the risk for over-calibration is plausible since field estimates of hydraulic parameters were not available. However, since parameters were allowed to vary within the expected range for braided river and deposits and values reported in the area (e.g., Hurr, 1983; Chen, 2004; Song and Chen, 2010; Yue et al., 2016), the method should provide reasonable results. For ET parameters (ε and λ; Eq. 3), calibration was implemented in Python using the SciPy optimization function curve-fit with the trust region method. This method applies a 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

Results demonstrate an excellent overall fit between modeled and observed groundwater levels. Trends are accurately captured and in sync with field observations. Figure 16 and Figure 17 present model results for a representative well at each study site (Well 112 at Fox, and Well 212 at Binfield) during 2014 through 2020 growing seasons (note, insufficient data from 2020 were available for well 112). These wells were selected to show typical model performance at each site and to highlight. Results for all well locations and years are provided in Appendix F.

Observed versus predicted groundwater levels are plotted for each site, with R^2 values of 0.95 and 0.92 for Fox and Binfield respectively (Figure 18). The model is less accurate when predicting groundwater levels during high-groundwater table events. Figures show that short-term peaks are often underpredicted, suggesting that the model may under-predict future scenarios where an even higher groundwater table is expected. Groundwater levels and river stage follow similar trends, with peaks and troughs in groundwater level lagging 1-2 weeks behind corresponding peaks and troughs in stage. The model predicts this relationship reasonably well.

The 2017 plot for the Fox Site reveals significant variations in groundwater level during summer months which are not predicted by the model. During this time, the Program experimented with applying pumped well water to the site from on-site wells (Brei, personal communication or PRRIP, unpublished data). Though pumped and applied water volume records were not compiled for this study, they likely explain differences between modeled and observed values. However, pumping is not one of the stresses included in Equation 4, therefore, another method would be needed to predict groundwater heads influenced by pumping (i.e., Theis or Neuman equation or a numerical groundwater model).

All model calibration values are within the expected range for braided river deposits and values reported in studies conducted nearby (e.g., Hurr, 1983; Chen, 2004; Song and Chen, 2010; Yue et al., 2016) (Table 10). 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.

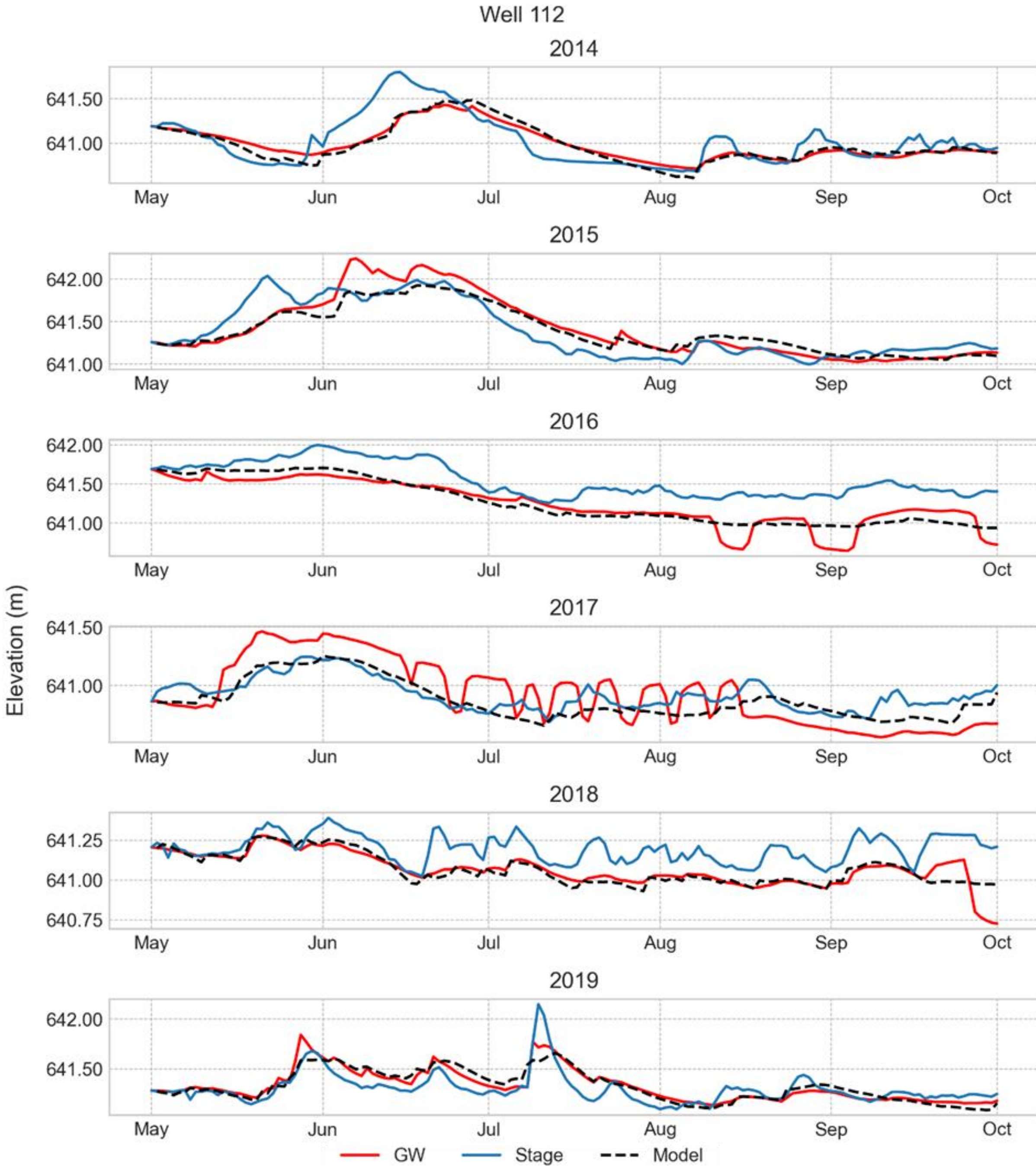


Figure 16 - Well groundwater levels, stage, and modeled groundwater levels for a representative well at the Fox site (well 112).

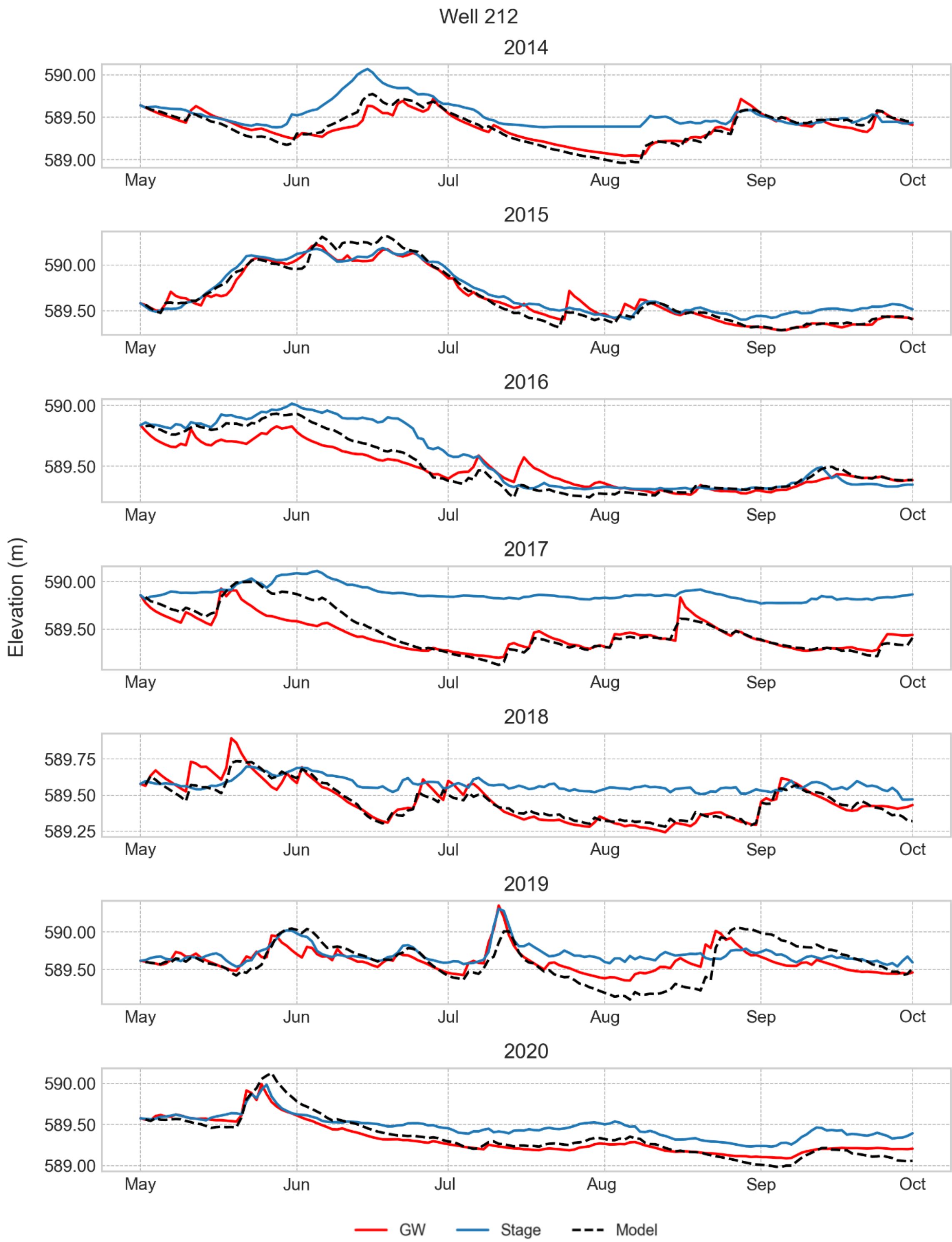
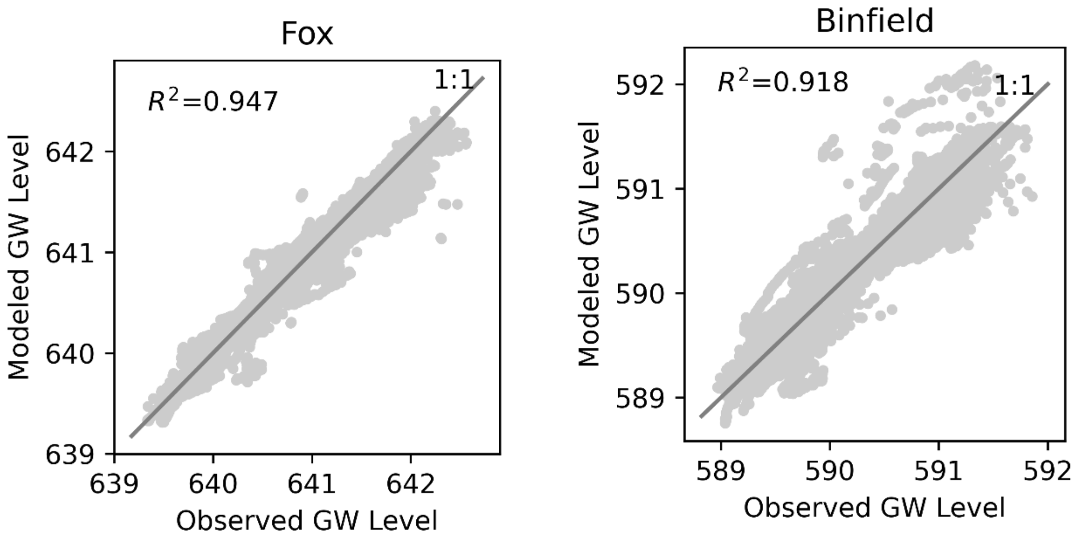


Figure 17 - Well groundwater levels, stage, and modeled groundwater levels for a representative well at the Binfield site (well 212).



143

144 *Figure 18 - Observed versus modeled groundwater levels (m) with associated R^2 for all wells at Binfield*
145 *and Fox sites.*

Table 10 - Fitted parameters from model calibration for groundwater observation wells at the Fox site (wells 101-116) and Binfield site (wells 201-216).

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

5.3. Discussion

As described previously, groundwater depths at the Fox site were found to be too deep over the majority of the site to support key wetland vegetation groups as estimated using the L7th DTGW statistic. Groundwater level increases of up to 0.93 m, and on average 0.59 m, were estimated as necessary to achieve a hypothetical management target of 50% spatial coverage with wetland vegetation during 2018. To demonstrate the utility of this model, we use it to estimate the groundwater level changes that could be produced with two plausible management actions at the Fox site.

Groundwater levels were especially low during 2018 and so the influence of management actions on hydrology is demonstrated retroactively during that year. For the first management scenario, we test how an increase in river stage (and therefore discharge) could be applied to increase groundwater levels at the Fox site by 0.59 m. In the second scenario, we test how surface water could be applied at the site, as modeled through the precipitation term in Equation 4, to similarly achieve the target change in groundwater level.

For the stage-increase scenario, we consider a two-week increased stage event that could represent a hypothetical release of water along the Platte River for this purpose. To simulate the two-week increase in river stage, a gaussian shaped stage curve was input into Equation 4 as a stepwise boundary stress along with calibrated parameters for Wells 101 through 116. The magnitude of the stage increase varied until the change in groundwater level at the well location matched the target increase of 0.59 m. The required magnitude of stage increase ranged from 0.6 to 1.52 across wells with changes due to variable hydraulic parameters and distance from the river. As expected, wells further from the channel required a greater magnitude of stage increase to raise groundwater levels.

An example of the resulting change in groundwater level caused by a stage-increase event is demonstrated for the 2018 growing season in Figure 19. The 2018 growing season recorded especially low groundwater levels at the Fox site. Well 116 is selected as an example location because it's relatively distant from the channel. In the example, the required increase in stage is 1.52 m to raise the groundwater level at well 116 by 0.59 m. During June of 2018, Platte River discharge ranged from just under 5.7 cubic meters per second (cms) (200 cfs) to 70 cms (2,472 cfs). Based on stage-discharge curve reported for the USGS Kearney gage (USGS #: 06770200), a 1.5- meter stage increase would correspond to an increase in discharge of over 566 cms (20,000 cfs) (Figure 2). For reference, the regulatory flood stage at this gage station, eight kilometers upstream of the Fox Site, is 2.1 feet, corresponding to a discharge of 507 m^3 s-1 (17,904 cfs). Therefore, the example scenario demonstrates how stage increases may not be a viable method for increasing groundwater levels to target levels, particularly along inland portions of the site. For areas closer to the channel where lower magnitude stage changes are required to reach groundwater targets (e.g., 0.6m), stage increases may be viable.

Similarly, we calculated the required application of surface water as the daily rate (L/T) that would be required to increase groundwater depths by the target value of 0.59 m. We simulate a 10-day surface recharge event by adding a constant rate value to the precipitation term in Equation 4. We varied the value until the target groundwater elevation change was achieved. The required rate of surface water applied to the site ranged from 0.03 to .055 m day-1 (approximately 1 to 2 in/day). Differences in rate were due purely to variable hydraulic parameters and were not directly affected by distance from the river. Locations with lower values for storage (i.e., specific yield, Sy) required lower application rates and, assuming calibrated parameters are representative of site conditions, may indicate finer grained parts of the site. The Fox site is approximately 247 acres (1.07 km^2), assuming 0.05 cm per day were applied across the site, the delivery rate would need to be 42.8 liters per minute (LPM) or 0.05 acre-ft per day. The 10-day precipitation event is similarly demonstrated for the location at well 116 (Figure 19). Notably, the model predicts an unrealistic retention of water following the surface water addition, as the model does not account for draining. 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 groundwater level for a short duration, predictions may be adequate.

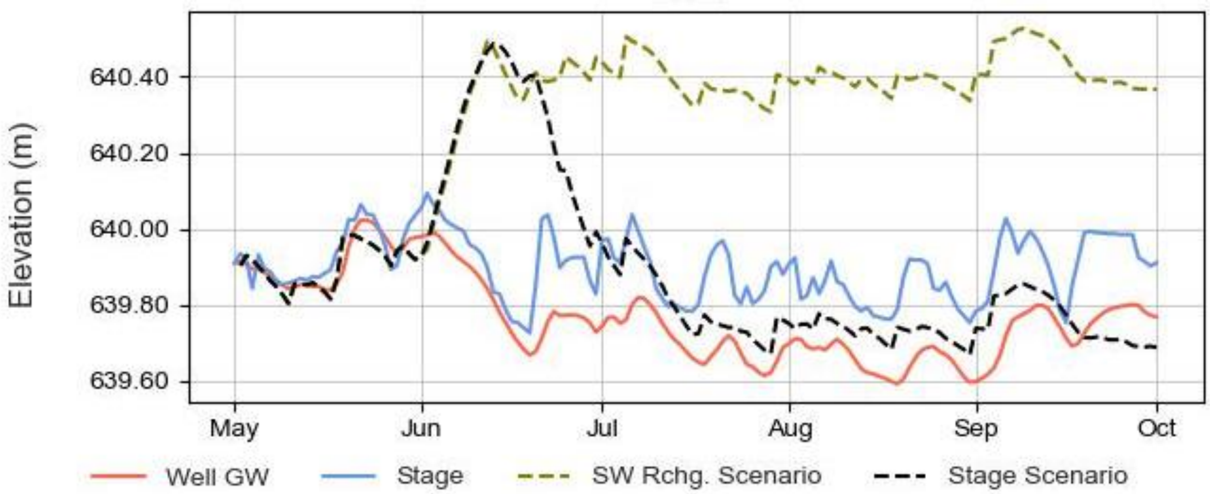


Figure 19– Plot demonstrating how two management scenarios, surface water recharge and stage increase, influence predicted groundwater levels at the selected example location of well 116.

6. River-floodplain elevation analysis

One goal for this study was to generate results that could inform management at other CPRV wet meadow sites. In Sections 1-5, we demonstrate and discuss the influence of river stage and topography on groundwater levels. As discussed throughout this report, the river stage is a primary driver for floodplain groundwater levels, and shallow, fluctuating groundwater tables are critical for sustaining wet meadows. However, the relationship between the absolute elevation of the river and ground surfaces was not explicitly addressed. Previous studies have 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 areas where the river surface elevation is similar to the adjacent floodplain to have relatively shallow groundwater tables fed by the river. Conversely, areas where the river is lower than the adjacent topography might be expected to have deeper groundwater levels in the adjacent floodplain. Similarly, for reaches where the river has incised relative to the adjacent floodplain, one might expect adjacent groundwater levels to also decrease. This relationship can be thought of as the potential for the river to supply water to the adjacent floodplain.

If clear relationships between absolute river elevation and wet meadow ground surface elevations can be quantified for archetypal wet meadow sites, those relationships could be used to predict other locations that might have suitable hydrology for wet meadows. Limited groundwater elevation data exists for the area immediately surrounding the river channel, where wet meadows tend to occur in the CPRV. Methods which enable predictions about groundwater levels without requiring the installation of wells and other expensive equipment would therefore provide value. Further, this could support wet meadow site management and identification of future sites for restoration viability.

Herein, we generate differenced maps over river surface and ground surface elevations to make predictions about which areas are more likely to have shallow groundwater (i.e., <1m of the surface). We then identify whether established wet meadow sites have distinguishable river-ground surface relationships that can be used to inform site management or restoration potential. By comparing predictions to groundwater data measured at field sites, we demonstrate the utility of this method which does not require field data collection, but rather uses readily available, high-spatial coverage datasets to make predictions about wet meadow hydrology in areas where groundwater data are sparse or unavailable. This constitutes an extension of the hydrological characterizations performed previously within this report.

6.1. Methods

Differenced maps of river and ground surface elevation were created to identify characteristic relationships between river stage and ground surface elevation at known and potential wet meadow sites. River stage elevation (water surface elevation) was generated for a continuous gridded raster using steady-state SRH2D models run from dry conditions for a range of relevant discharges (750-3500 cfs) which span targets designated by the USFWS as having potential benefits for wet meadows (Table 11). The modeled river stage was then extrapolated laterally into the floodplain which involved generating regular transects oriented perpendicular to the river centerline with 0.5 km spacing and a lateral span of 5.5 km into the floodplain. Each transect was assigned the mean value of gridded river surface elevations that intersected the transect, effectively resulting in a set of river surface elevation contours that extend laterally beyond the river. Contours were converted to an elevation surface using the ArcGIS 'Topo to Raster' tool which uses an iterative finite difference interpolation technique to optimize computational efficiency of local interpolation methods, such as inverse distance weighted (IDW) interpolation, without losing the surface continuity of global interpolation methods, such as Kriging and Spline. The USGS National Map 3DEP 1-m DEM ground elevation surface was then subtracted from the extrapolated water elevation surface to produce a gridded raster representing the elevation difference between the water (river) surface and ground surface elevation, referred to as the differenced elevation surface (DEV). DEV is positive where the ground surface is below the river surface and negative where the ground surface is above the river surface (e.g., Figure 20).

Table 11 - 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).

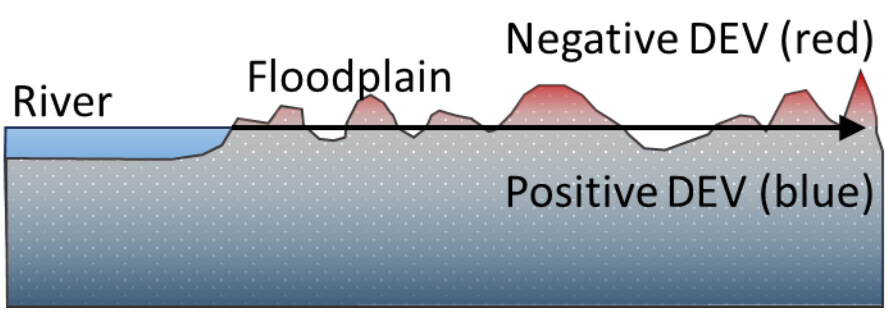
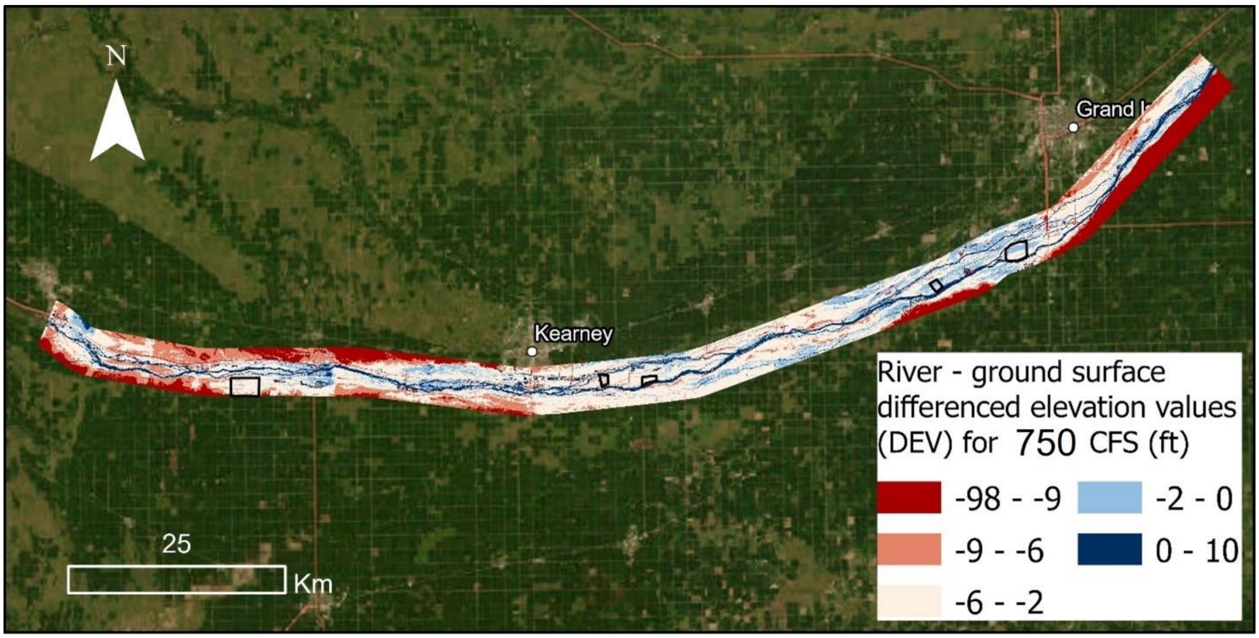


Figure 20 - Schematic illustrating river and ground elevation relationship for a cross section perpendicular to the central Platte River and floodplain with resulting values of the differenced elevation surface (DEV).

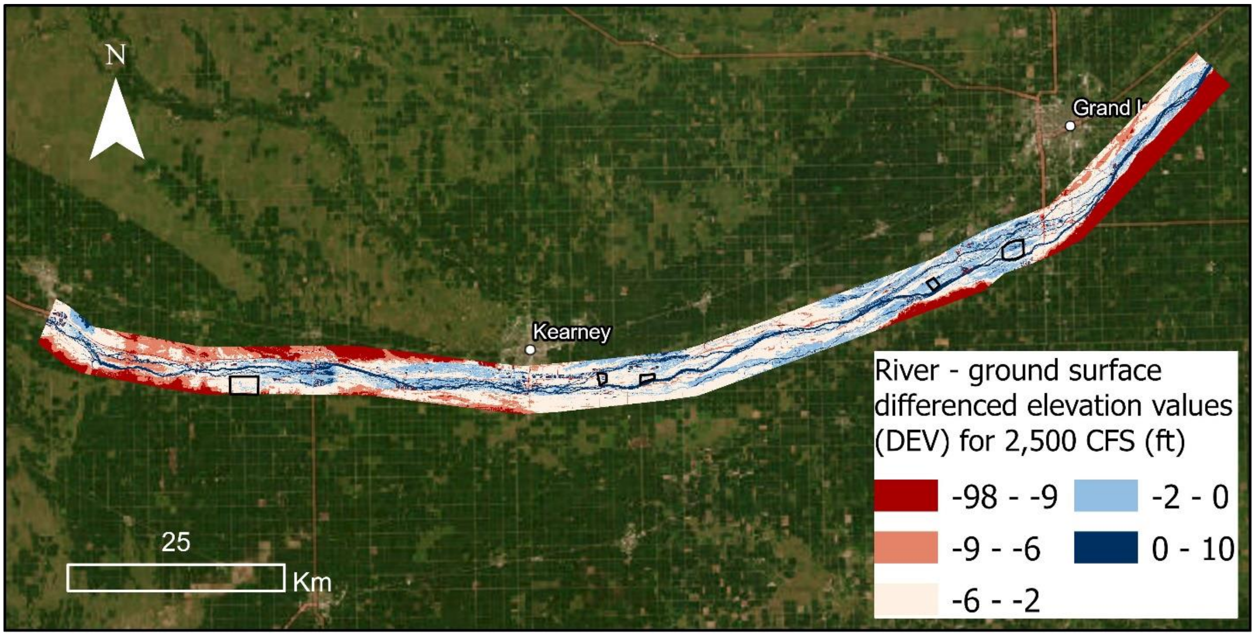
6.2. Results and Discussion

Results from the elevation differencing analysis are presented in Figure 21 through Figure 23 for modeled river discharges of 750 cfs, 2,500 cfs, and 3,500 cfs. Differenced elevation values (DEV) range from -90 ft to 10 ft, with positive values indicating areas where the modeled river surface is above the ground surface and negative values indicating the river surface is below the ground surface. The active channel contains values close to or greater than zero, and in theory should equal zero. Color ranges presented in DEV figures were selected to generally match patterns in L7th results, as discussed subsequently.



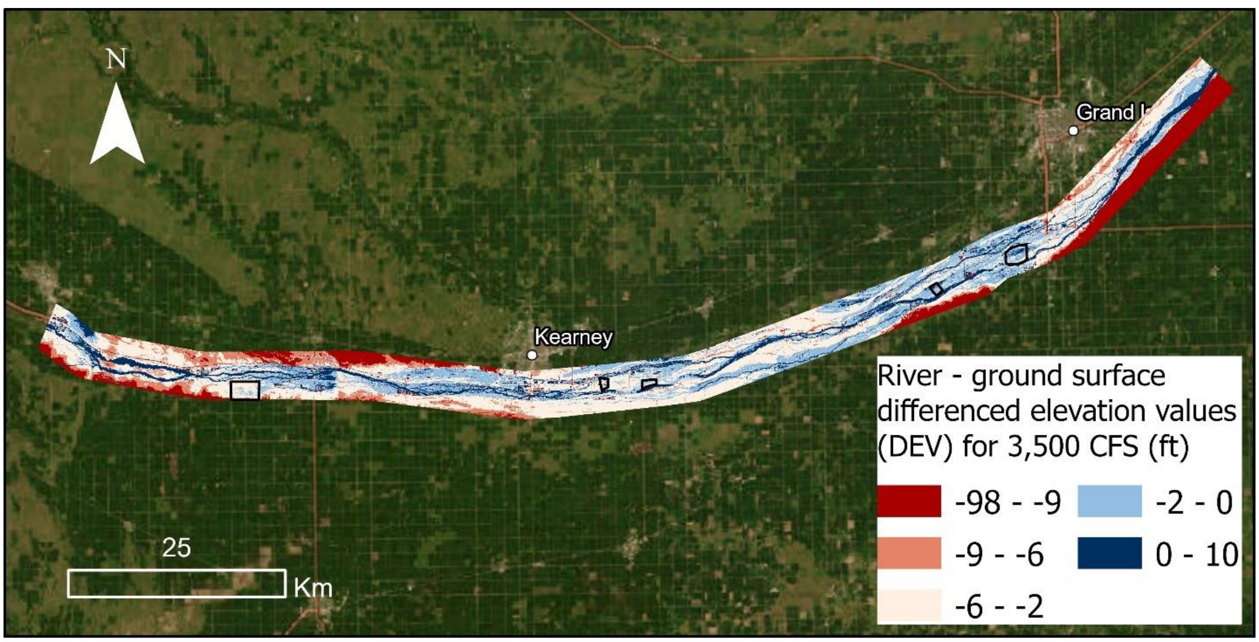
Earthstar Geographics, Nebraska Game & Parks Commission, Esri, HERE, Garmin, SafeGraph, FAO, METI/NASA, USGS, EPA, NPS

Figure 21 – Differenced river stage and ground surface elevation raster for discharge of 750 cfs. Black squares indicate current study sites and those included in Wesche et al. 1994.



Earthstar Geographics, Nebraska Game & Parks Commission, Esri, HERE, Garmin, SafeGraph, FAO, METI/NASA, USGS, EPA, NPS

Figure 22 – Differenced river stage and ground surface elevation raster for discharge of 2,500 cfs. Black squares indicate current study sites and those included in Wesch et al. 1994.



Earthstar Geographics, Nebraska Game & Parks Commission, Esri, HERE, Garmin, SafeGraph, FAO, METI/NASA, USGS, EPA, NPS

Figure 23 – Differenced river stage and ground surface elevation raster for discharge of 3,500 cfs. Black squares indicate current study sites and those included in Wesch et al. 1994.

Visual observations from Figure 21 through Figure 23 reveal important patterns regarding river and ground surface relationships. Off-channel zones within 0.5 km of the channel tend to be characterized by DEVs between -1 and 1 m, indicating the ground surface is within 1 m of the modeled river surface, even at the lowest evaluated discharge of 750 cfs. Areas further from the channel (>0.5m) and inland parts of islands tend to have lower, more negative DEVs, indicating the ground surface is higher than the active river surface. Along some reaches, a sharp boundary between light blue and pink and pink and red exists. These zones represent topographic rises associated with geologic transitions from modern Holocene floodplain deposits to earlier Holocene floodplain surfaces that are now 2 to 3 m above the modern floodplain (Hansen et al., 2014). Off-channel zones where north and south channels converge also tend to be closer in elevation to the river surface. Slightly negative to positive DEVs (>-2 ft) almost always occur in patterns that mimic braided river deposits, indicating these areas are largely unmodified and retain natural topography. The near channel DEV tends to increase with distance downstream, and the region upstream of Overton tends to have more negative DEVs. This could be related to riverbed incision that has occurred along the upstream reach. As expected, the overall DEV becomes increasingly positive (less negative) with increased modeled discharge. In other words, the 3,500 cfs DEV surface shows a greater extent of positive and slightly negative DEVs compared to the 750 cfs DEV, indicating more of the adjacent floodplain is below or near the river surface at greater flows.

The primary goal of this analysis was to identify key elevation relationships between the river and ground surface elevation at wet meadow sites. DEVs for the Elm Creek, Fox, Rowe Sanctuary, Binfield, and Mormon Island wet meadow sites are presented in Figure 24 through Figure 26. These figures demonstrate DEV values for established wet meadow sites, with ranges from < -3 to 1 m and the majority of site areas having values in the -2 and 0 m range.

While Figures 24-26 reveal a broad range of DEVs, some key relationships can be identified for wet meadow sites. For the 3,500 cfs flow, all sites have at least some areas characterized by DEV > 0 ft, indicating that the modeled 3,500 cfs river flow is above the ground surface for at least part of each wet meadow site. Similar to the observation for reach wide DEVs, portions of wet meadow sites with DEV > -1 m are almost exclusively present in relic channel patterns, indicating established wet meadow sites are in areas with minimal anthropogenic reworking of braided river deposits. As discussed in Section 3, groundwater hydrology datasets indicate that select wet meadow sites become increasingly wetter in the downstream direction. DEVs reveal a similar trend, with increasingly positive values in the downstream direction. Elm Creek has the greatest extent of negative DEVs, and Rowe Sanctuary has the greatest extent of positive DEVs, especially at higher flow.

The utility of the DEV method can also be seen by considering 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.

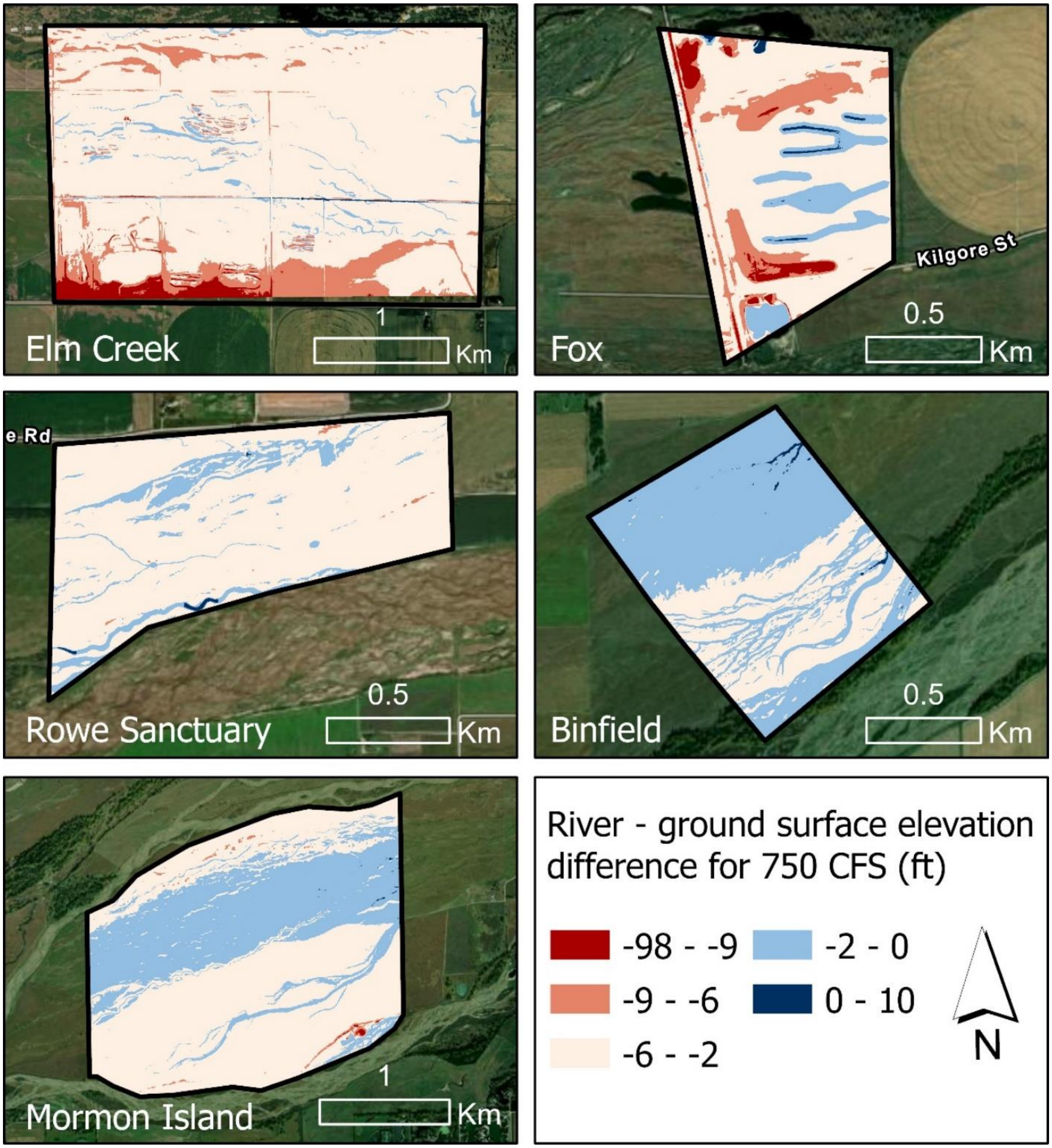


Figure 24 - Differenced river and ground surface elevation rasters for Elm Creek, Fox, Rowe Sanctuary, Binfield, and Crane Meadows with modeled discharge of 750 cfs.

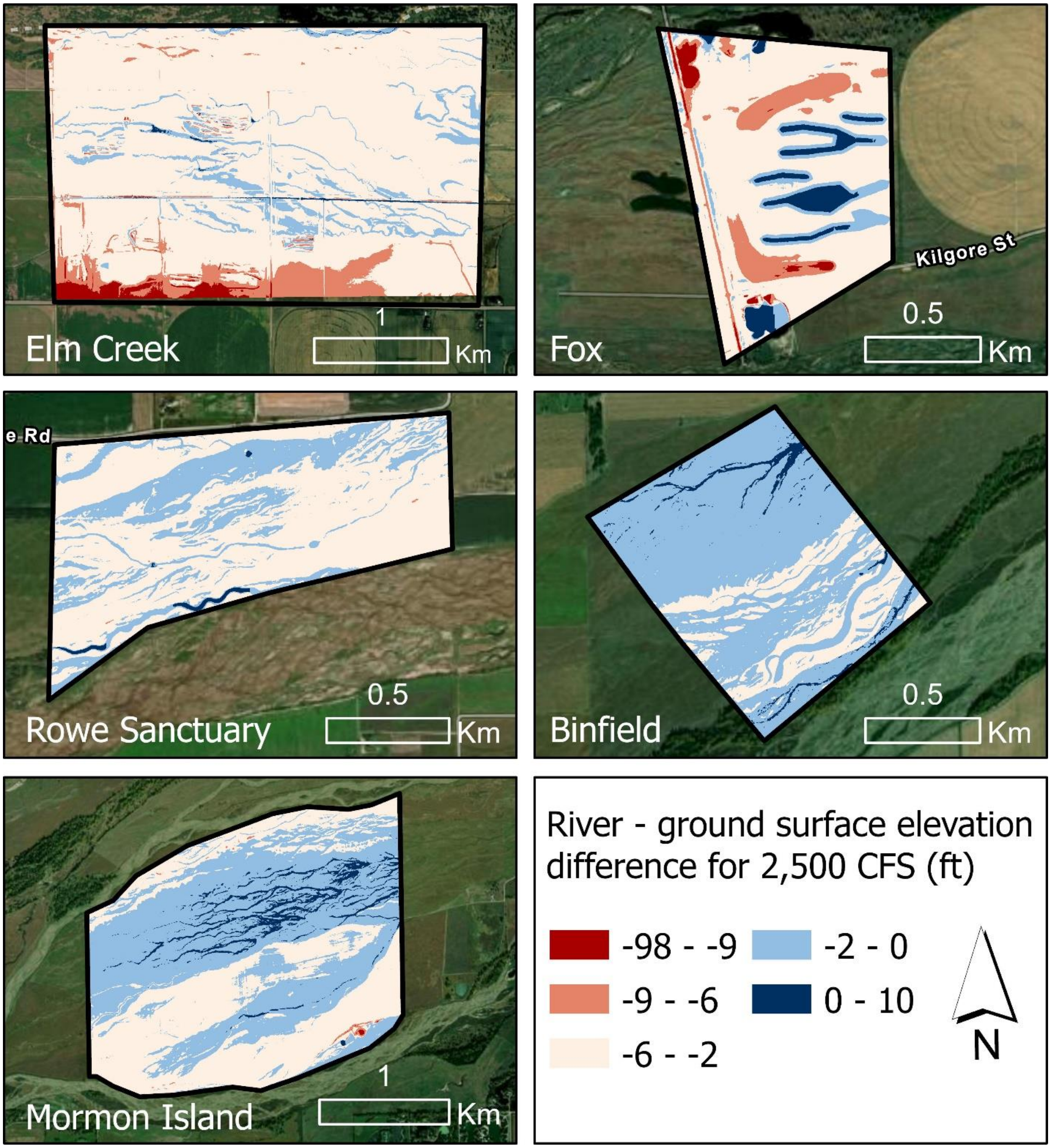


Figure 25 - Differenced river and ground surface elevation rasters for Elm Creek, Fox, Rowe Sanctuary, Binfield, and Crane Meadows with modeled discharge of 2500 cfs.

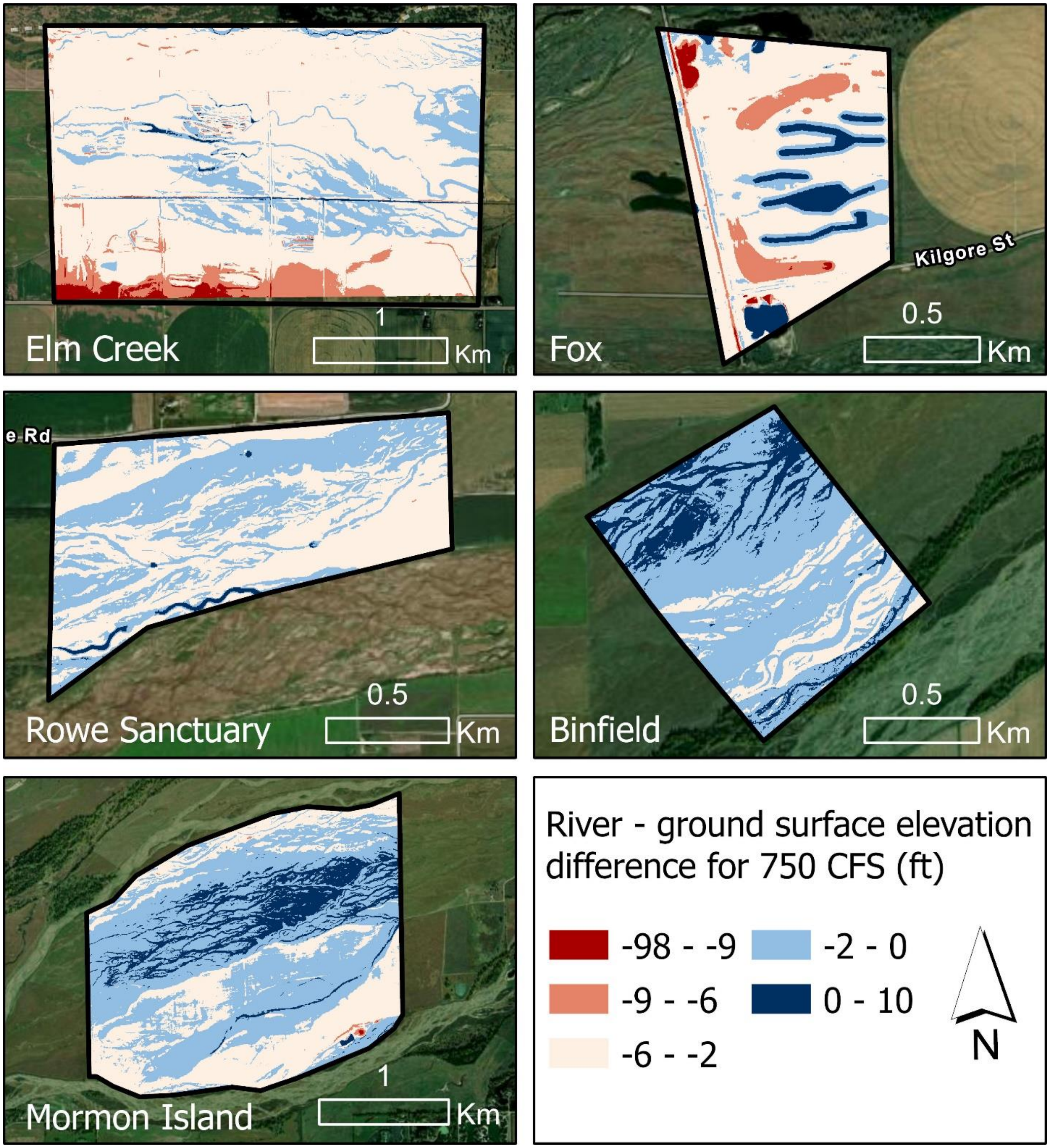


Figure 26 - Differenced river and ground surface elevation rasters for Elm Creek, Fox, Rowe Sanctuary, Binfield, and Crane Meadows with modeled discharge of 2500 cfs.

DEVs were compared to L7th predicted vegetation and landcover datasets for the Fox and Binfield sites to evaluate how DEV compares to other established classification methods. Select landcover datasets included the 2005 Platte River vegetation and landcover dataset (Brei and Bishop, 2008) and an ecotope based landcover classification that combines Brei and Bishop (2008) with wetland

and flooding frequency datasets as described by Baasch et al. (2022). Reproduced DEVs for 3,500 cfs are presented along with comparison datasets in Figure 28 through Figure 34. 3,500 cfs results are discussed herein, but patterns and resulting conclusions are generally the same across all discharges evaluated. Also, DEVs and L7th values have a finer resolution than landcover classes (1m versus 10m), so area coverage is not directly comparable.

Patterns of greater DEV generally correspond to patterns of greater L7th values and mesic wet meadow landcover classes. Comparing patterns of classes, DEVs are typically greater near the channel and lower further from the channel than corresponding L7ths, demonstrated particularly well in the darker blue categories for Figure 28 and Figure 29. Patterns of DEV and L7th are very similar considering that L7th maps were produced with extensive, long-term data collection and subsequent processing whereas DEV required no field data collection and minimal data processing. Notably, landcover classes show broader distribution of wet landcover classes than L7th or DEV.

Table 12 provides a summary of corresponding classes and values for the four datasets. The L7th emergent categories were generally associated with mesic wet meadow landcover, sedge meadow L7th categories with xeric wet meadow landcover, and mesic prairie L7th categories with meadow sand ridge. There was no distinct landcover class associated with dry ridge L7th categories, though undisturbed grassland was concurrent at the Fox site. Notably, "riparian shrubland" landcover classes were associated with L7th emergent categories, indicating wooded areas have relatively shallow groundwater tables. The 2008 vs 2022 landcover classifications were highly similar at study sites due to use of the Brei and Bishop (2008) layer as a base by Baasch et al. 2022. The main difference was that the 2022 landcover classification had added classes for 'wet prairie', 'meadow marsh', and 'agricultural wetland'. Notably, the majority of the Fox site was characterized as 'ag' by Brei and Bishop prior to site excavation and restoration and as 'upland agriculture' with low areas classified as 'agricultural wetland' by Baasch, which is not currently accurate for the site. As such, the excavated areas at the Fox site were not incorporated into either landcover classification. Animated Figures 1 and 2, included with select versions of this report, demonstrate how landcover class boundaries align.

Table 12 - Comparison of wet meadow classification datasets

Dataset	DEV (3500 cfs)	L7th (2015)	Brei and Bishop (2008)	Ecotope (Baasch et al., 2022)
Corresponding Landcover Type	0-3 m	Emergent (> 0.2 m)	Riparian shrubland, riparian woodland, mesic wet meadow	Shrubland, woodland, meadow-marsh, wet prairie
	-0.6 to 0 m	Sedge Meadow (-0.3 to 0.2m)	Xeric wet meadow, meadow sand ridge	Prairie
	-2 to -0.6 m	Mesic Prairie (-1.35 to -0.2m)	Meadow sand ridge, xeric wet meadow	Prairie
	<-3 m	Dry Ridge (<-1.35m)	Undisturbed grassland	Prairie

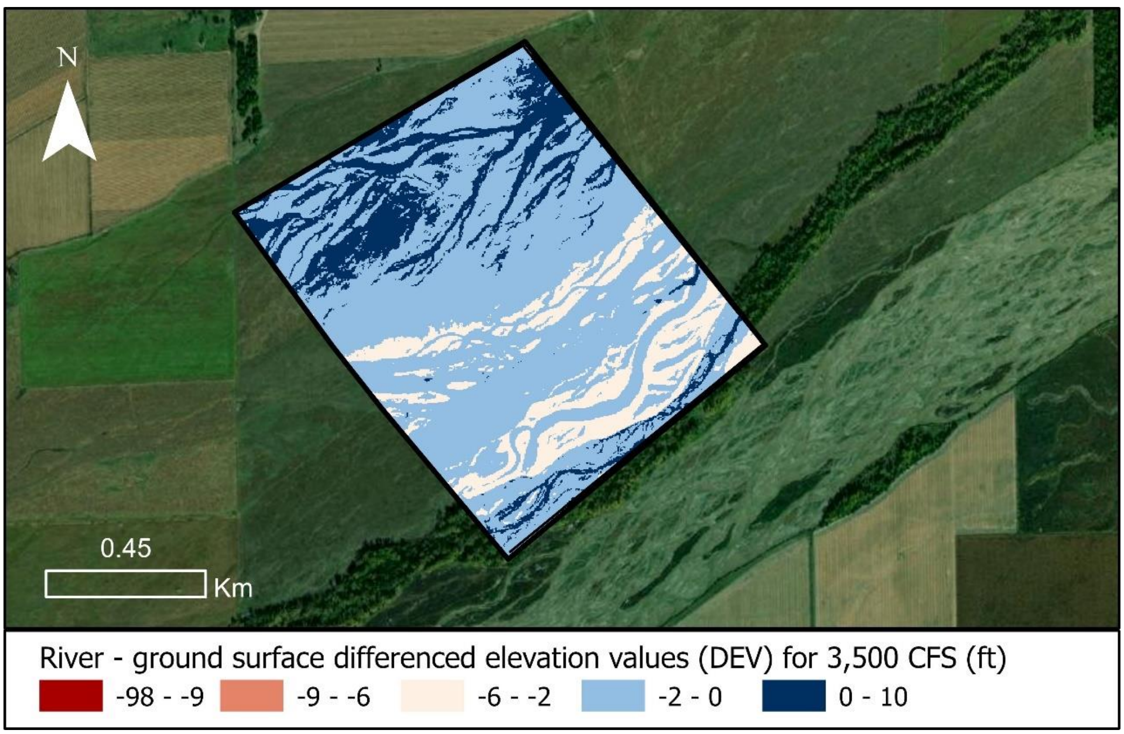
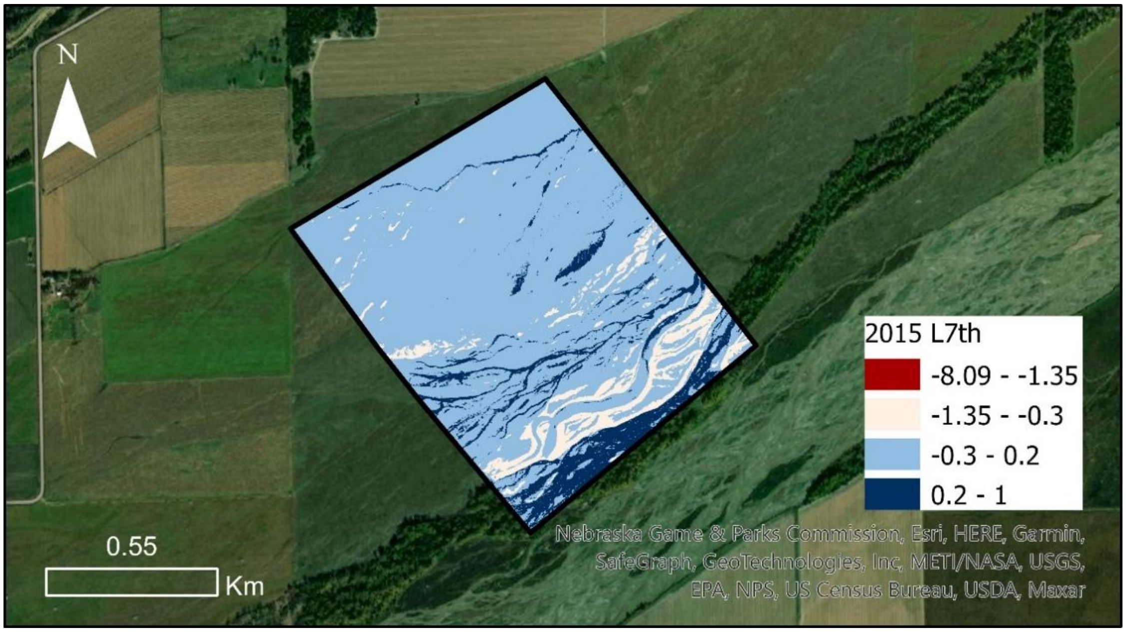
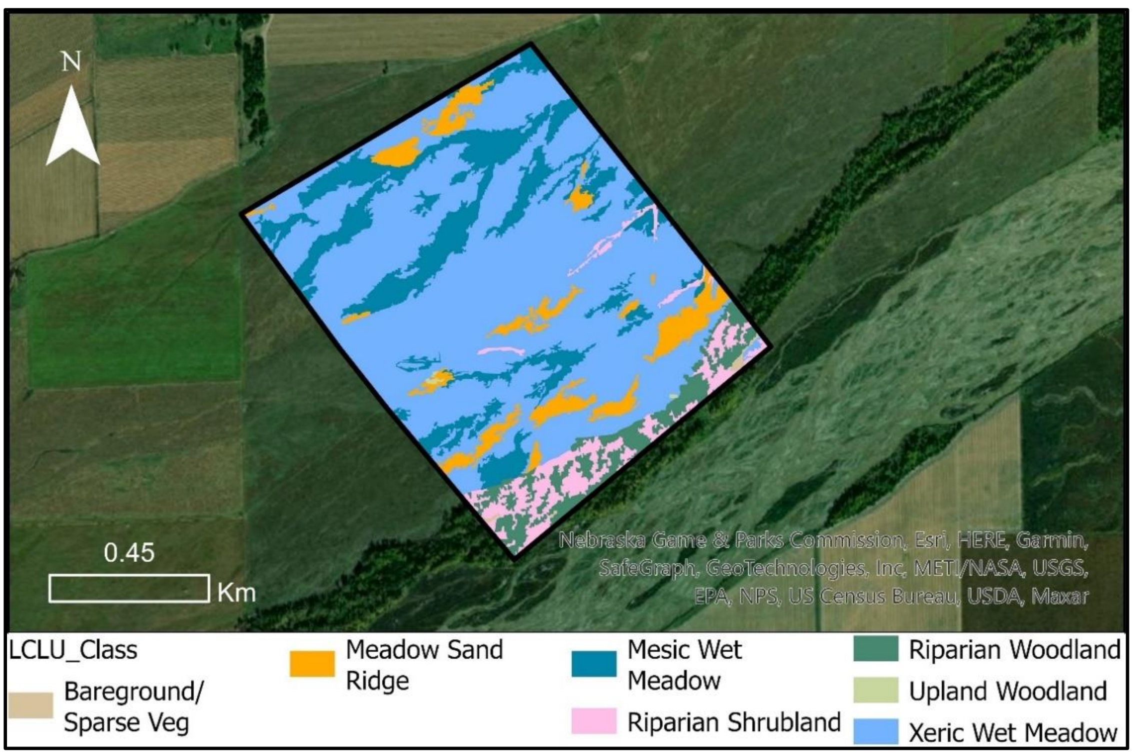


Figure 27 - Differenced elevation surface for 3,500 cfs at the Binfield Site.

358 

359 *Figure 28 - 2015 L7th DTGW for the Binfield Site.*

360 

361 *Figure 29 - 2008 Brei and Bishop landcover classes for the Binfield Site.*

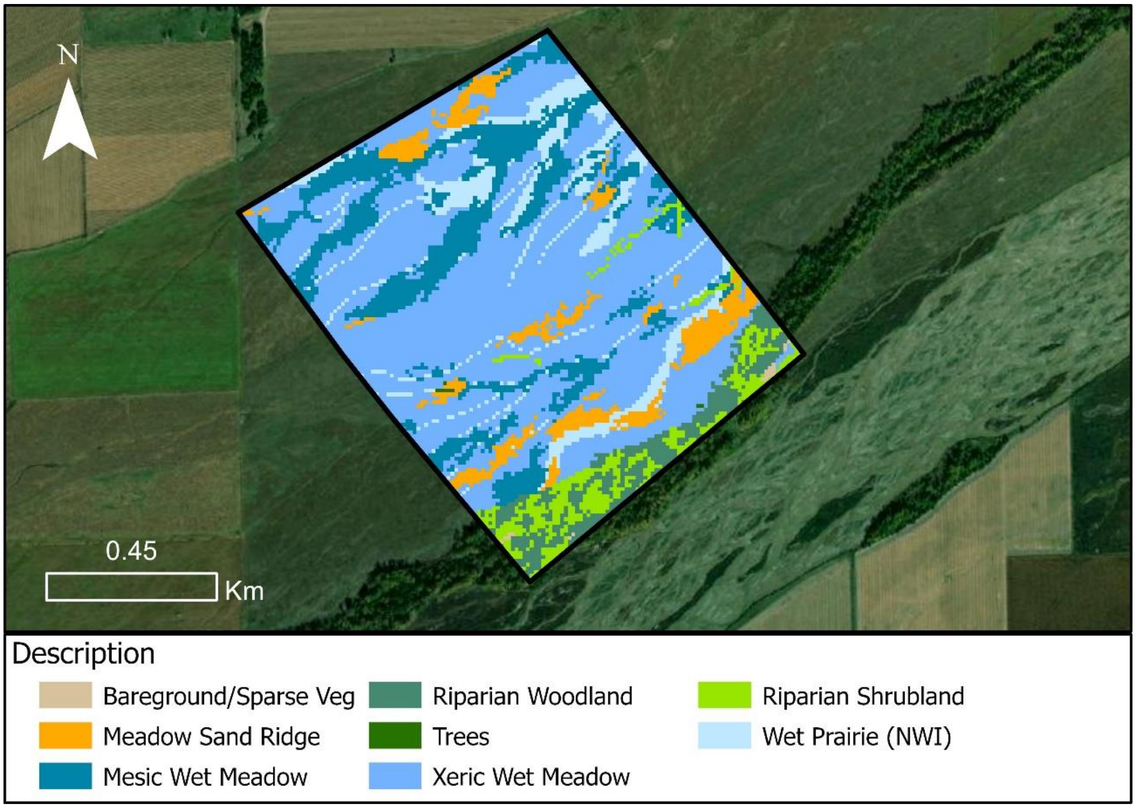


Figure 30 - 2022 Baasch et al. 2022 ecotope-based landcover classes for the Binfield Site. Note, Mesic Wet Meadow is lumped with meadow-marsh by Basch et al.

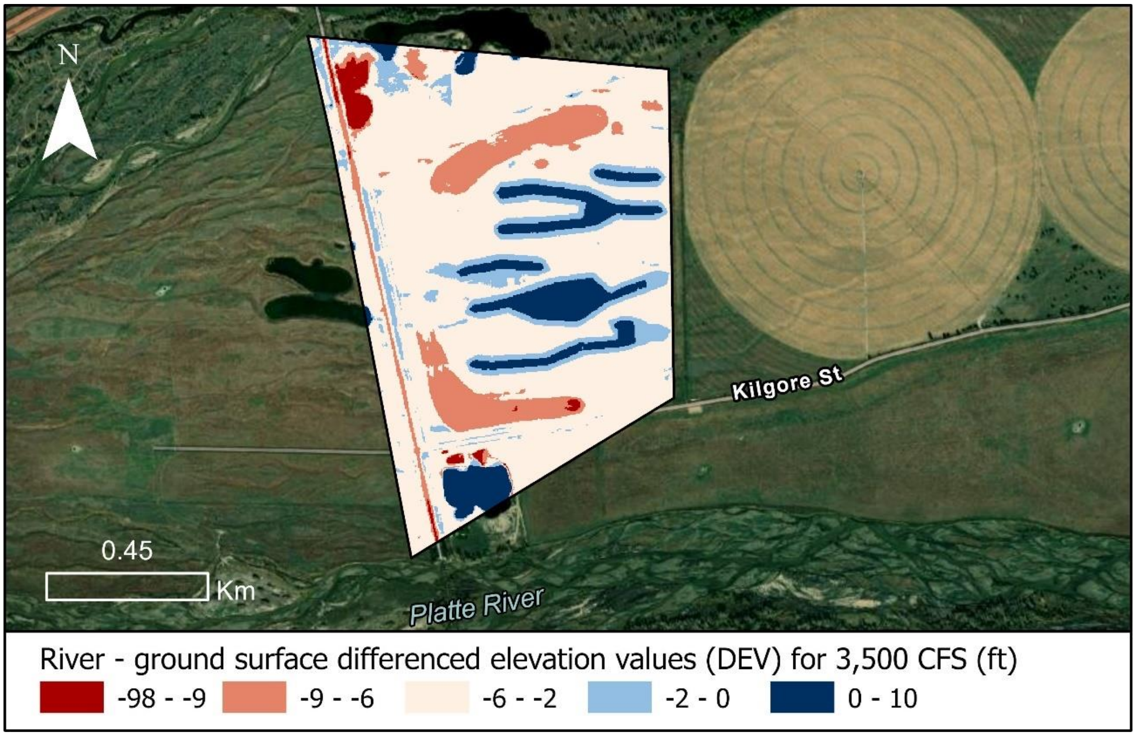


Figure 31 - - Differenced elevation surface for 3,500 cfs for the Fox Site.

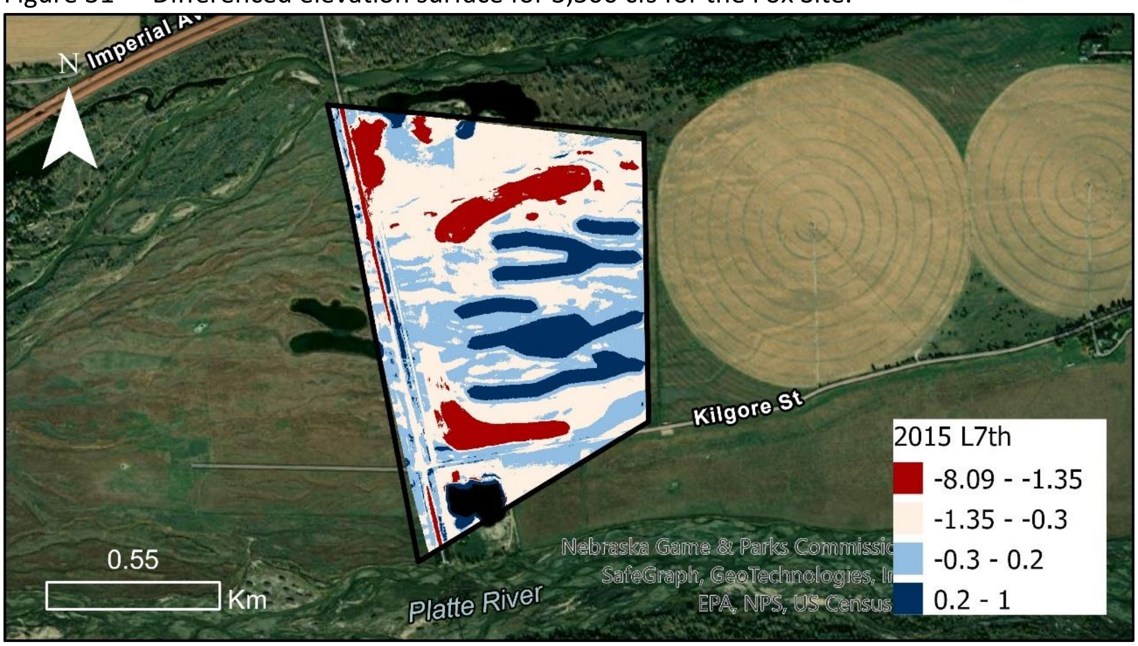
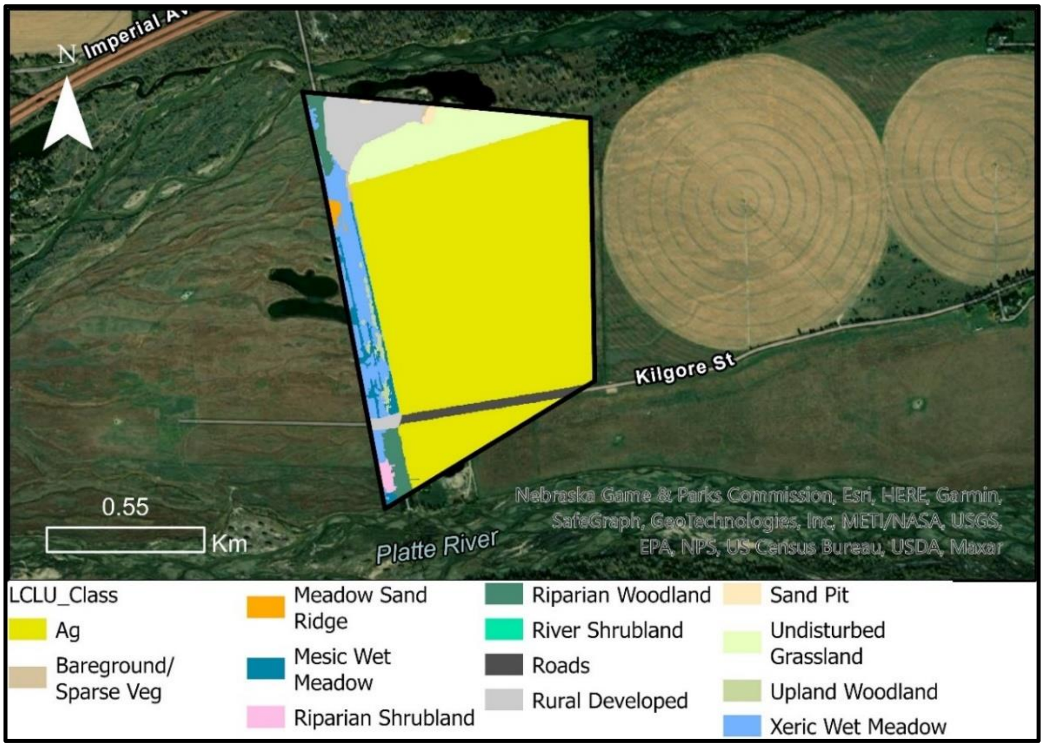
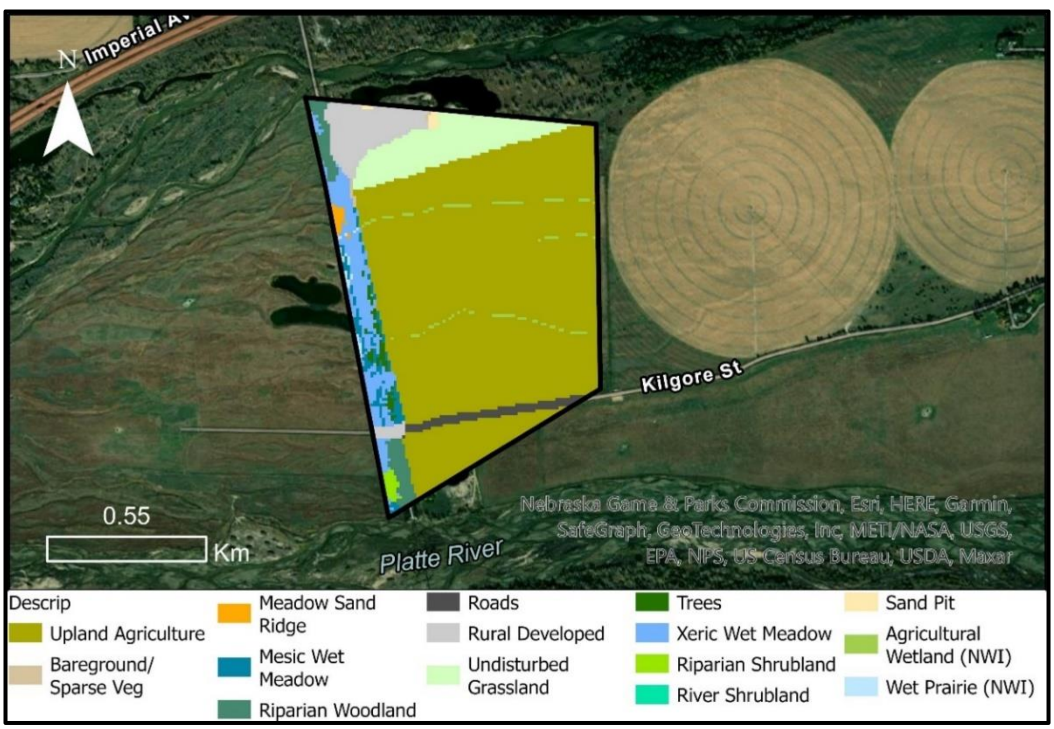


Figure 32 - 2015 L7th DTGW for the Fox Site.



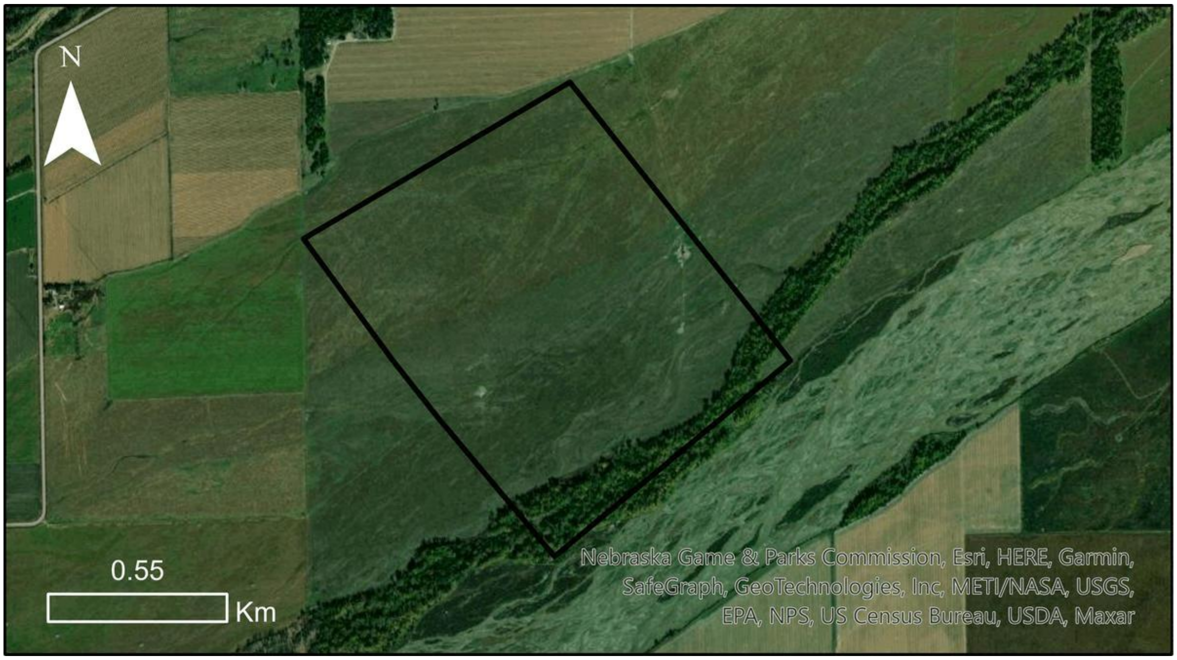
369

370 *Figure 33 - 2008 Brei and Bishop landcover classes for the Fox Site prior to restoration.*

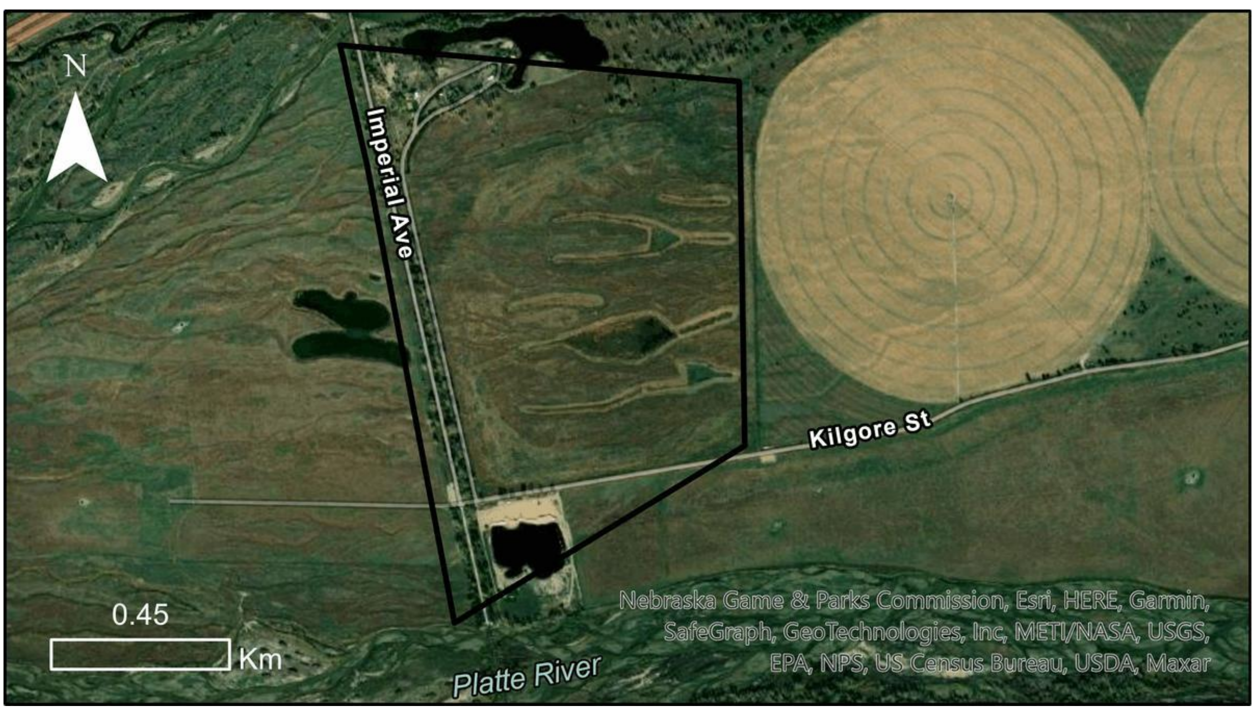


371

372 *Figure 34 - 2022 Baasch et al. 2022 ecotope-based landcover classes for the Binfield Site. Note, Mesic*
373 *Wet Meadow is lumped into meadow-marsh by Basch et al.*



Animated Figure 1– Compilation of 4 landcover class datasets for Binfield Site in order: 1) Satellite imagery, 2) L7th for 2015, 3) DEV, 4) Landcover and vegetation (Brei and Bishop, 2008), and 5) Ecotope-based landcover (Baasch et al., 2022).



Animated Figure 2 – Compilation of 4 landcover class datasets for Fox Site in order: 1) Satellite imagery, 2) L7th for 2015, 3) DEV, 4) Landcover and vegetation (Brei and Bishop, 2008), and 5) Ecotope-based landcover (Baasch et al., 2022).

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. Grassland sites currently managed by the Program could be screened using this method for consideration for future restoration or management as wet meadows. Figure 34 includes the area around the Fox and Binfield sites, 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.

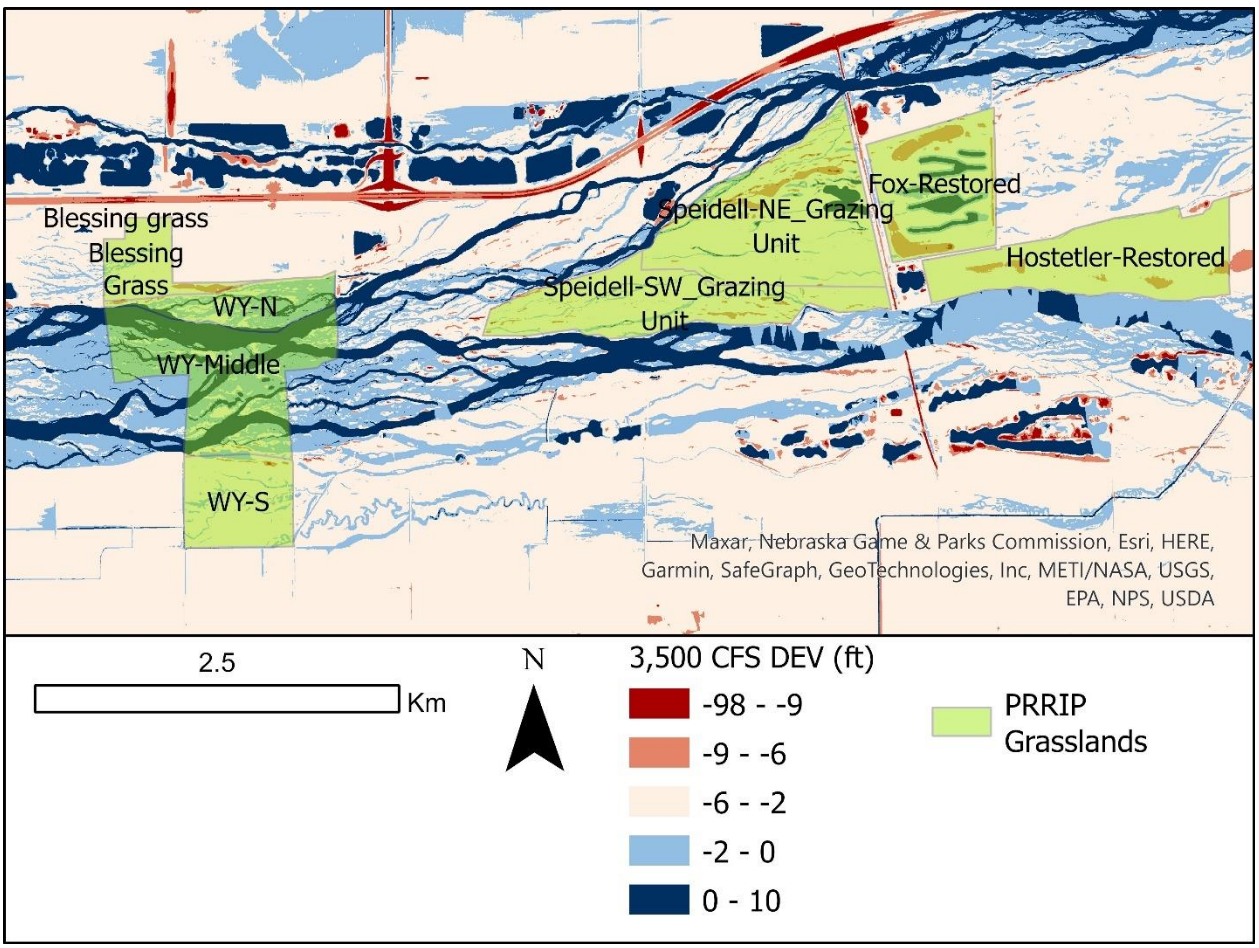


Figure 35 – Map of differenced raster surfaces along with other grassland sites managed by the Program. Light green indicates PRRIP managed properties.

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. Link

Baasch, D. M., Farrell, P. D., Pearse, A. T., Brandt, D. A., Caven, A. J., Harner, M. J., ... & Metzger, K. L. (2019). Diurnal habitat selection of migrating Whooping Crane in the Great Plains. Link

Baasch, D. M., A. J. Caven, J. G. Jorgensen, R. Grosse, M. Rabbe, D. M. Varner, and T. LaGrange. 2022. Whooping Crane (Grus americana) use patterns in relation to an ecotope classification in the Central Platte River Valley, Nebraska, USA. Avian Conservation and Ecology 17(2):35. https://doi.org/10.5751/ACE-02311-170235 Link

Brei, J., Bishop, A. A. (2008). Platte River Vegetation Mapping Project 2005 Land Cover Methods Summary. Great Plains GIS Partnership. Link

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 Link

Caven, A. J., Pearse, A. T., Brandt, D. A., Harner, M. J., Wright, G. D., Baasch, D. M., ... & Lacy, A. E. (2022). Whooping crane stay length in relation to stopover site characteristics. Link

Chen, X. (2004). Streambed hydraulic conductivity for rivers in south-central Nebraska 1. JAWRA Journal of the American Water Resources Association, 40(3), 561-573. Link

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

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. Link

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

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. Link

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. Link

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

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. Link

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. Link

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. Link

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. Link

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

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. Link

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

Grosse, R. (2021). GIS Processing Methods for “Whooping Crane Use Patterns in Relation to Vegetation Classification in the Central Platte River Valley, NE.” NE-S6 2020 #F20AS00070. Link

Hanson, P.R., Dillon, J.S., Bruihler, J., and Howard, L.M., 2014, Surficial Geology of the Kearney 7.5 Minute Quadrangle, Nebraska. Link

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. Link

Howlin, S., Nasman, K. (2017) Correlates of Whooping Crane Habitat Selection and Trends in Use in the Central Platte River, Nebraska. Platte River Recovery Implementation Program, Kearney, NE, and West EcoSystems Technology, Inc., Cheyenne, WY, USA. Link

HPRCC, (2021) High Plains Regional Climate Center, accessed September 2021, https://hprcc.unl.edu/awdn/

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. Link

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

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. Link

LaGrange, T. 2022. Guide to Nebraska's Wetlands and their Conservation Needs. 3rd ed. The Nebraska Game and Parks Commission, Lincoln, NE. Link

Lingle, G. R., Wingfield, G. A., & Ziewitz, J. W. (1991). The migration ecology of Whooping Cranes in Nebraska, USA. In Proceedings of the 1987 International Crane Workshop. International Crane Foundation, Baraboo, WI (pp. 395-401). Link

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). Link

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. Link

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. Link

Mitsch, W. J., Gosselink, J.G. (2015) Wetlands, John Wiley & Sons, Incorporated, ProQuest Ebook Central. Link

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

Pickens, B. A., King, S. L., Vasseur, P. L., Zimorski, S. E., & Selman, W. (2017). Seasonal movements and multiscale habitat selection of whooping crane (Grus americana) in natural and agricultural wetlands. Waterbirds, 40(4), 322-333. Link

PRRIP (2012). Fox Wet Meadow Excavation P11-005. Link.

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

PRRIP (2016). PRRIP GRASSLAND VEGETATION ASSESSMENT FINAL REPORT Link

PRRIP (2019). PRRIP GRASSLAND VEGETATION ASSESSMENT FINAL REPORT. Link

PRRIP (2021). Wet Meadows Data QC Report. Link.

PRRIP (2022). PRRIP GRASSLAND VEGETATION ASSESSMENT REPORT. 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. Link

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. Link

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. Link

Song, J., & Chen, X. (2010). Variation of specific yield with depth in an alluvial aquifer of the Platte River valley, USA. International Journal of Sediment Research, 25(2), 185-193. Link

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

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

US Army Corps of Engineers (USACE). (2012). National Wetland Plant List Indicator Rating Definitions. Link

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

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. Link

Whiles, M.R. and B.S. Goldowitz. 1998. Biological responses to hydrologic fluctuation in wetland sloughs of the central Platte River. Pages 3-18 in G. Lingle, editor. Proceedings of the ninth Platte River ecosystem symposium. University of Nebraska Cooperative Extension-Platte Watershed Program, Kearney, Nebraska. Link

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

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

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. Link

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. Link

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

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. Link

Appendix A – Wet Meadow Definitions

Joint Study Biology Workgroup – USFWS 1990 -

B. The Grassland Complex

The vegetative community referred to as the Grassland Complex historically extended beyond the current high bank of the river channel. The Platte River Whooping Crane Trust analyzed the probable changes in the areal extent of the grassland complex vegetation based on the present area of cover types and the assumption that cropland and most trees did not exist in pre-settlement times within the river channel.

Losses of wet meadows (grasslands in the historic high banks of the river), sandhills prairie, alfalfa and emergent wetlands on lands within 3.5 miles of the river channel have been substantial. In the Big Bend of the Platte River grassland losses have ranged from 56% near Odessa to about 80% near Grand Island and Shelton. The average loss of the grassland complex in the Big Bend reach within 3.5 miles of the river channel has been estimated at 72% (Currier et al. 1985).

Losses of the wet meadow complex along the North Platte River have been considerably less than on the Platte River, ranging from 25% near Lewellen to 37% near North Platte. An average of 31% of the grassland have been converted to other uses along the North Platte River (Table 8).

Much of the remaining native grasslands are of marginal value to endangered species and other migratory birds because they are located in small, disjunct tracts within the Platte and North Platte River valleys. In the Big Bend region over 80% of the grassland ~~complex (about 2000 tracts) are~~ less than 50 acres in size (Currier et al. 1985). Small tracts of grassland make up 22% of the grassland area in the Big Bend, but their mean size is 9 acres. About 40% of the grassland area in the Big Bend is concentrated in a few relatively large tracts of 300 acres or more (mean = 546 acres). For many bird species the smallest grassland tracts (0-50 acres) do not provide suitable habitat no matter how many are available. Human disturbance in the form of

PRRIP Land Plan, 2005

2. Wet Meadow Habitat	Characteristics
Location	Within 2 miles of the above-described channel area.
Size	Approximately 640 contiguous acres or more.
Distance from Disturbance	In general, not less than 0.5-mile distant or appropriately screened from potential disturbance. Potential disturbances may include roads, railroads, occupied dwellings, bridges or other activities that would disturb target species from using a site.
Vegetation Composition	Native prairie grasses and herbaceous vegetation, lacking or mostly lacking sizable trees and shrubs, occurring in a mosaic of wetland (hydrophytic) and upland (non-hydrophytic) plants.
Hydrology	Swales subirrigated by ground water seasonally near the soil surface and by precipitation and surface water, with the root zone of the soil continuously saturated for at least 5 - 12.5% of the growing season. Except immediately following precipitation events, higher areas may remain dry throughout the year.
Topography and Soils	The topography is generally level or low undulating surface, dissected by swales and depressions. Mosaic of wetland soils with low salinity in swales and non-wetland soils occurring in uplands.
Food Sources	Capable of supporting aquatic, semi-aquatic, and terrestrial fauna and flora characteristic of wet meadows; especially aquatic invertebrates, beetles, insect larvae, and amphibians.

Brinley Buckley et al., 2021 – "Wet meadow characteristics vary based on geographic location throughout the world, including montane, arid, and prairie landscapes, they share similar defining characteristics. Wet meadows have hydric soil features, support wetland vascular plant species, and have temporary and recurrent hydroperiods. The intermittent cycles of inundation serve as a control over chemical and biological processes and disseminate biotic and abiotic material, creating heterogeneity and influencing species richness, abundance, production, and trophic structure. Wet meadows in the CPRV receive moisture from groundwater, streamflow, precipitation, and overland flooding, and the interplay among these drivers and how they vary seasonally are poorly understood."

Currier and Henszey 1996 – "A complex of grassland and wetland areas in central and western Nebraska within close proximity to the Platte River channel and with a hydrologic connection to river flows. By this definition, wet meadows are confined to the river floodplain, and for the most part, are located within 1 to 2 miles of the channel. They generally have pooled or ponded standing water during a portion of the year (primarily spring and early summer) and are hydrologically interconnected with the river through a common groundwater table and on occasion by surface water overflow."

LaGrange 2010 – Wetlands situated on floodplain soils with rapid permeability and receiving minimal to regular out of bank flooding.

Mitsch and Gosselink,2015 – Grassland with waterlogged soil near the surface but without standing water for most of the year.

Tiner, 2016 - Herbaceous wetlands—wet meadows (grasses, sedges, rushes, and/or forbs)—have formed on wet mineral soils. Some were originally forested wetlands but were cleared and are maintained for pastures, while others were always herbaceous due to local conditions (e.g., montane wet meadows) and semiarid or arid conditions (e.g., Nebraska Sandhills). Many, if not most, are associated with groundwater seepage, especially those occurring on gentle slopes. In semiarid regions, precipitation is the major source of groundwater recharge and plant growth (ET) exerts a strong influence on groundwater levels. Wet meadows may have standing shallow water for short periods, especially in micro depressions, but typically their soils are saturated for much of the growing season. In the Prairie Pothole Region where they often occur as one of the upper zones around the rim of the potholes during wet years, they are subjected to brief flooding after snowmelt. In humid regions, they may have standing water for longer periods and occur at the upper edges of marshes. In western mountains and in semiarid and arid regions, many wet meadows are supported by springs that emerge from adjacent slopes, faults, or regional groundwater systems, while others are saturated due to perched groundwater conditions. In the Nebraska Sandhills, wetlands in one interdunal valley may be in a recharge area and may drawdown more rapidly than wetlands in another valley at the end of a groundwater flow path where water levels are maintained for longer periods. In the Great Lakes, wet meadows occur at elevations above the coastal marshes, where they are minimally inundated during high lake-level years but remain too dry during low lake-level years to be invaded by more robust emergent plants.

Appendix B – PRRIP Wet Meadow Tracts

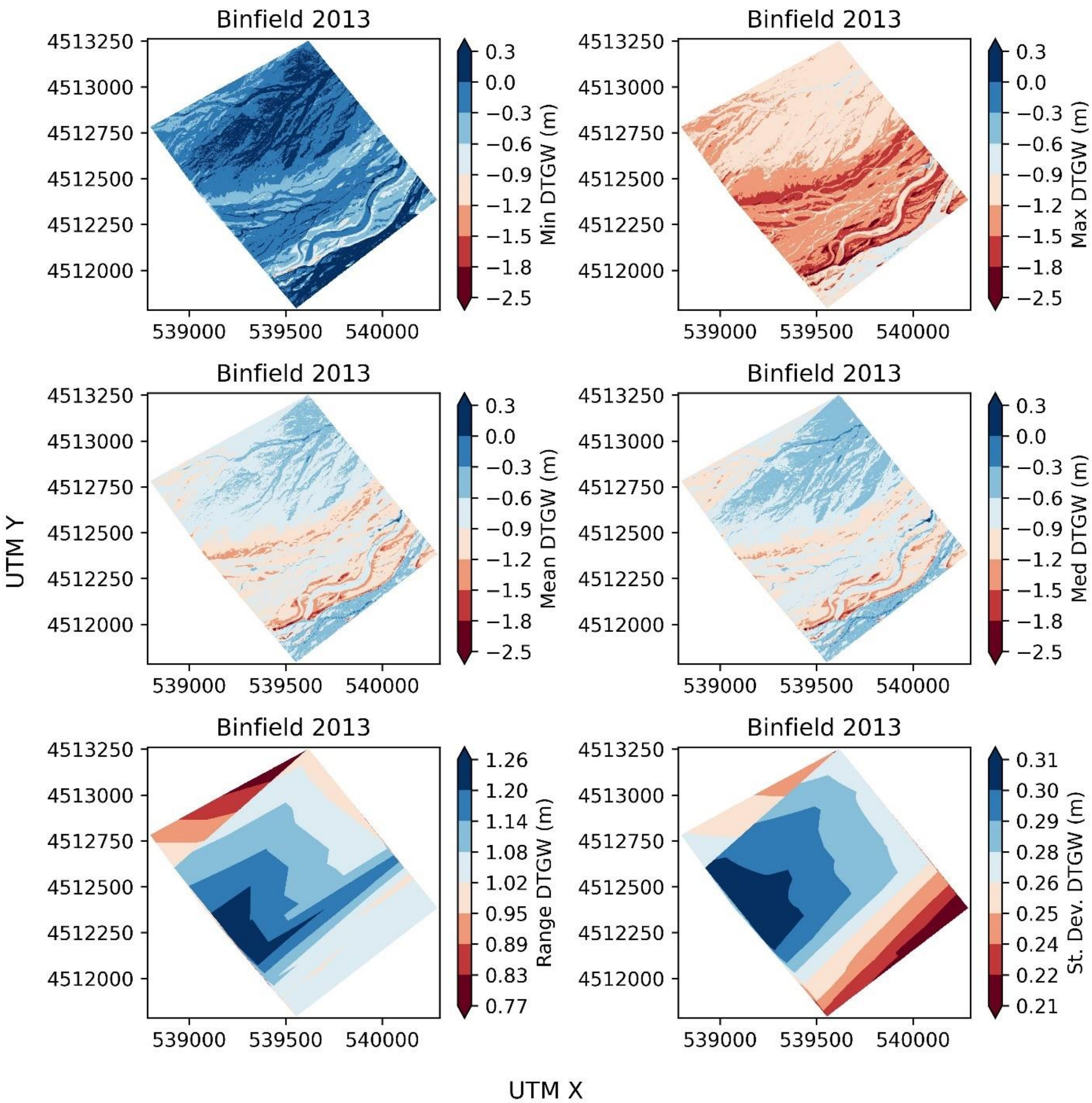
Table B1 – PRRIP managed tracts containing lowland grassland.

Tract #	Historic Name	County	Area (Acres)
2008001	WY-S	Kearney	116.066325
2009004	Hostetler-Restored	Buffalo	227.932296
2009001	Fox-Restored	Buffalo	177.364609
2010001	Morse-N	Phelps	166.274385
2010001	Morse Restored Crop	Phelps	30.198415
2010001	Morse-hay-S	Phelps	43.709619
2008001	WY-Middle	Buffalo	243.242295
2008001	WY-N	Buffalo	81.349249
2008002	NW-BSR	Phelps	96.385487
2010004	Binfield West Pasture	Hall	361.165503
2010004	Binfield South Hay meadow	Hall	29.375311
2010004	Binfield East Pasture	Hall	178.836897
2010004	Binfield West Haymeadow	Hall	124.138751
2010004	Binfield South meadow	Hall	56.81152
2008002	CWR Marshall Calfing	Phelps	92.281169
2009005	McCormick North Island	Buffalo	73.701431
2009007	Cook Hay-W	Phelps	48.848575
2009007	Cook Hay-E	Phelps	10.874718
2012002	John-North wet meadow	Buffalo	373.055172
2012001	Sullwald-Hay meadow	Buffalo	35.686508
2012004	DeBoer Wetland	Gosper	69.829884
2012003	Blessing Grass	Buffalo	67.63525
2009003	Dyer-Grazing Unit-S	Phelps	98.83293
2015001	Speidell-NE_Grazing Unit	Buffalo	298.273016
2015001	Speidell-SW_Grazing Unit	Buffalo	204.327286
2012003	Blessing Grass	Buffalo	67.73662
2013001	Leihs-North Wetland	Dawson	33.111937
2013001	Leihs-SW grass/forage	Dawson	10.02971
2008002	CWR East Lloyd Island	Dawson	186.132277
2008002	CWR South OCSW	Dawson	72.835992
2015003	Blue Hole East Hay Meadow	Buffalo	8.117969
2008002	SW-BSR	Phelps	121.345494
2010001	Morse-SW_BSR	Phelps	121.890213
2010001	Morse Middle BSR	Phelps	128.95638
2008002	NE-BSR	Phelps	166.209524
2021001	Meyers Wet Meadow	Merrick	86.349822
2018001	Dippel South Acrreation	Buffalo	180.166181
2018001	Dippel East Pasture	Buffalo	156.935921
2019001	Bergen South Pasture	Hamilton	69.736277
2019001	Bergen North Pasture	Hamilton	49.734458
2018001	Dippel West & Central Pasture	Buffalo	164.152393
2018001	Dippel Hay Meadow	Buffalo	9.6
		TOTAL	**4939.24**
		WET MEADOW	**2010.95**

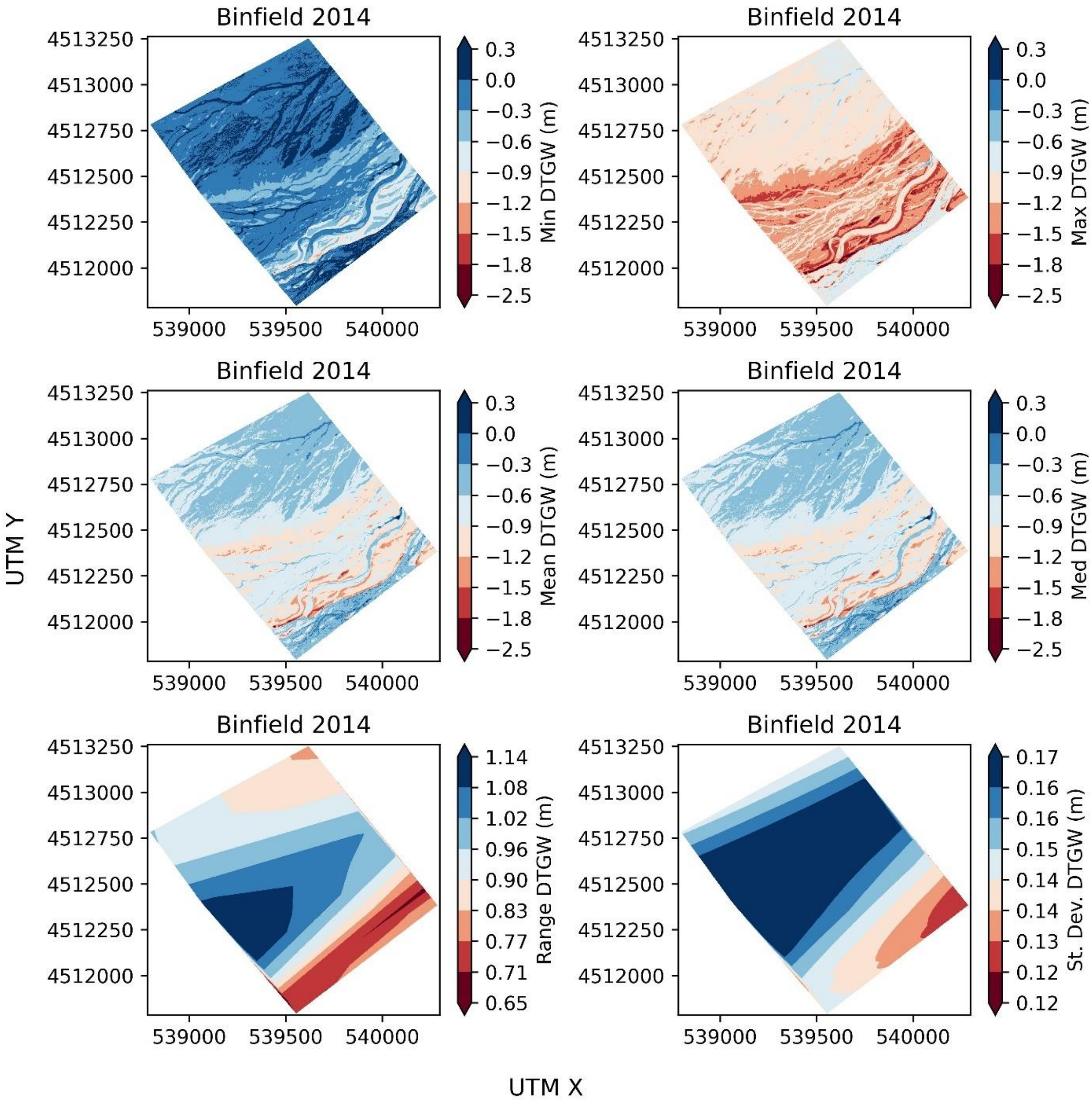
Appendix C – Groundwater contours from existing groundwater model

These figures support the decision to use linear interpolation for daily groundwater tables for spatial statistics. The figures demonstrate hydraulic head contour results from groundwater models for each site, demonstrating smooth, evenly distributed potentiometric head surfaces between wells.

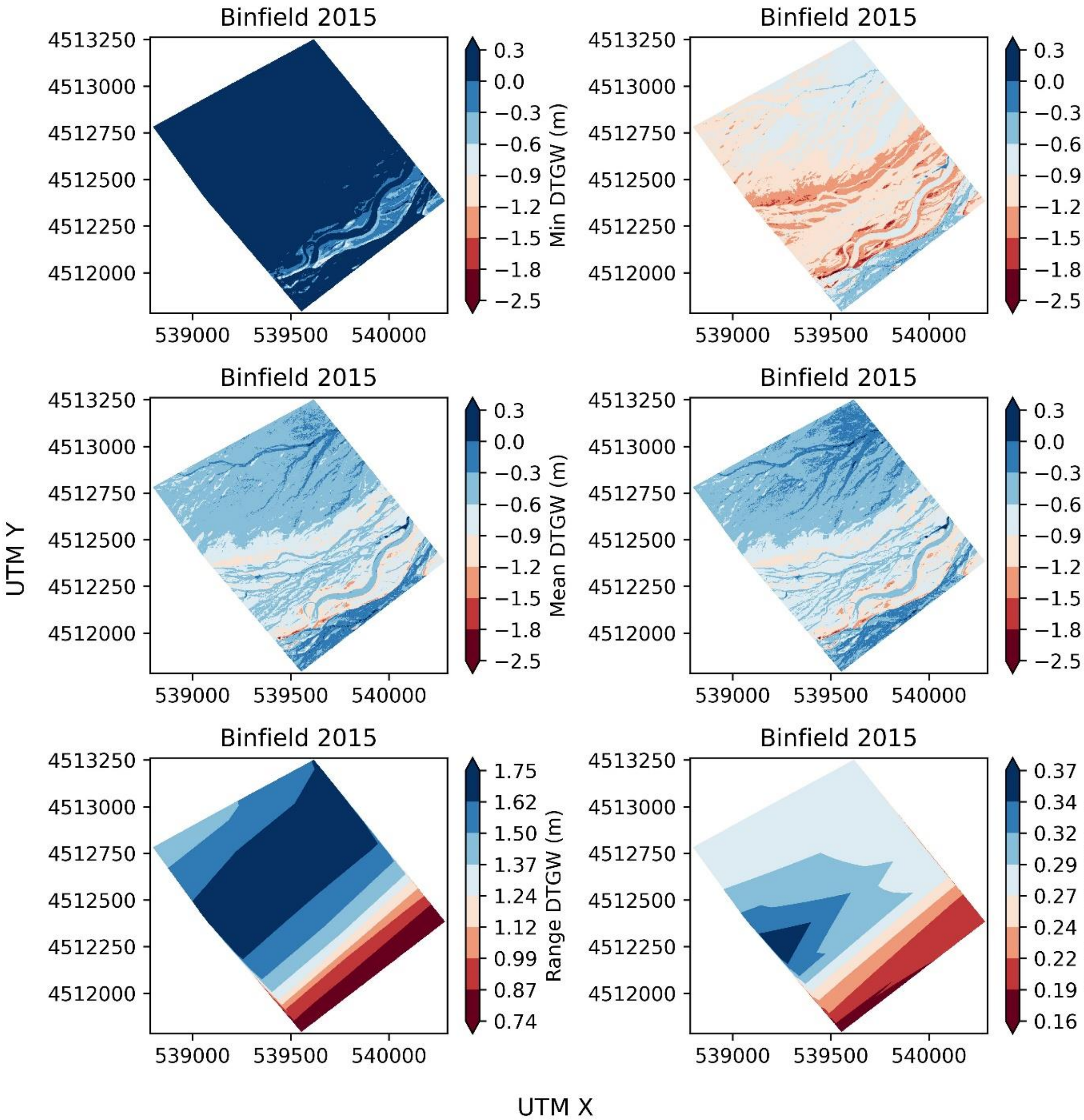
1 ## Appendix D – Groundwater statistics



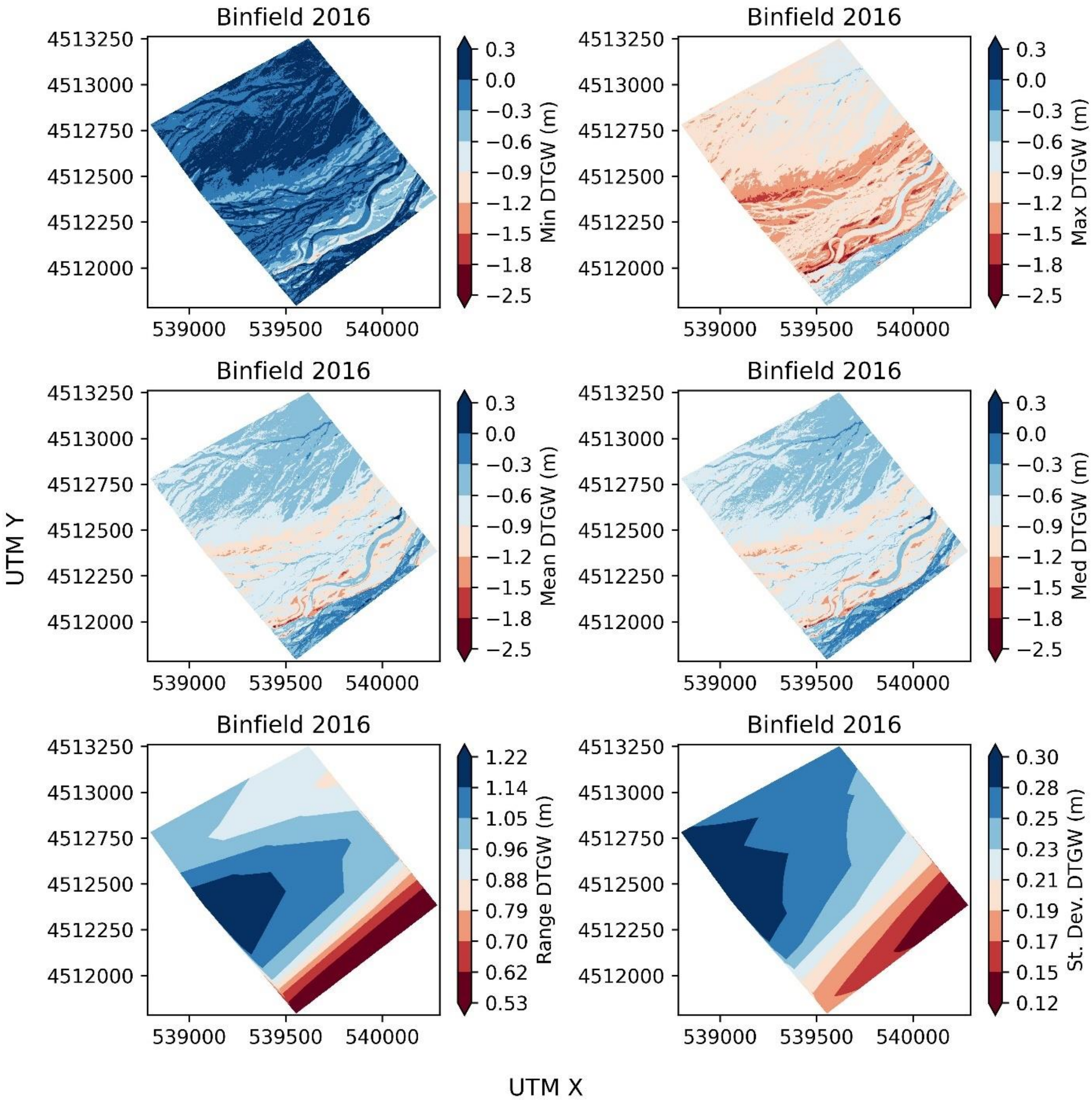
2

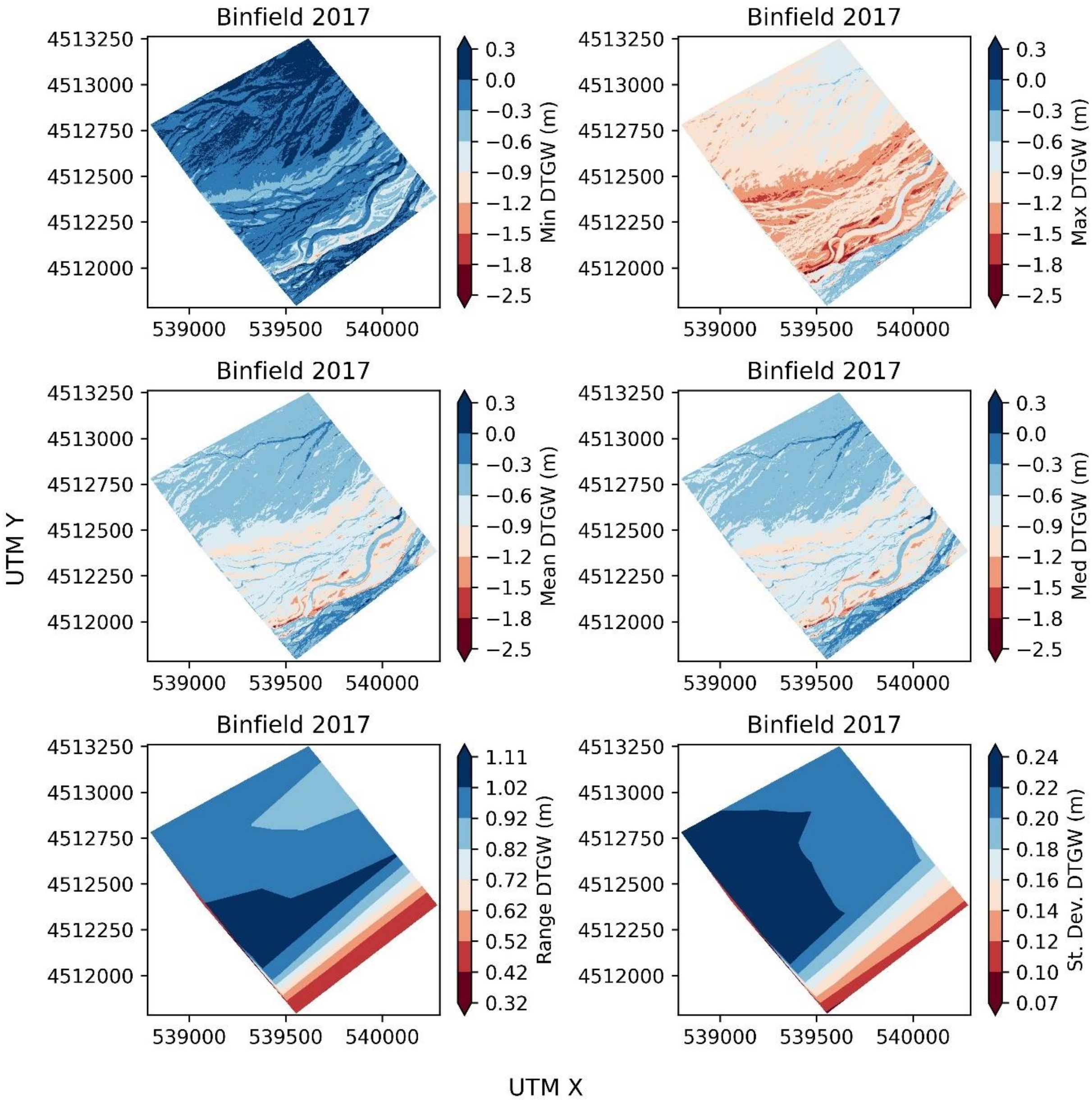

Binfield 2014
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
UTM Y
UTM X
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.14
1.08
1.02
0.96
0.90
0.83
0.77
0.71
0.65
0.17
0.16
0.16
0.15
0.14
0.14
0.13
0.12
0.12

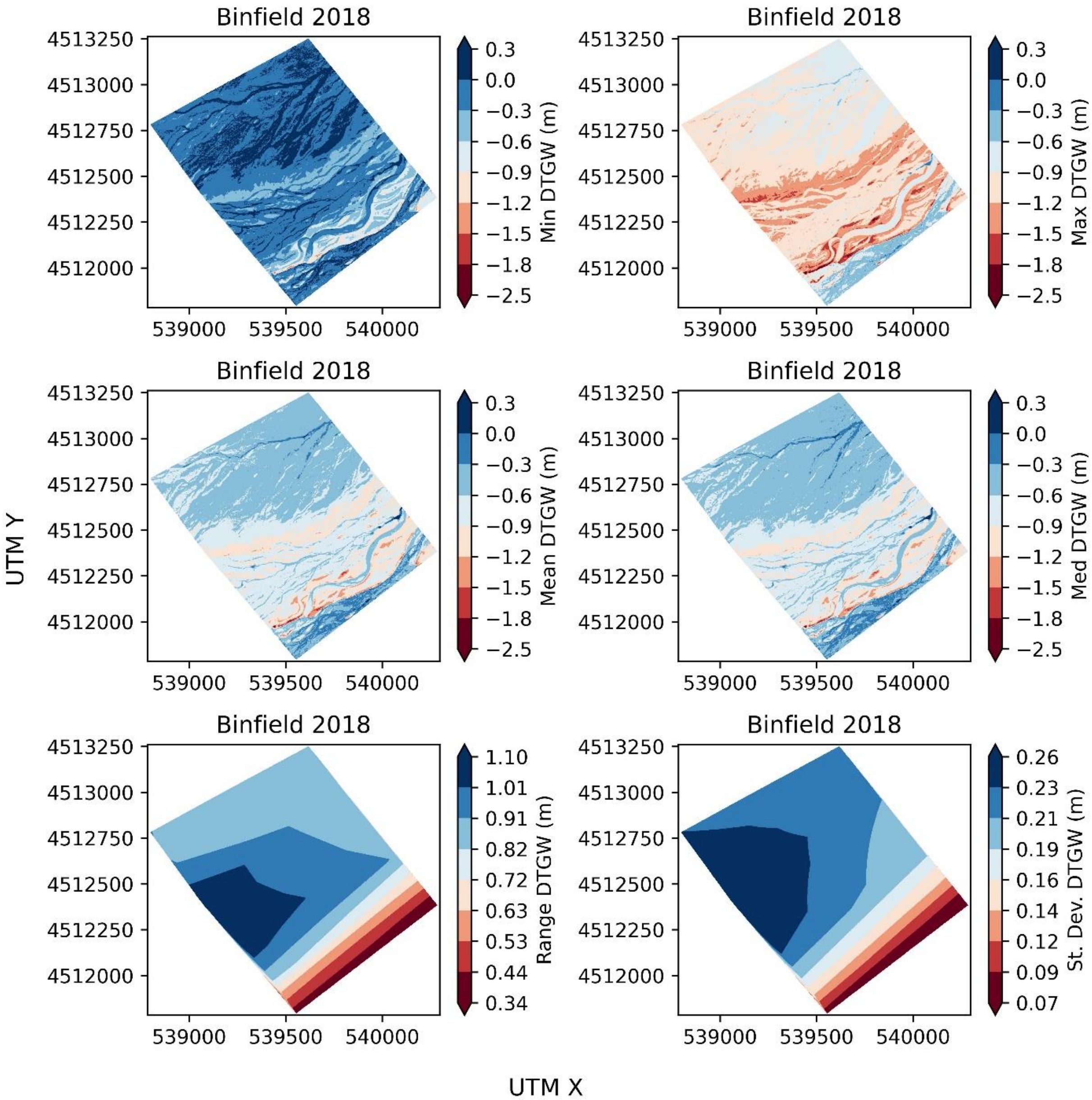
3


Binfield 2015
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.75
1.62
1.50
1.37
1.24
1.12
0.99
0.87
0.74
0.37
0.34
0.32
0.29
0.27
0.24
0.22
0.19
0.16
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
UTM Y
UTM X

4

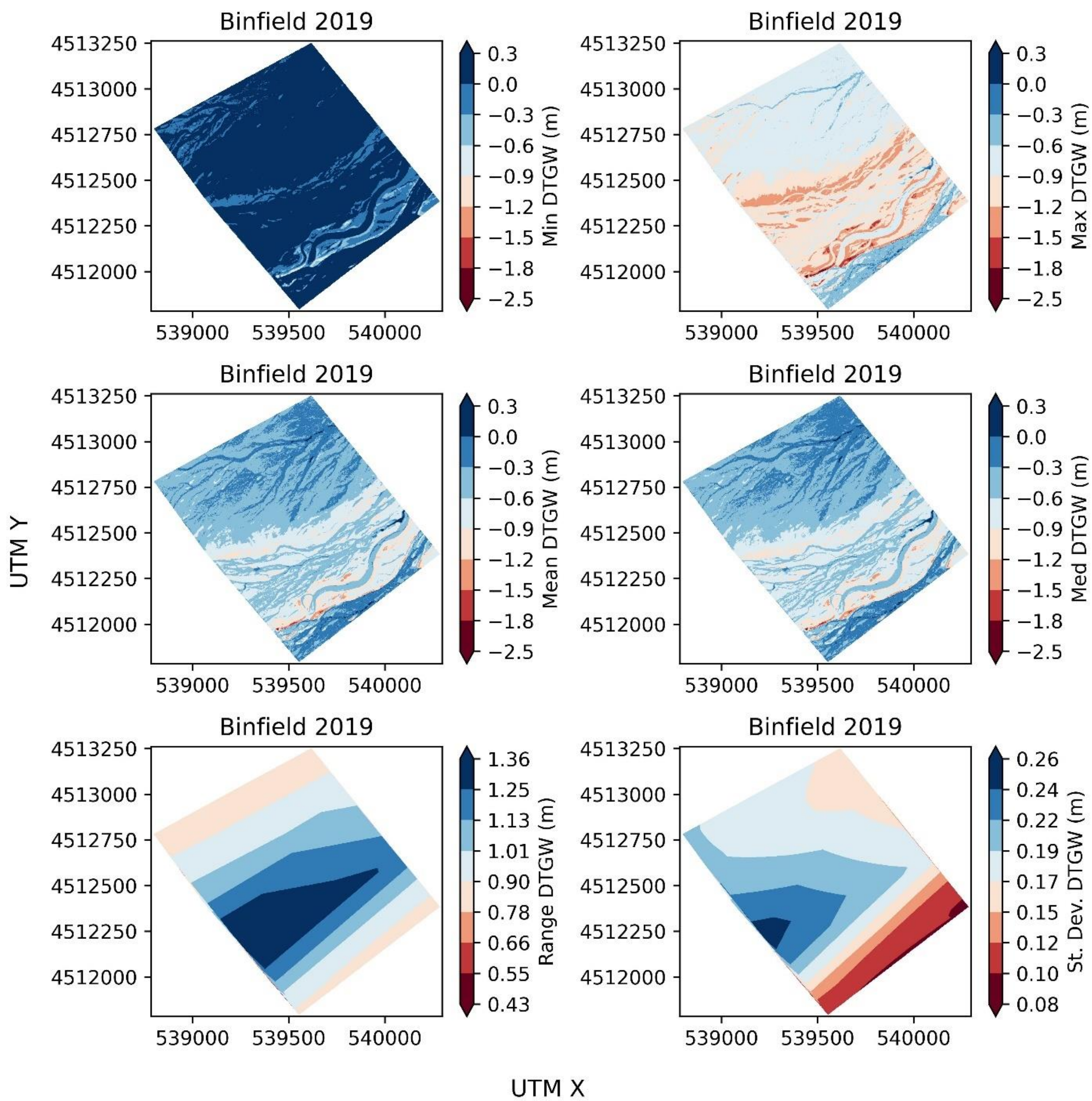

Binfield 2016
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.22
1.14
1.05
0.96
0.88
0.79
0.70
0.62
0.53
0.30
0.28
0.25
0.23
0.21
0.19
0.17
0.15
0.12
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
UTM Y
UTM X


Binfield 2017
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.11
1.02
0.92
0.82
0.72
0.62
0.52
0.42
0.32
0.24
0.22
0.20
0.18
0.16
0.14
0.12
0.10
0.07
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
UTM Y
UTM X

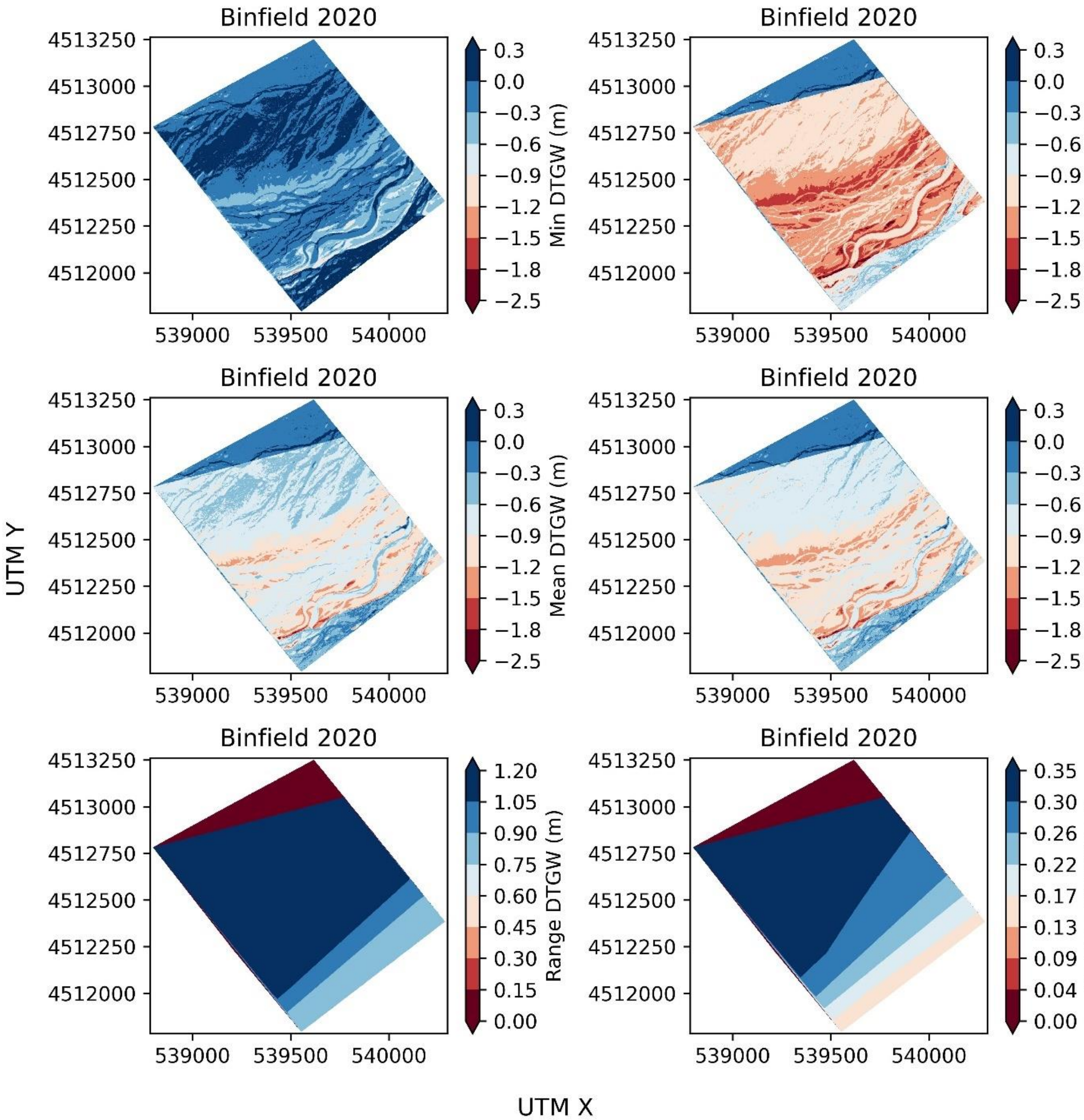


Binfield 2018
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.10
1.01
0.91
0.82
0.72
0.63
0.53
0.44
0.34
0.26
0.23
0.21
0.19
0.16
0.14
0.12
0.09
0.07
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
UTM Y
UTM X

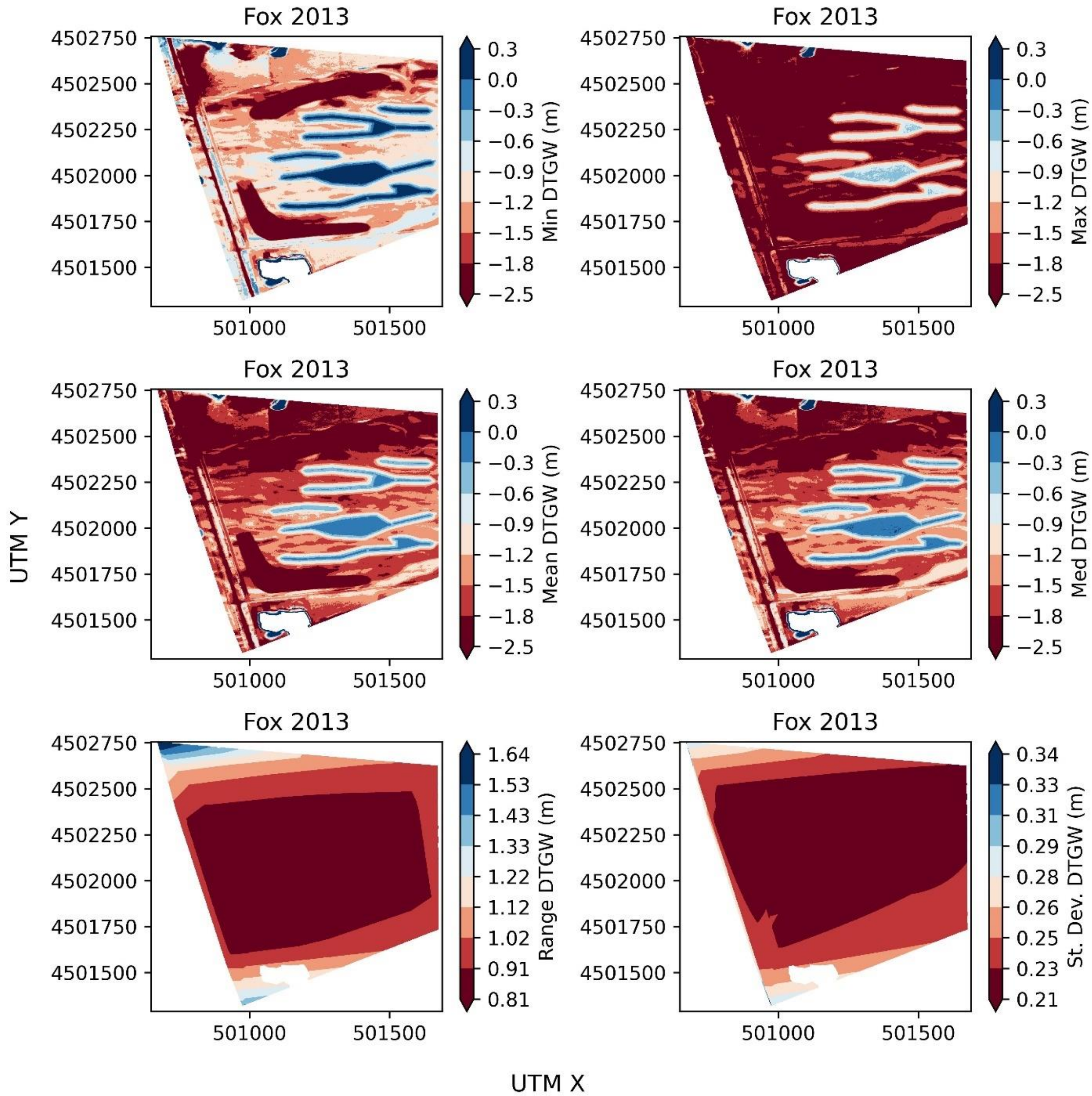
7


Binfield 2019
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.36
1.25
1.13
1.01
0.90
0.78
0.66
0.55
0.43
0.26
0.24
0.22
0.19
0.17
0.15
0.12
0.10
0.08
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
UTM Y
UTM X

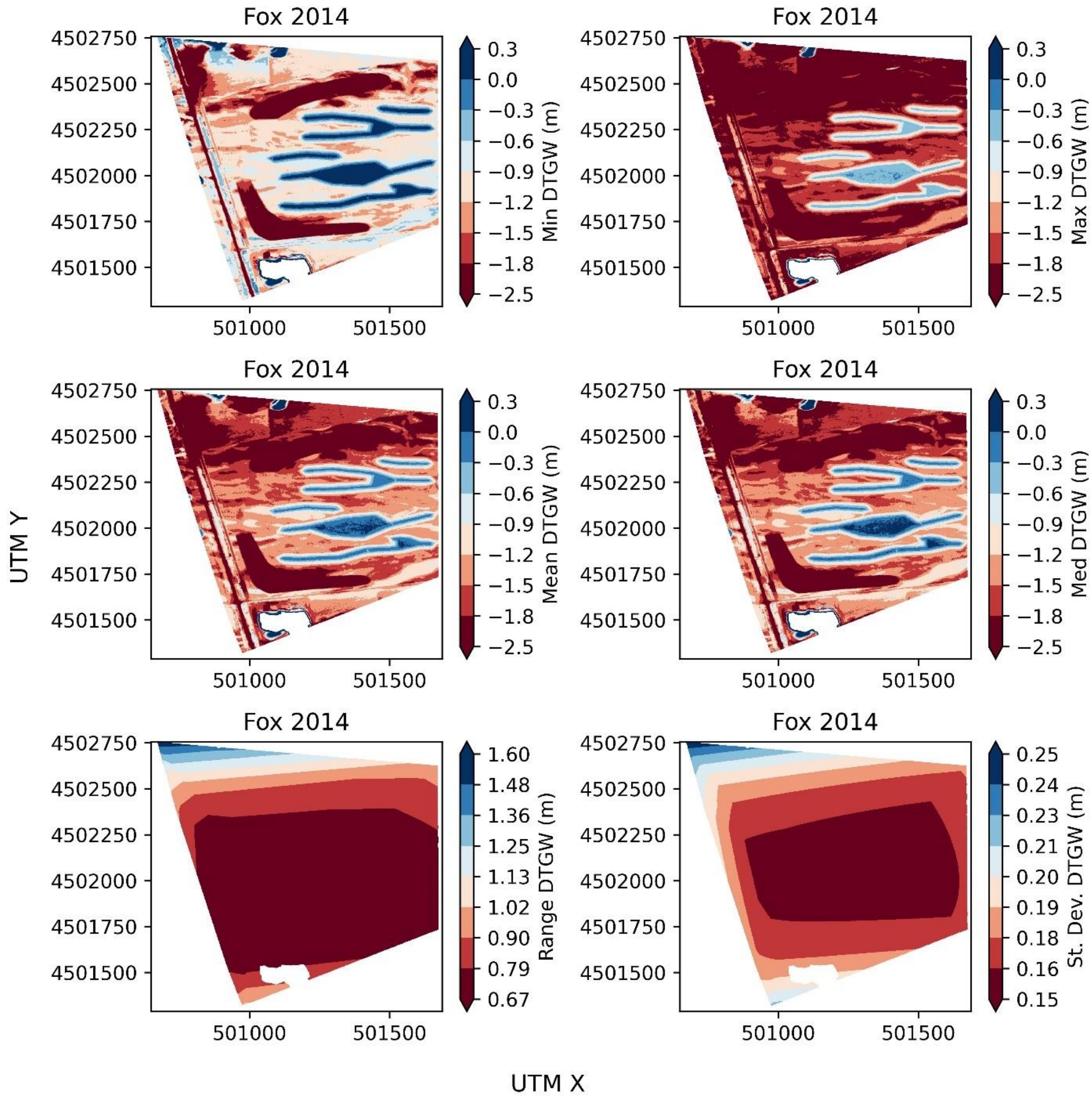
8


Binfield 2020
Binfield 2020
Binfield 2020
Binfield 2020
Binfield 2020
Binfield 2020
UTM Y
UTM X
4513250
4513000
4512750
4512500
4512250
4512000
539000
539500
540000
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
Range DTGW (m)
1.20
1.05
0.90
0.75
0.60
0.45
0.30
0.15
0.00
St. Dev. DTGW (m)
0.35
0.30
0.26
0.22
0.17
0.13
0.09
0.04
0.00

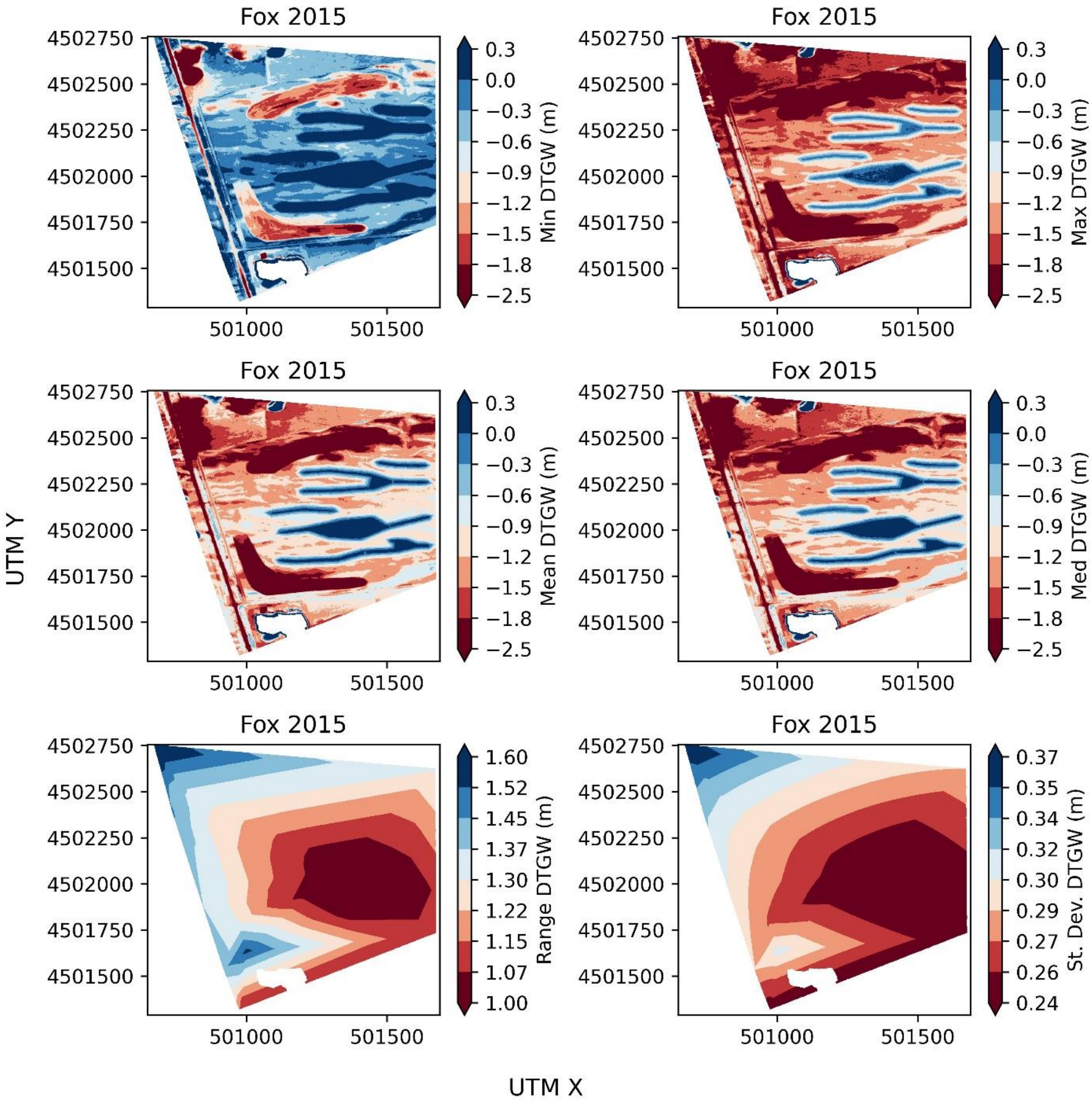
9


Fox 2013
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.64
1.53
1.43
1.33
1.22
1.12
1.02
0.91
0.81
0.34
0.33
0.31
0.29
0.28
0.26
0.25
0.23
0.21
4502750
4502500
4502250
4502000
4501750
4501500
501000
501500
UTM Y
UTM X

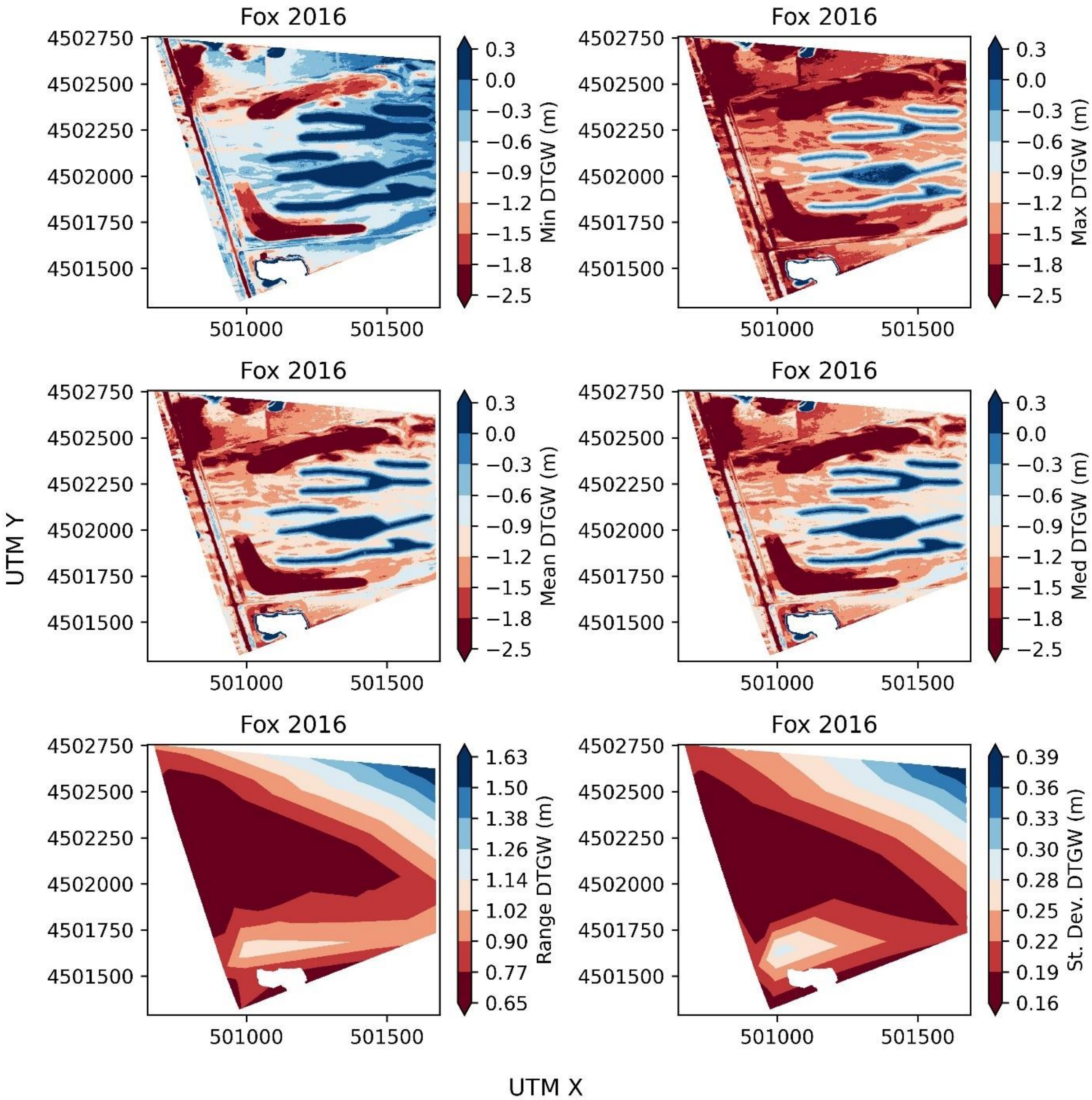
10


Fox 2014
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.60
1.48
1.36
1.25
1.13
1.02
0.90
0.79
0.67
0.25
0.24
0.23
0.21
0.20
0.19
0.18
0.16
0.15
4502750
4502500
4502250
4502000
4501750
4501500
501000
501500
UTM Y
UTM X

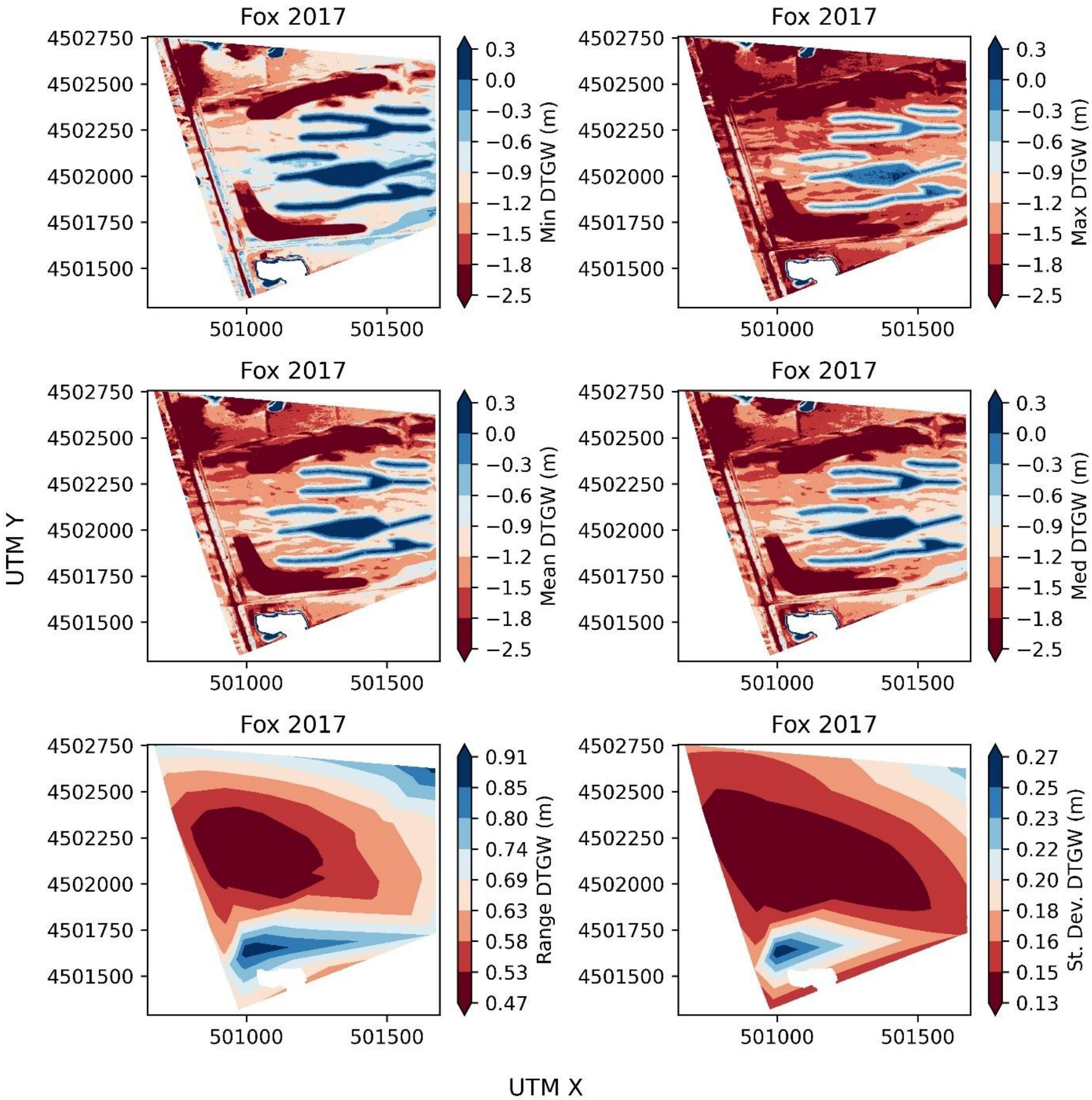
11


Fox 2015
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
UTM Y
UTM X
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.60
1.52
1.45
1.37
1.30
1.22
1.15
1.07
1.00
0.37
0.35
0.34
0.32
0.30
0.29
0.27
0.26
0.24
4502750
4502500
4502250
4502000
4501750
4501500
501000
501500

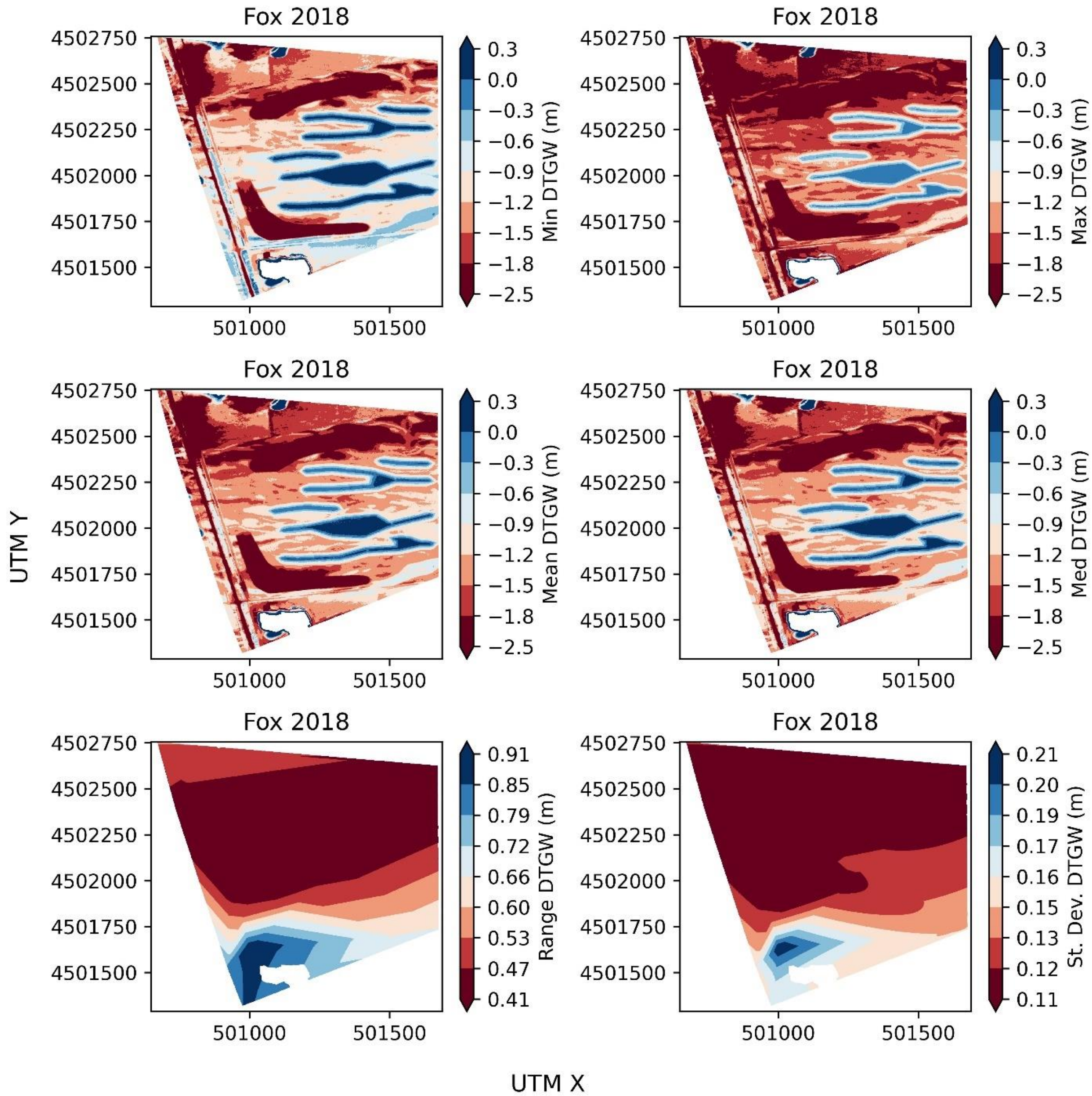
12


Fox 2016
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
Range DTGW (m)
1.63
1.50
1.38
1.26
1.14
1.02
0.90
0.77
0.65
St. Dev. DTGW (m)
0.39
0.36
0.33
0.30
0.28
0.25
0.22
0.19
0.16
UTM Y
4502750
4502500
4502250
4502000
4501750
4501500
UTM X
501000
501500

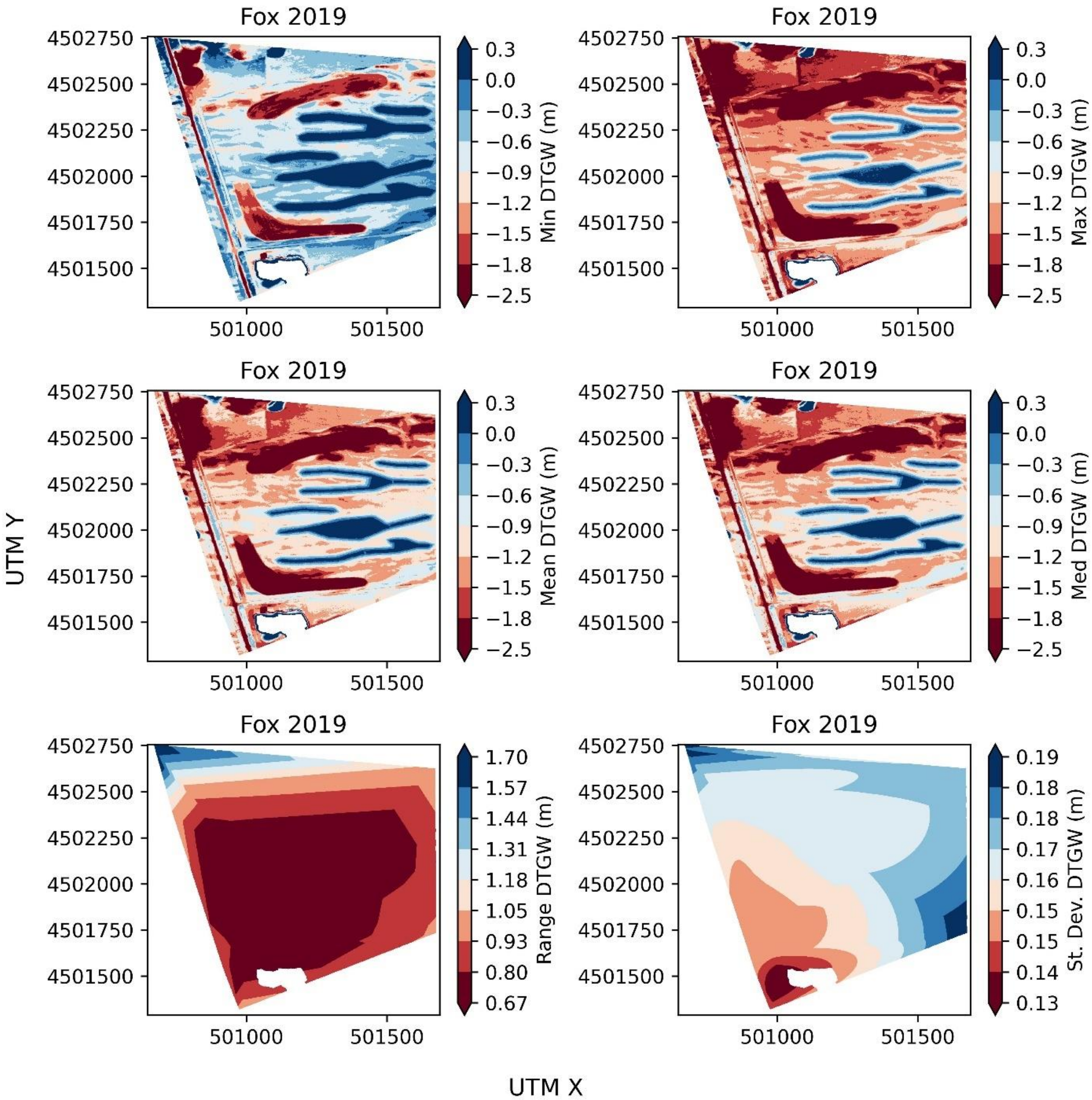
13


Fox 2017
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
0.91
0.85
0.80
0.74
0.69
0.63
0.58
0.53
0.47
0.27
0.25
0.23
0.22
0.20
0.18
0.16
0.15
0.13
4502750
4502500
4502250
4502000
4501750
4501500
501000
501500
UTM Y
UTM X

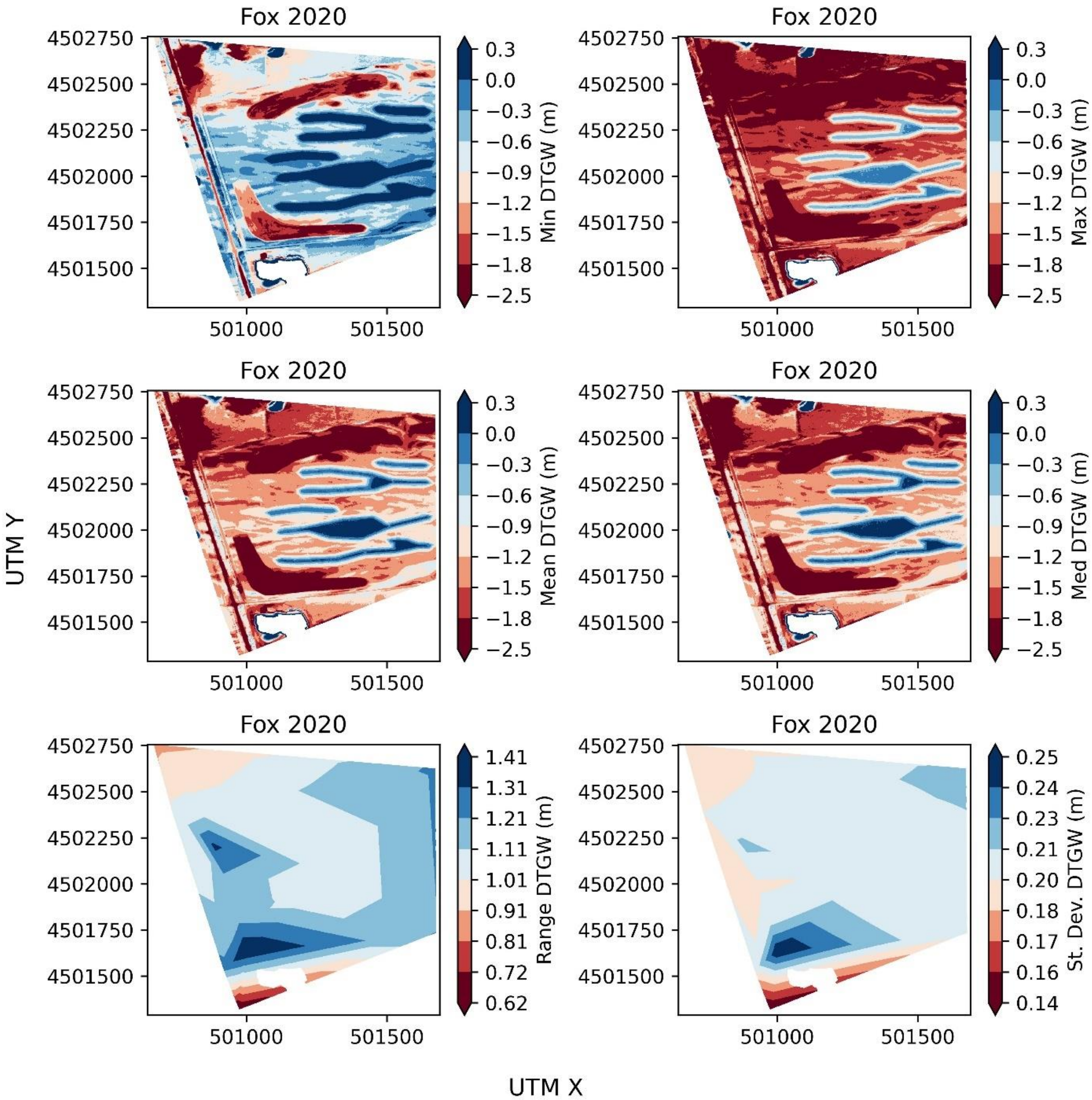
14


Fox 2018
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3 0.0 −0.3 −0.6 −0.9 −1.2 −1.5 −1.8 −2.5
0.91 0.85 0.79 0.72 0.66 0.60 0.53 0.47 0.41
0.21 0.20 0.19 0.17 0.16 0.15 0.13 0.12 0.11
4502750 4502500 4502250 4502000 4501750 4501500
501000 501500
UTM Y
UTM X

15


Fox 2019
Fox 2019
Fox 2019
Fox 2019
Fox 2019
Fox 2019
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
UTM Y
UTM X
4502750
4502500
4502250
4502000
4501750
4501500
501000
501500
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.70
1.57
1.44
1.31
1.18
1.05
0.93
0.80
0.67
0.19
0.18
0.18
0.17
0.16
0.15
0.15
0.14
0.13

16


Fox 2020
Min DTGW (m)
Max DTGW (m)
Mean DTGW (m)
Med DTGW (m)
Range DTGW (m)
St. Dev. DTGW (m)
0.3
0.0
−0.3
−0.6
−0.9
−1.2
−1.5
−1.8
−2.5
1.41
1.31
1.21
1.11
1.01
0.91
0.81
0.72
0.62
0.25
0.24
0.23
0.21
0.20
0.18
0.17
0.16
0.14
4502750
4502500
4502250
4502000
4501750
4501500
501000
501500
UTM Y
UTM X

17

Appendix E – Ground surface – groundwater elevation relationships

DTGW distributions were tested for normality by plotting histograms and performing Shapiro Wilk tests for annual and daily subsets of DTGW values. Annual subsets included yearly sets of DTGW values for individual wells (i.e., to test temporal distributions) and daily subsets included daily gridded DTGW raster surfaces (to test spatial distributions). Example histogram results are shown in Figure A1. Plots of distributions showed outliers, multiple modes and skewness, and p-values for all statistical tests were less than 0.05, indicating non-normal distributions. Resistant statistical metrics (e.g., median, interquartile range, median absolute deviation) are therefore most appropriate for summarizing wet meadow DTGW datasets. Notably, mean and median DTGW values were similar (Table 1 and Table 2).

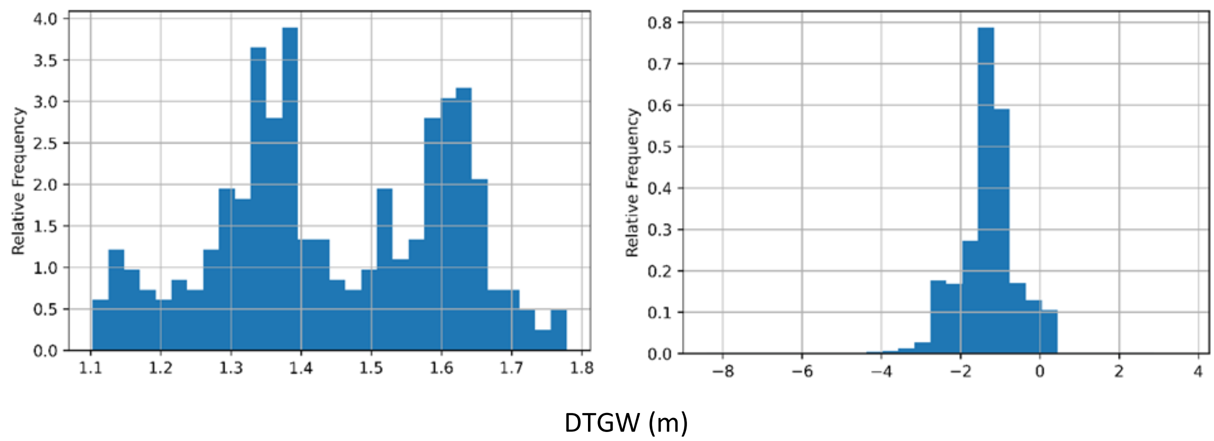


Figure A1 - Histograms exemplifying distributions for subsets of DTGW data from the Fox site. The left subset includes all hourly DTGW measurements for well 105 during 2014, and right subset includes spatially interpolated groundwater depths at the Fox site on 7/1/2020.

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 A2).

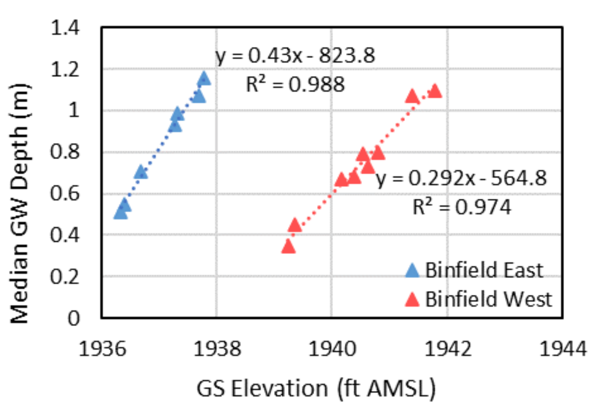


Figure A2 – Plot demonstrating strong-positive relationship between ground surface elevation and median groundwater depth (m) for Fox and Binfield transects.

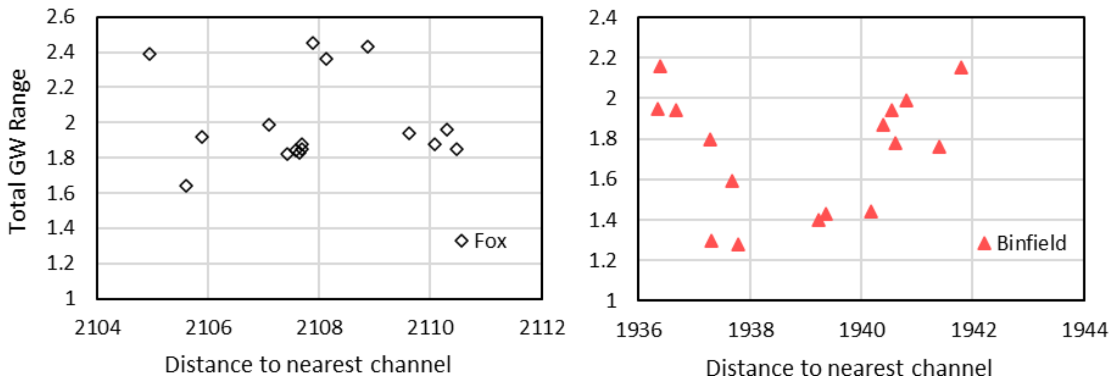


Figure A3 – Plot of distance from channel vs study period GW range.

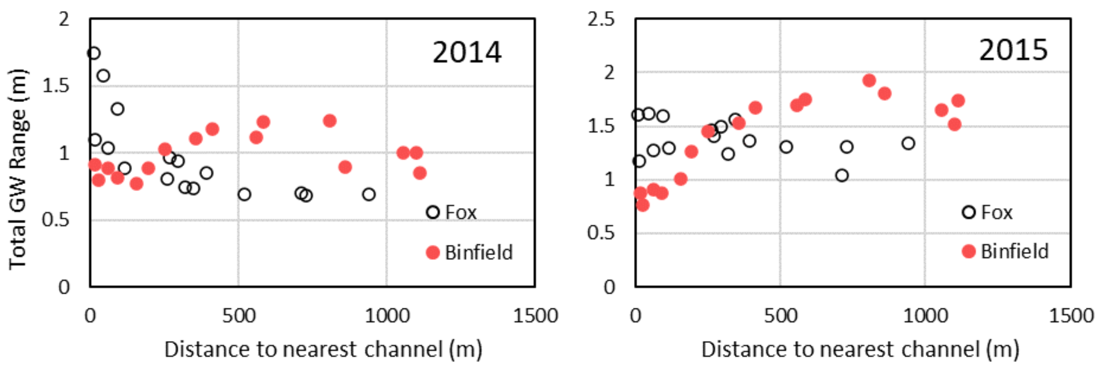


Figure A4 – Plot showing relationship between annual range in groundwater levels and the distance to the channel for a dry year (2014) and wet year (2015).

Appendix F – Grassland Survey – L7th Supplement

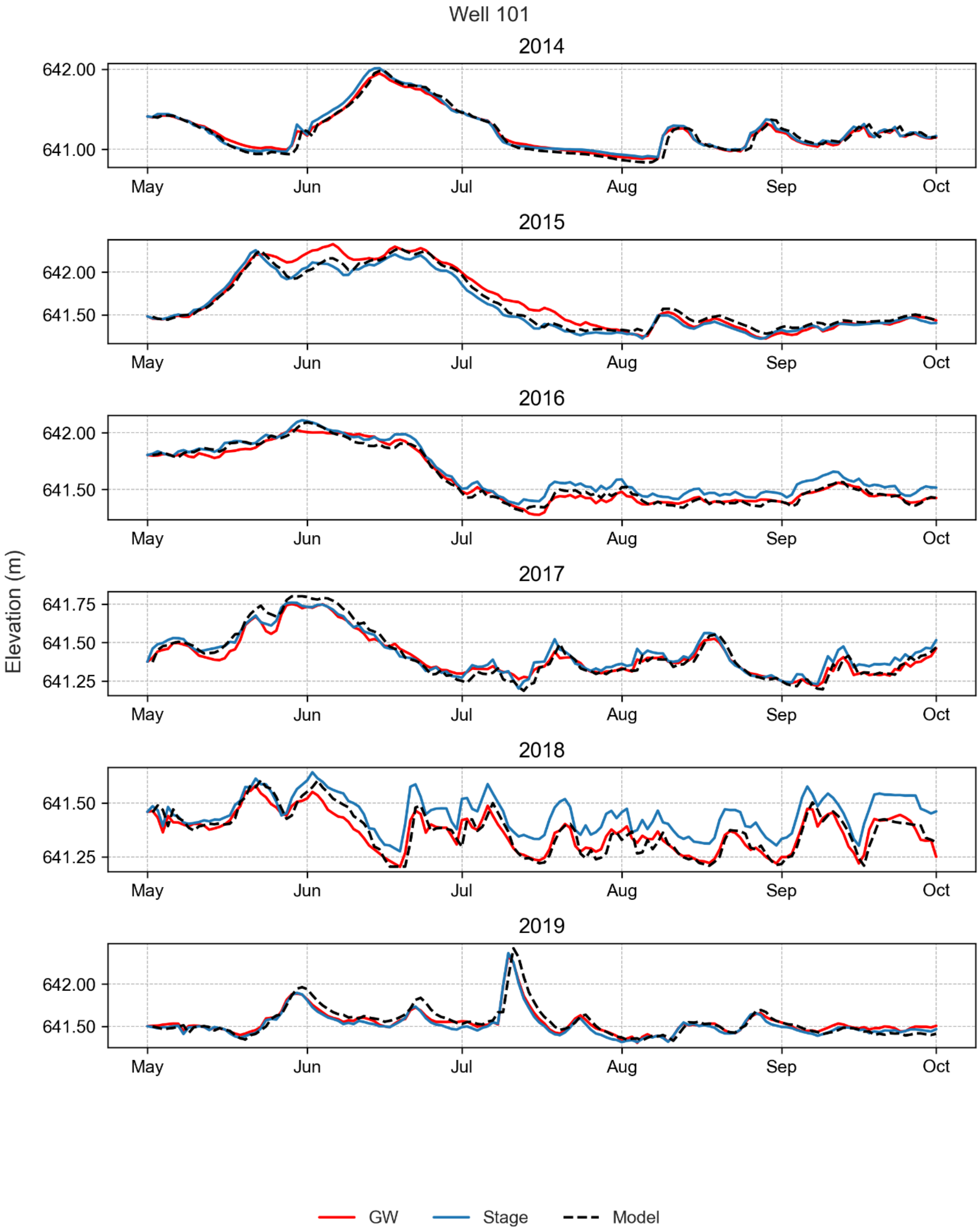
Table E1 – L7th vegetation category predictions.

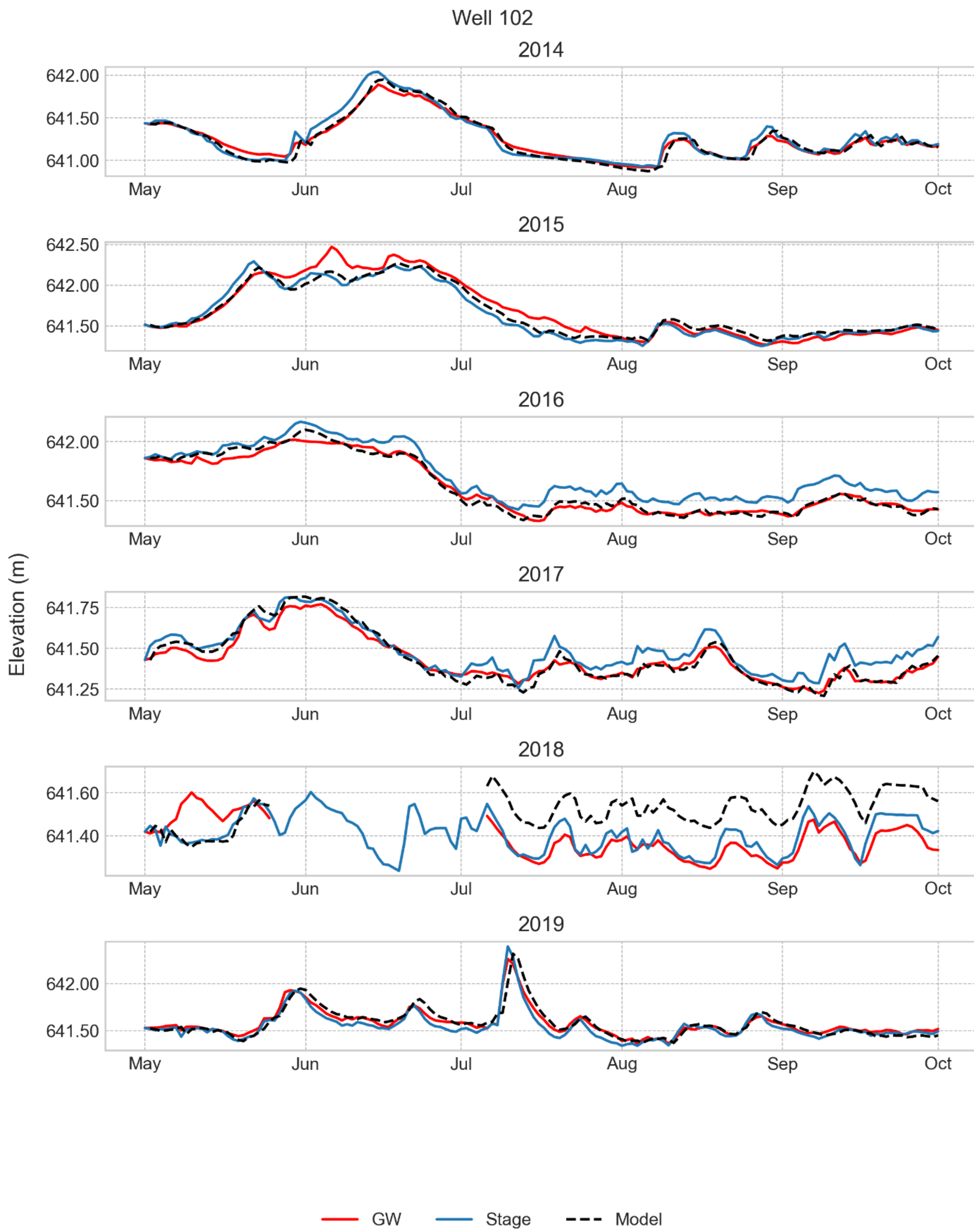
	GIS ID	Survey ID	2013	2014	2015	2016	2017	2018	2019	2020
Fox	311	177	MP	MP	SM	MP	MP	MP	MP	MP
	313	176	MP	MP	SM	MP	MP	MP	MP	MP
	315	178	DR	DR	DR	DR	DR	DR	DR	DR
	317	175	DR	DR	MP	DR	DR	DR	DR	DR
	324	174	MP/DR	MP/DR	MP	SM/MP	MP	MP/DR	MP	MP
	358	163	MP	MP	MP	MP	MP	MP	MP	MP
Binfield	199	144	SM/MP	SM/MP	SM/E	SM	SM/MP	SM/MP	SM	SM/MP
	205	141	MP	MP	SM/MP	SM/MP	SM/MP	MP	SM	SM/MP
	215	138	SM	SM	SM	SM	SM	SM	SM	SM
	207	140	MP	MP	SM/MP	MP	MP	MP	SM/MP	MP
	213	137	SM	SM	SM	SM	SM	SM	SM	SM
	217	135	MP	SM	SM	SM	SM	SM	SM	NA
	219	134	MP	SM	SM	SM	SM	SM	SM	NA
	211	136	SM	SM	SM	SM	SM	SM	SM	SM
	209	139	SM/MP	SM/MP	SM/E	SM	SM	SM/MP	SM	SM/MP
	197	142	MP	MP	SM/MP	MP	MP	MP	MP	MP

Table E2 – Percent of grassland survey vegetation in each wetland indicator status category.

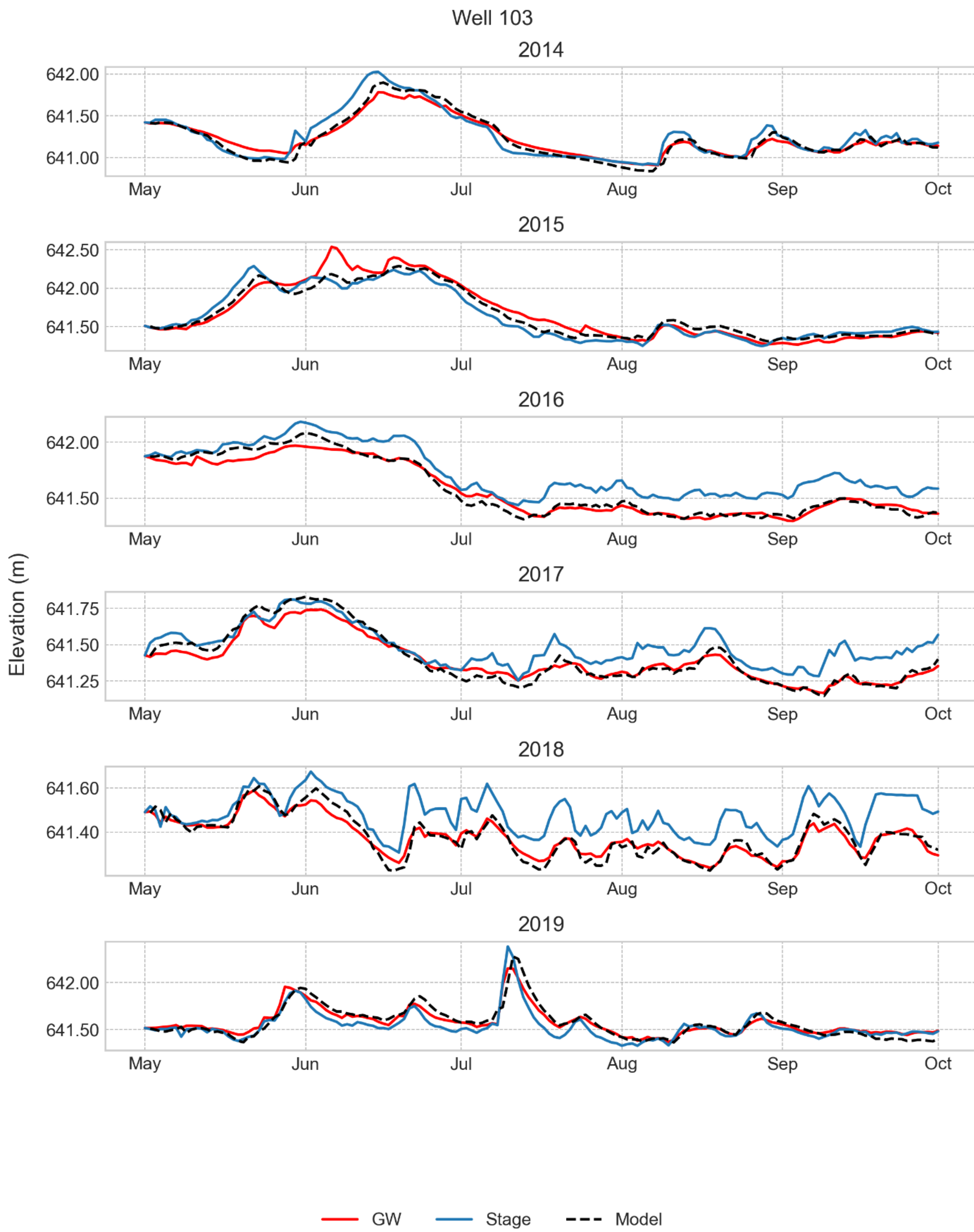
	Site #		2016					2019					2022				
	GIS	*Survey*	*OBL*	*FACW*	*FAC*	*FACU*	*UPL*	*OBL*	*FACW*	*FAC*	*FACU*	*UPL*	*OBL*	*FACW*	*FAC*	*FACU*	*UPL*
Fox	311	177	No Data for Survey at Fox 2016					0	0	23	77	0	0	0	5	95	0
	313	176						0	0	16	84	0	0	0	0	98	2
	315	178						0	0	11	78	11	0	0	0	87	13
	317	175						0	0	8	55	38	0	0	29	71	0
	324	174						4	0	28	67	1	0	0	35	64	1
	358	163						0	6	0	65	29	0	0	1	75	24
Binfield	199	144	4	19	15	59	3	17	18	25	33	7	14	22	18	44	2
	205	141	0	35	28	33	4	9	18	28	42	4	10	21	23	45	2
	215	138	0	41	18	23	19	3	56	34	6	0	11	51	27	12	0
	207	140	0	20	18	47	15	3	23	20	51	3	0	7	22	71	0
	213	137	4	24	23	39	10	14	36	23	28	0	17	33	34	16	0
	217	135	3	34	22	30	11	32	23	22	23	0	17	40	29	13	0
	219	134	1	25	22	40	11	44	22	29	5	0	26	25	42	7	0
	211	136	0	43	16	27	13	30	32	21	17	1	15	36	25	24	0
	209	139	0	45	13	41	2	0	29	22	37	12	0	26	8	66	0
	197	142	6	38	13	38	4	3	24	24	46	3	7	24	13	56	0

Appendix G – Model calibration results

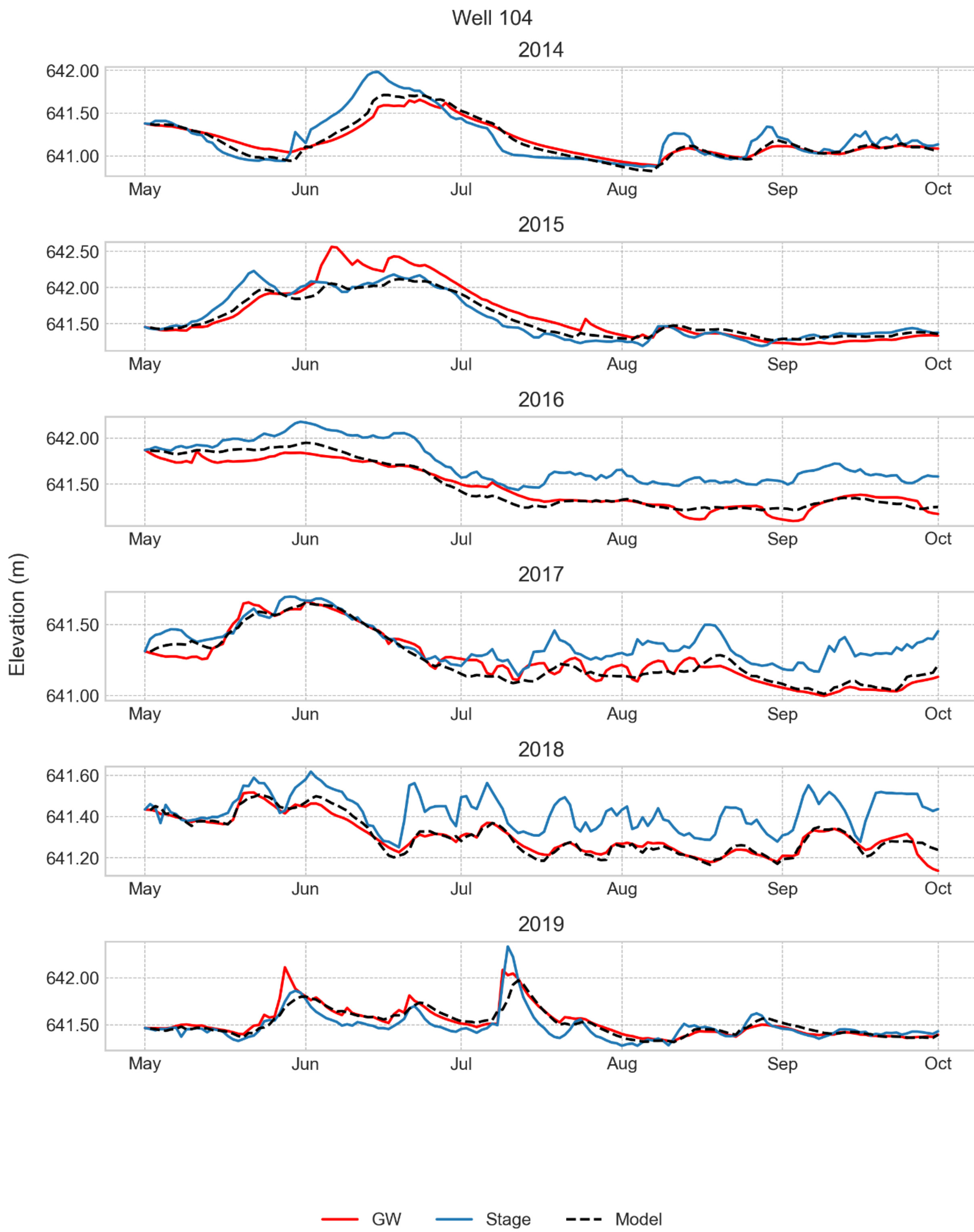



Well 102
2014
642.00
641.50
641.00
May
Jun
Jul
Aug
Sep
Oct
2015
642.50
642.00
641.50
2016
642.00
641.50
2017
641.75
641.50
641.25
2018
641.60
641.40
2019
642.00
641.50
Elevation (m)
GW
Stage
Model

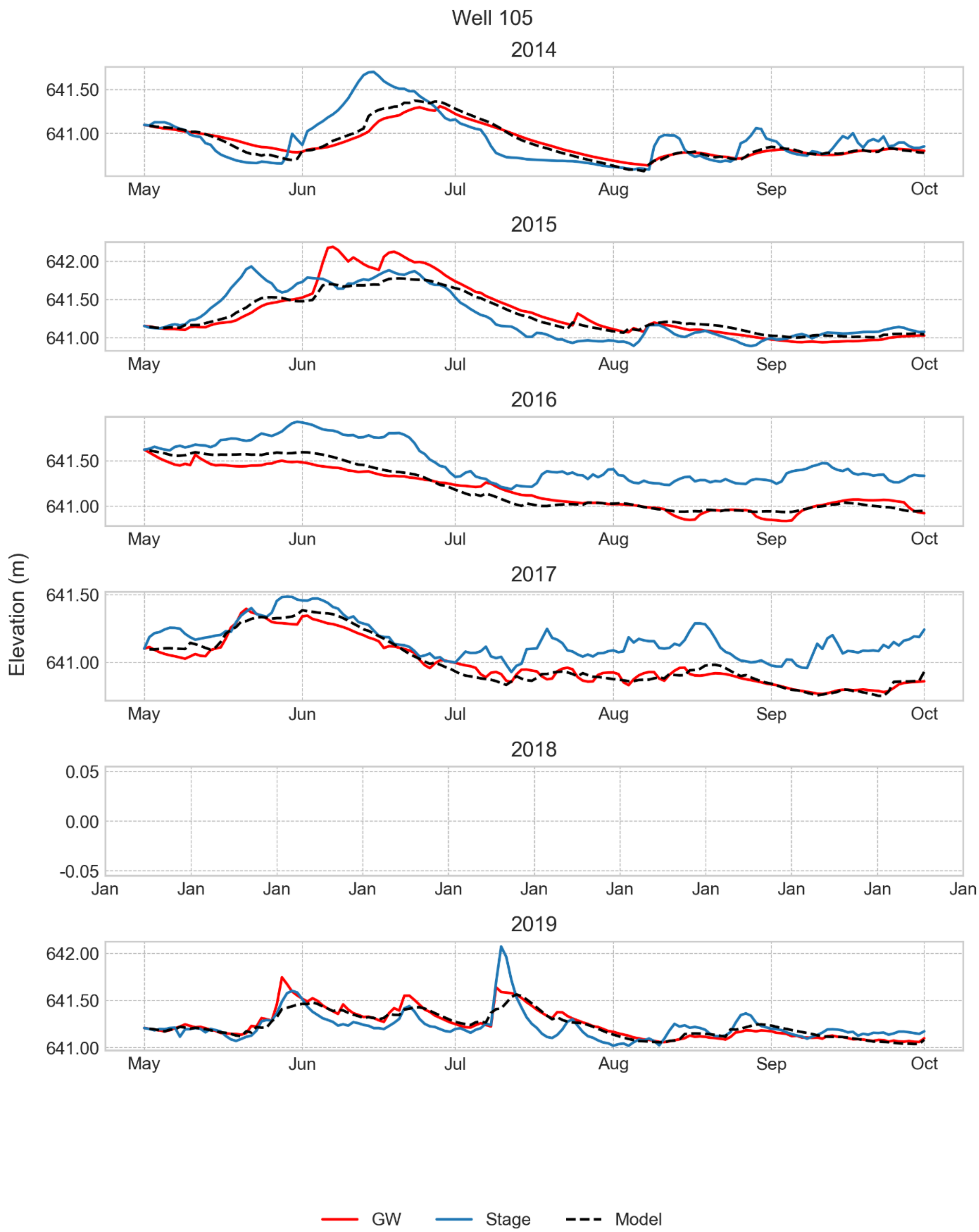
7


Well 103
2014
2015
2016
2017
2018
2019
Elevation (m)
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

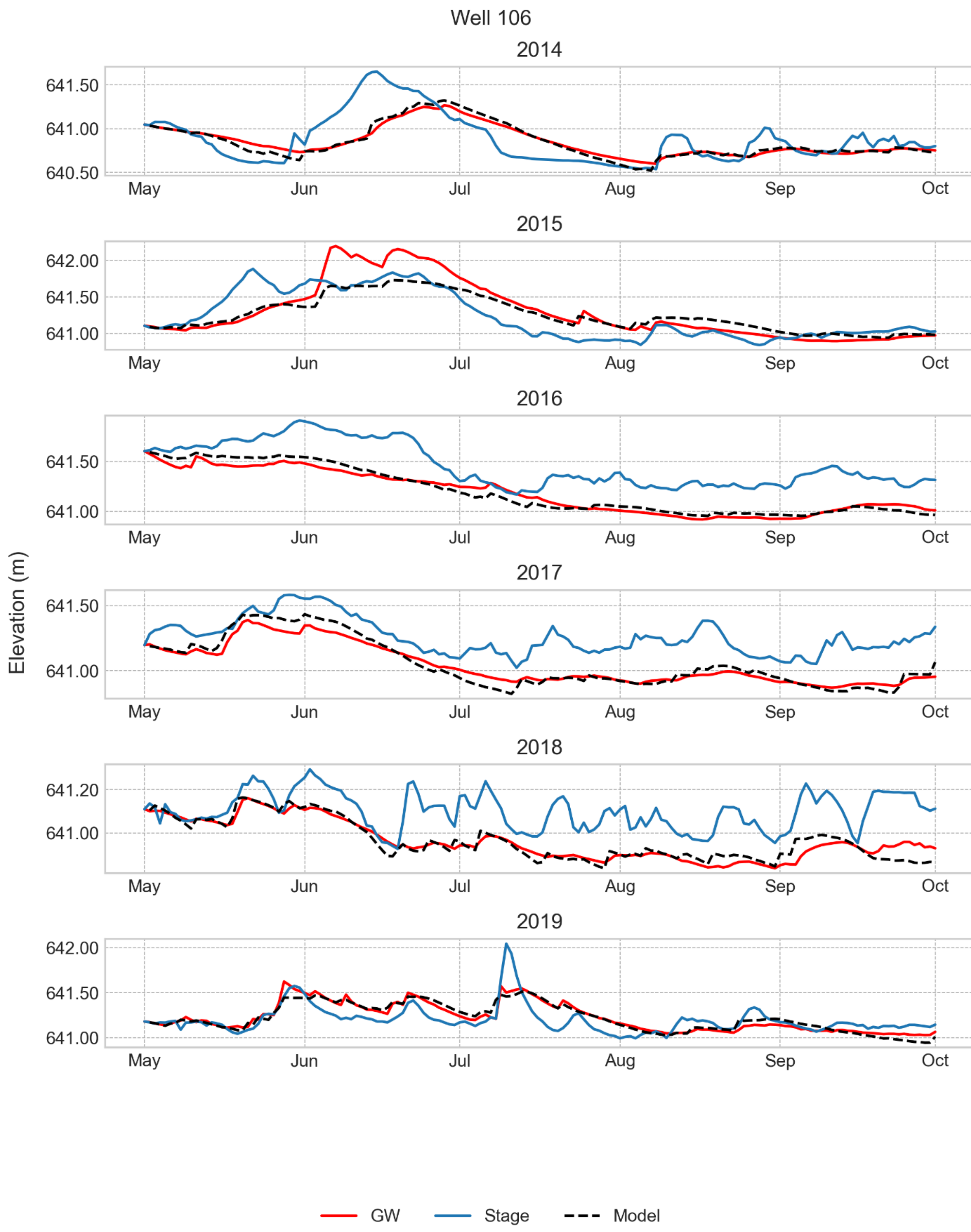
8


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

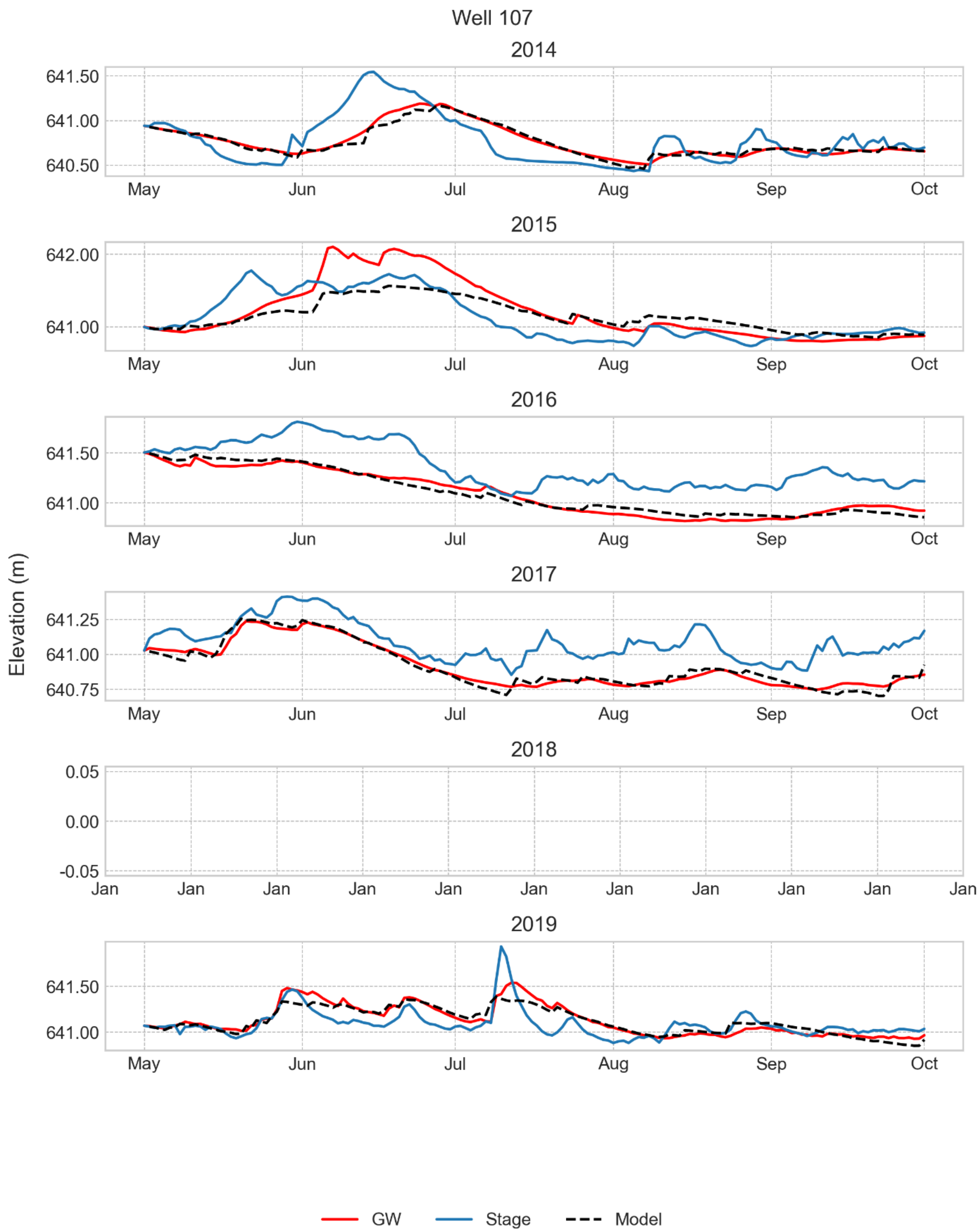
9


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

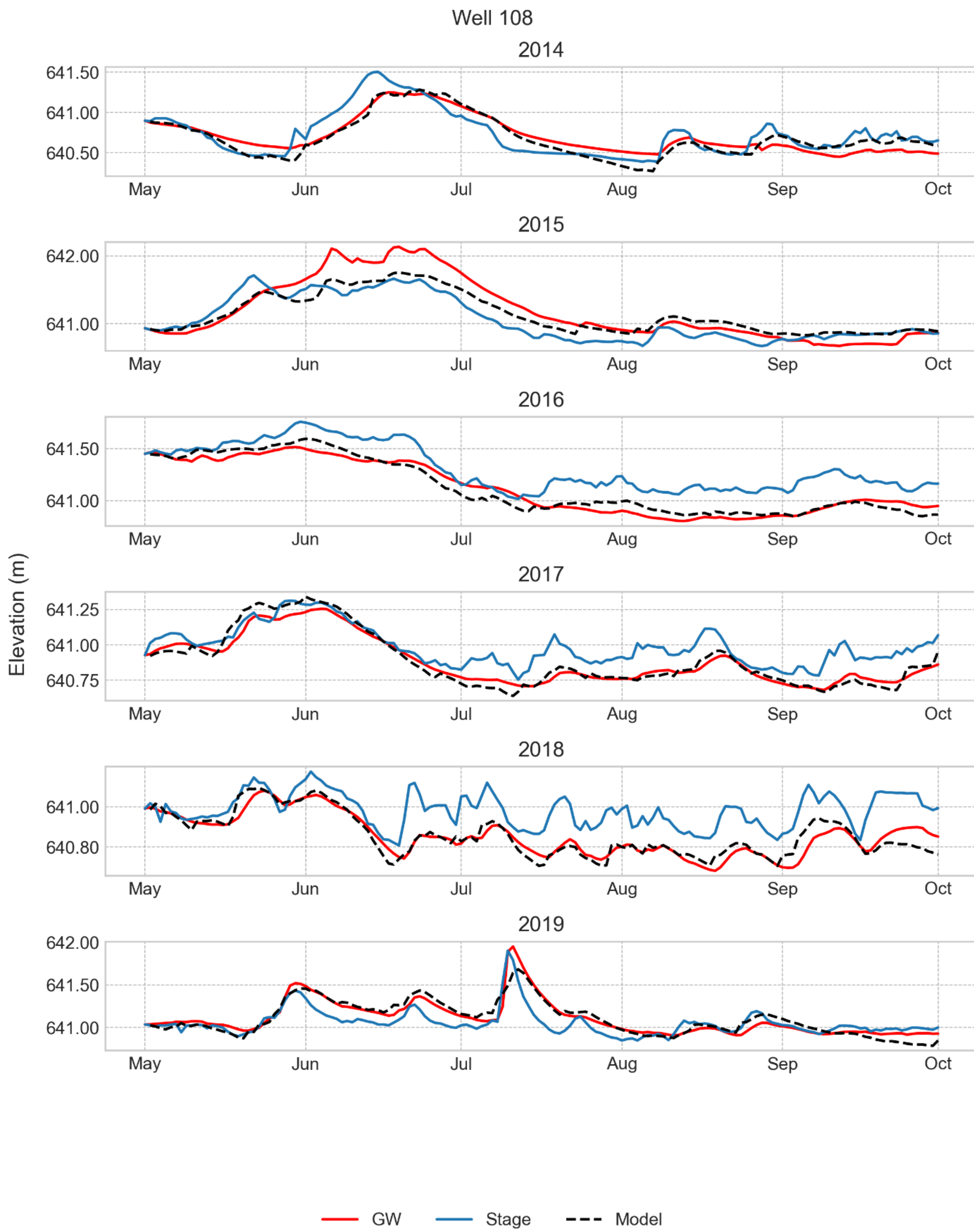
10


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

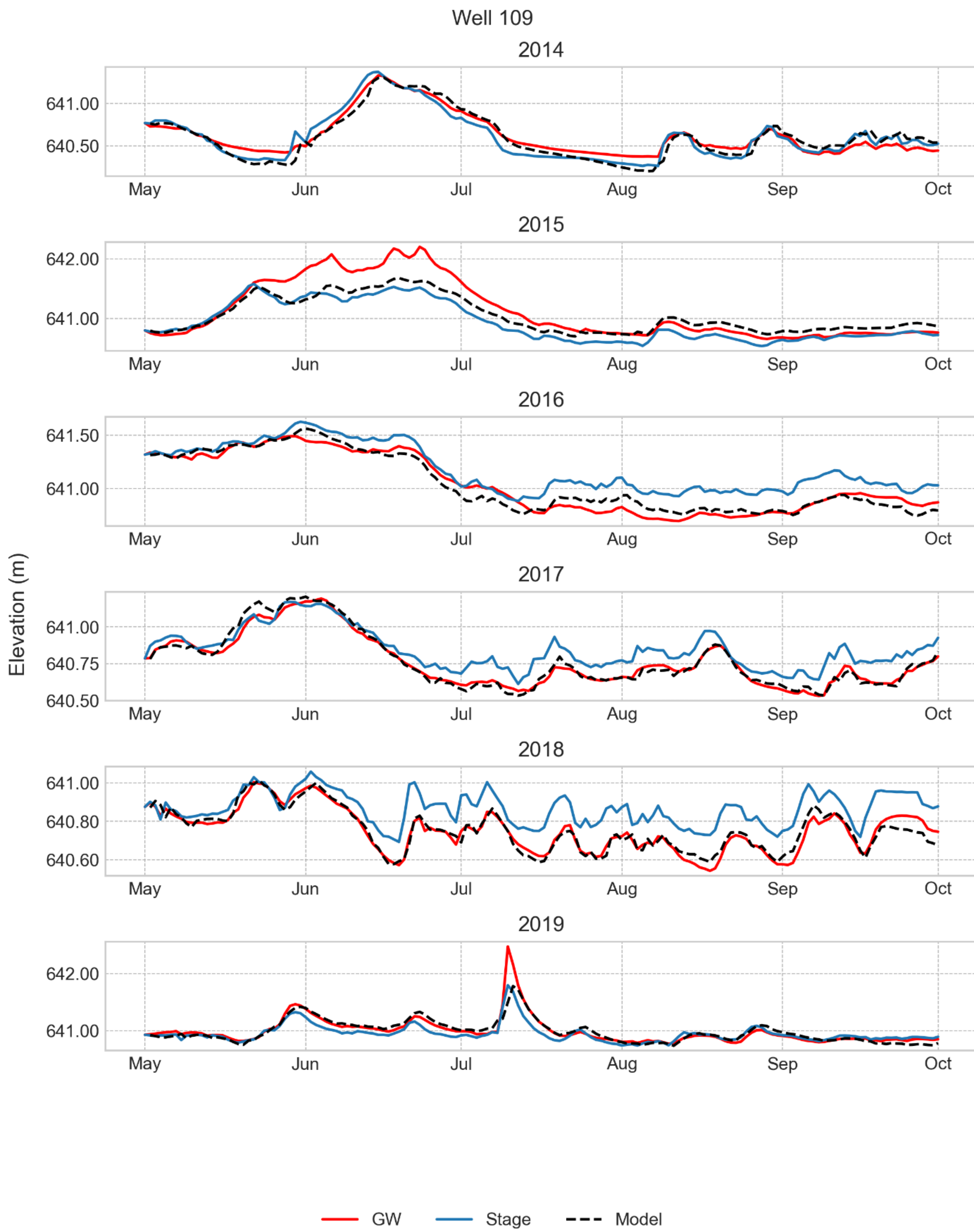
11


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

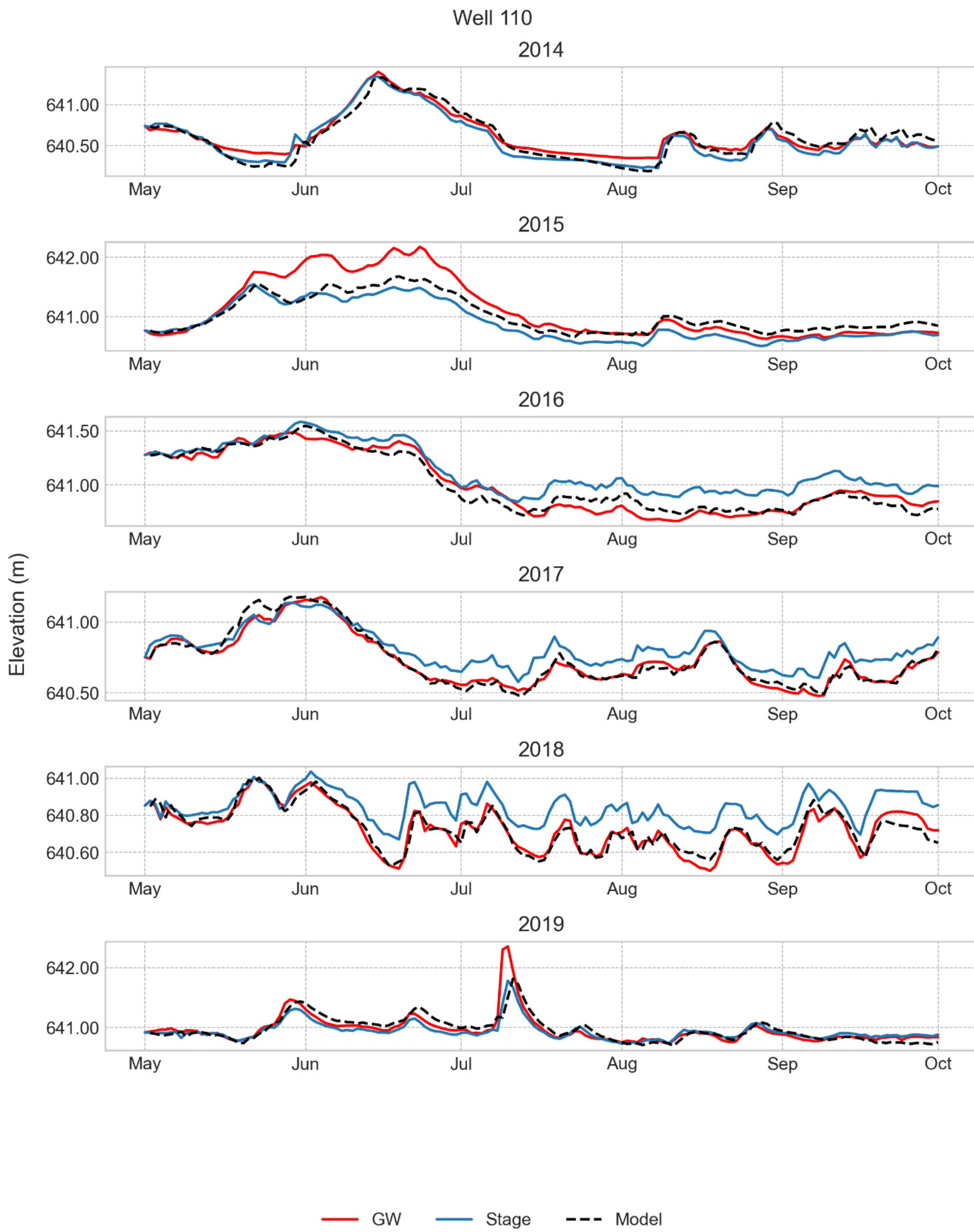
12


Well 108
2014
2015
2016
2017
2018
2019
Elevation (m)
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

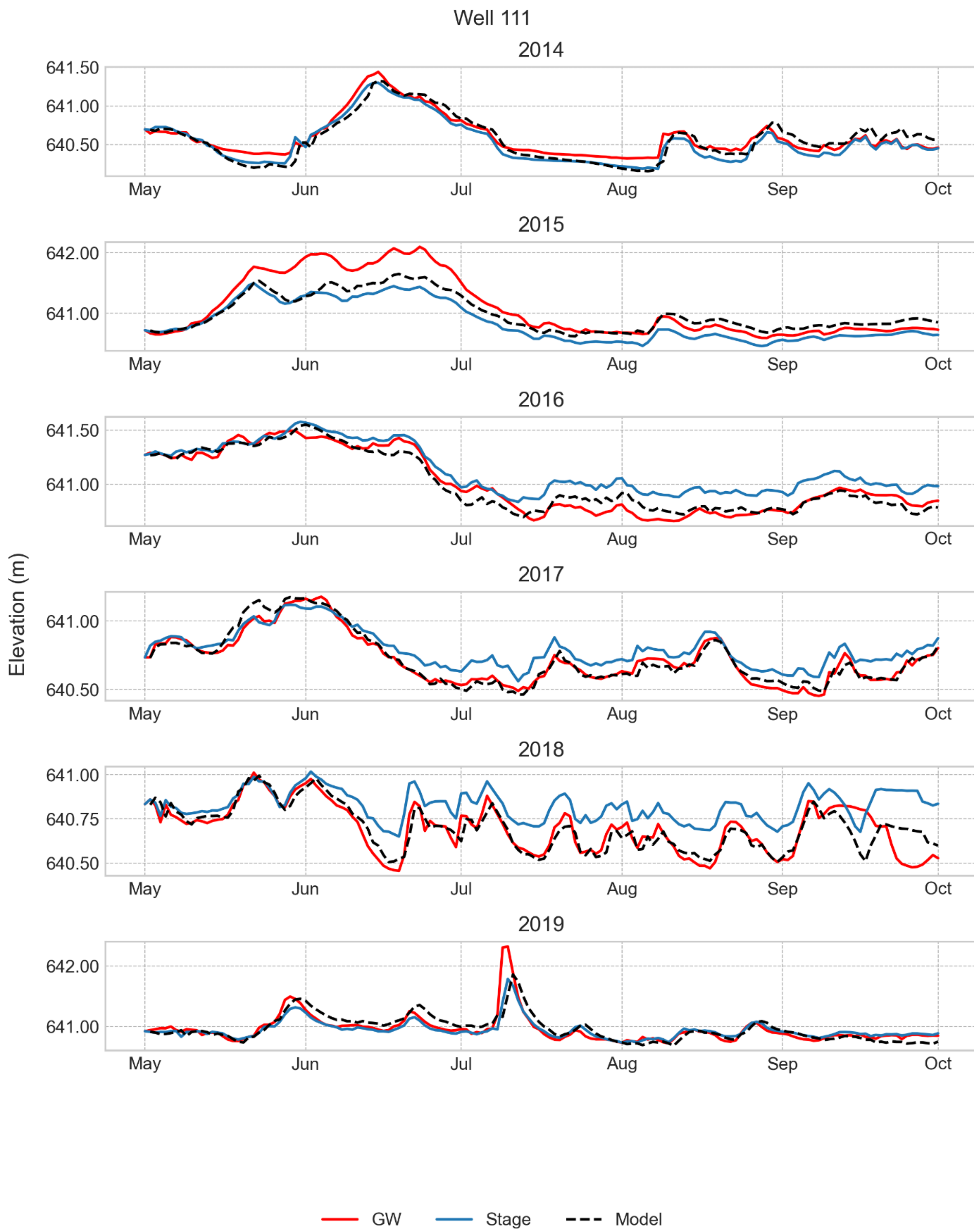
13


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

14

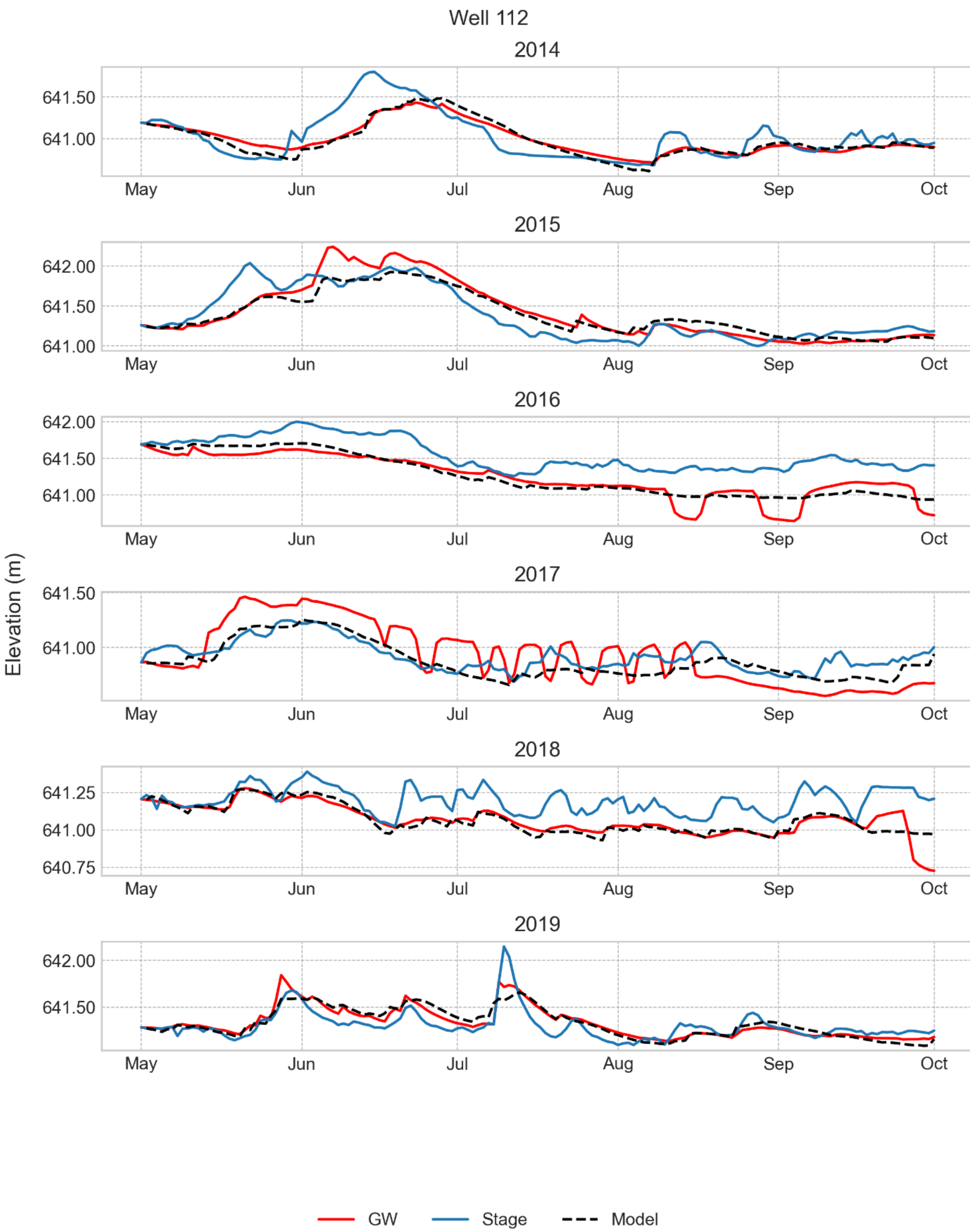

Well 110
2014
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.00
640.50
2018
641.00
640.80
640.60
2019
642.00
641.00
Elevation (m)
GW
Stage
Model

15

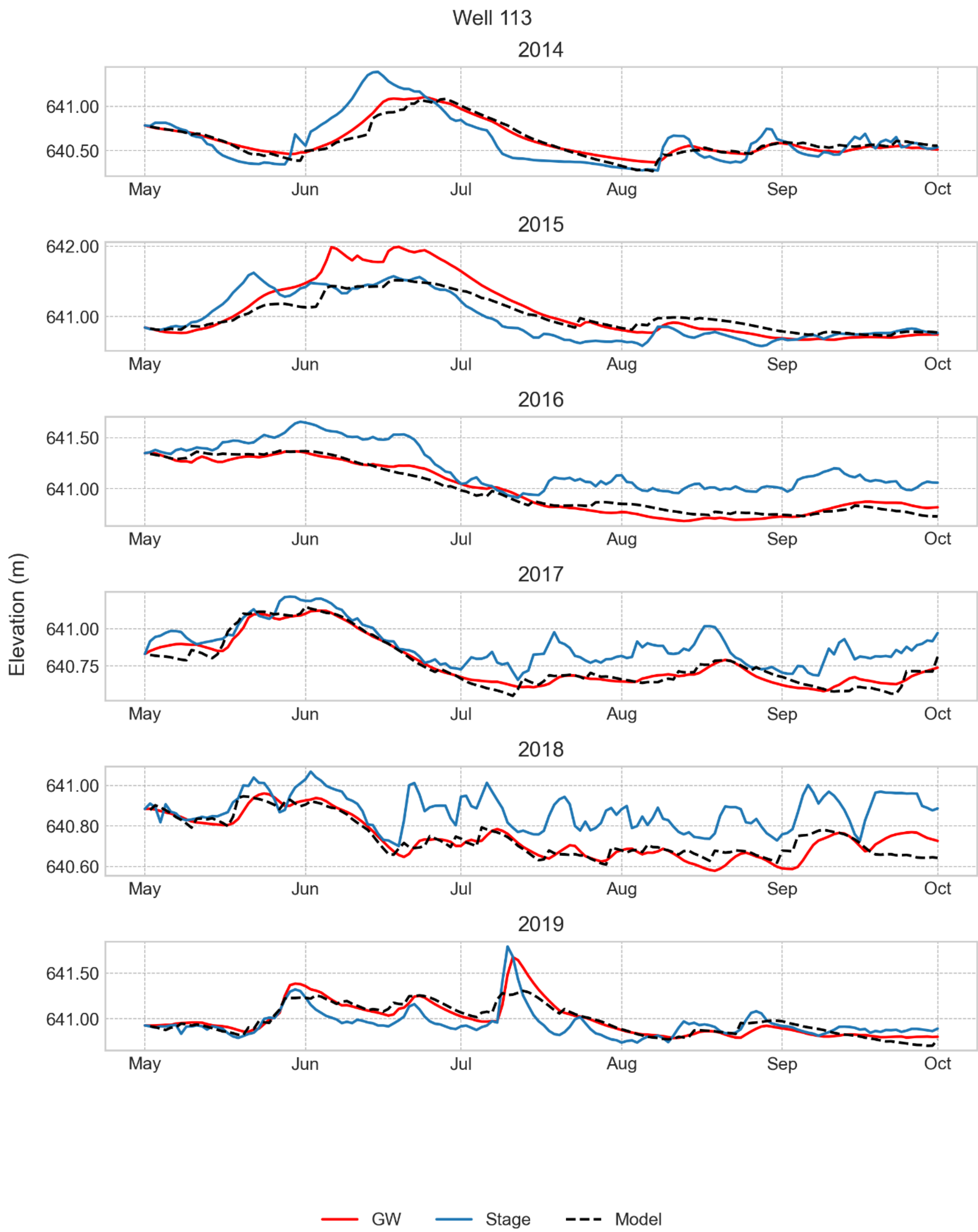


Well 111
2014
641.50
641.00
640.50
May
Jun
Jul
Aug
Sep
Oct
2015
642.00
641.00
2016
641.50
641.00
2017
641.00
640.50
2018
641.00
640.75
640.50
2019
642.00
641.00
Elevation (m)
GW
Stage
Model

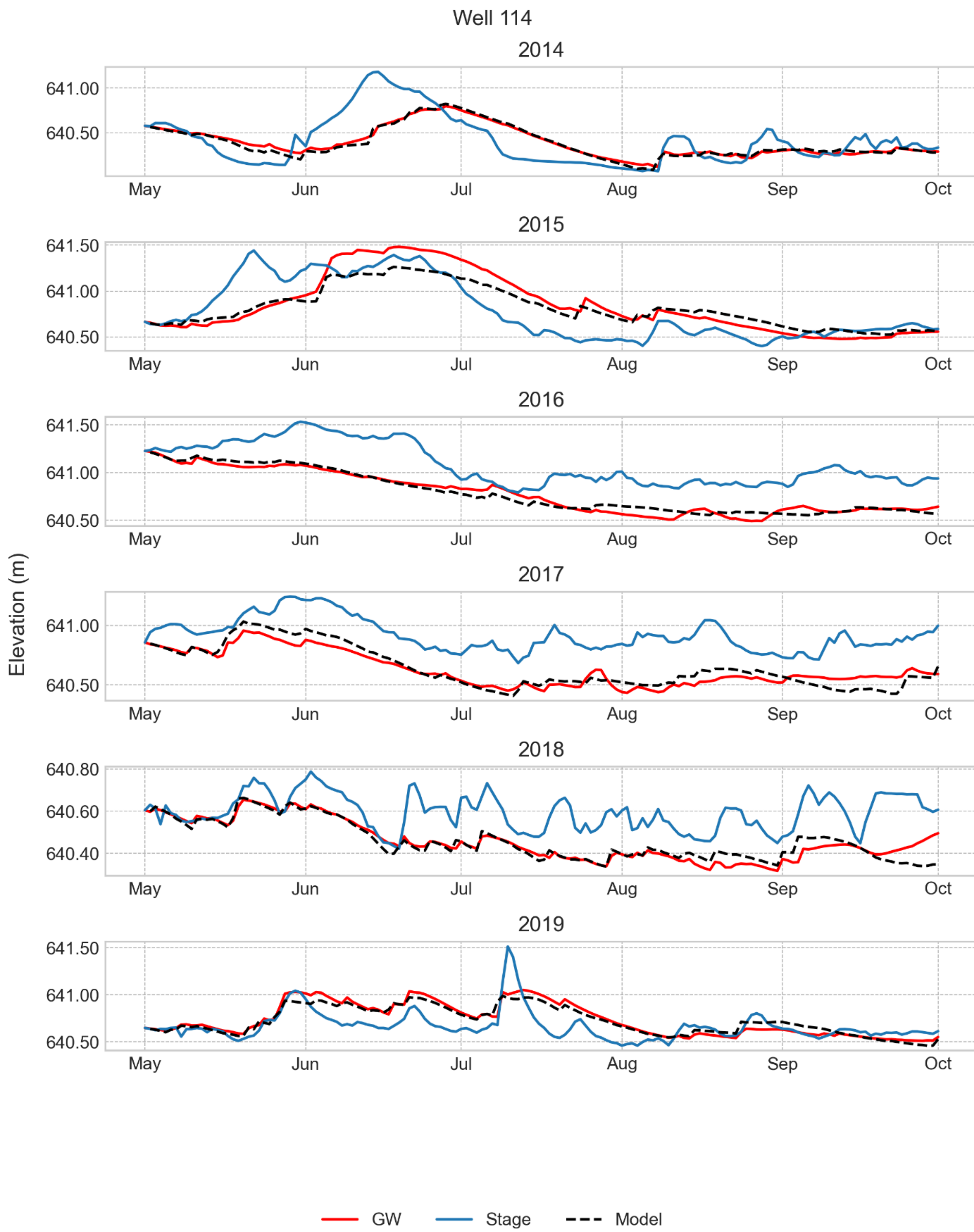
16


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

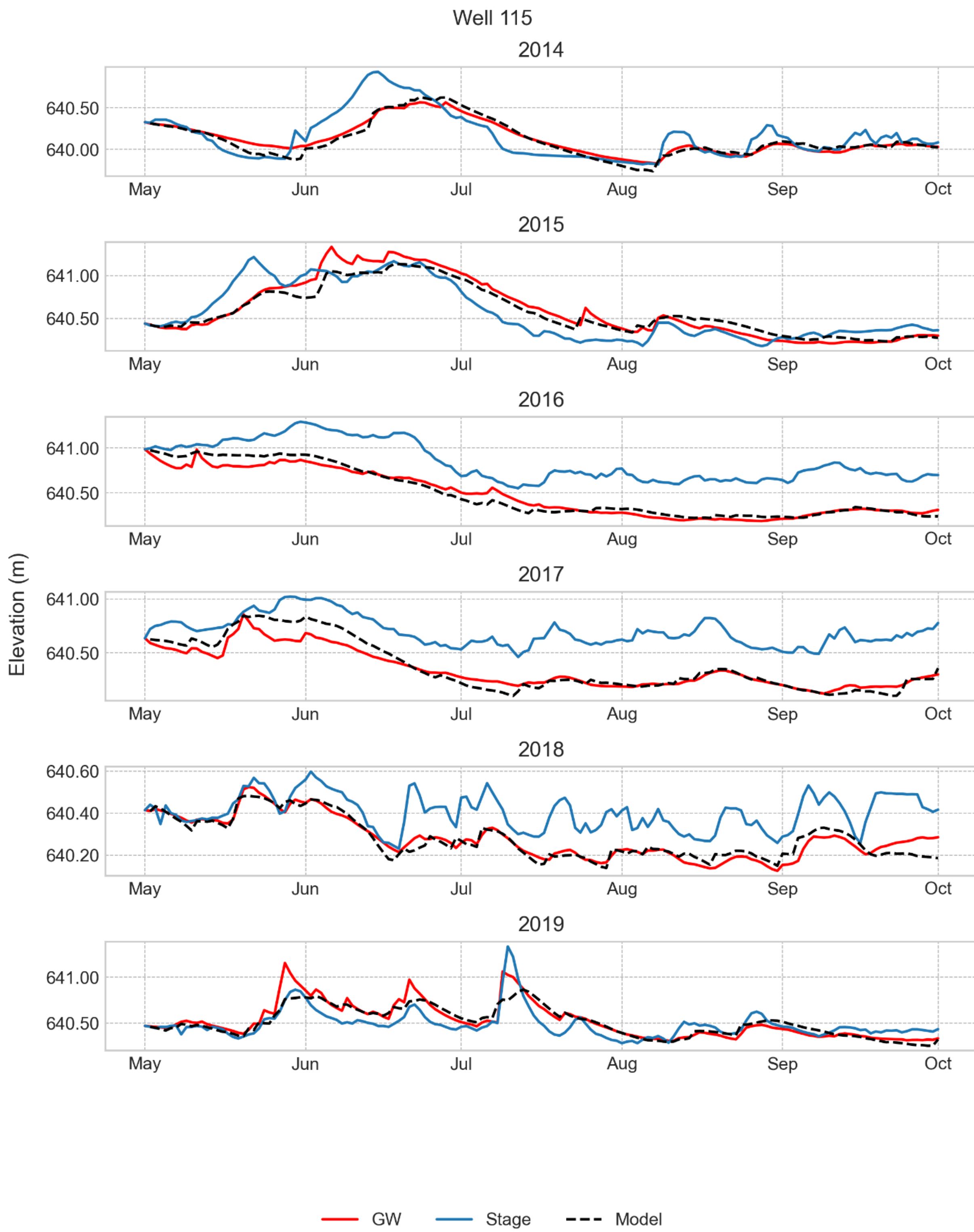
17


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

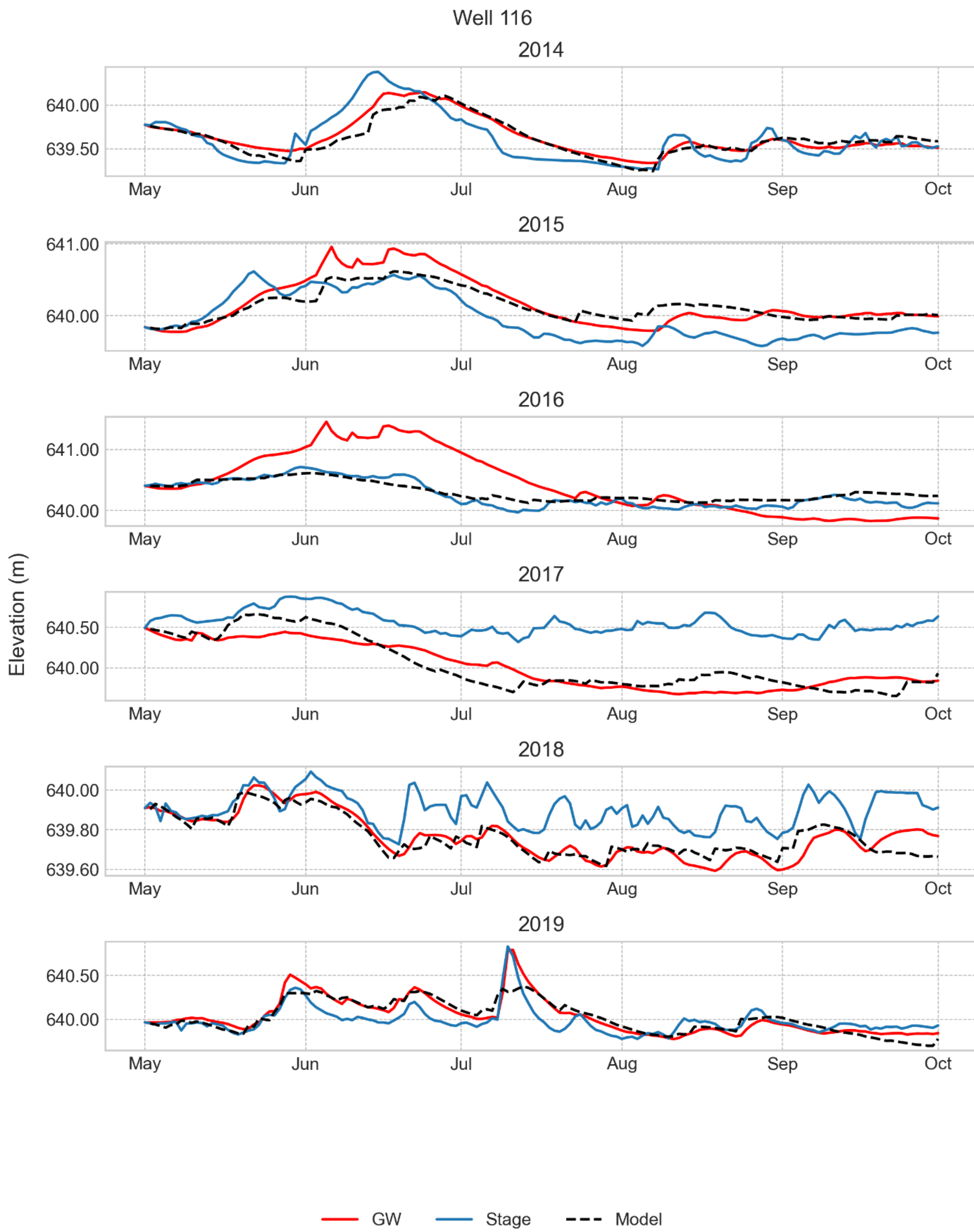
18


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

19

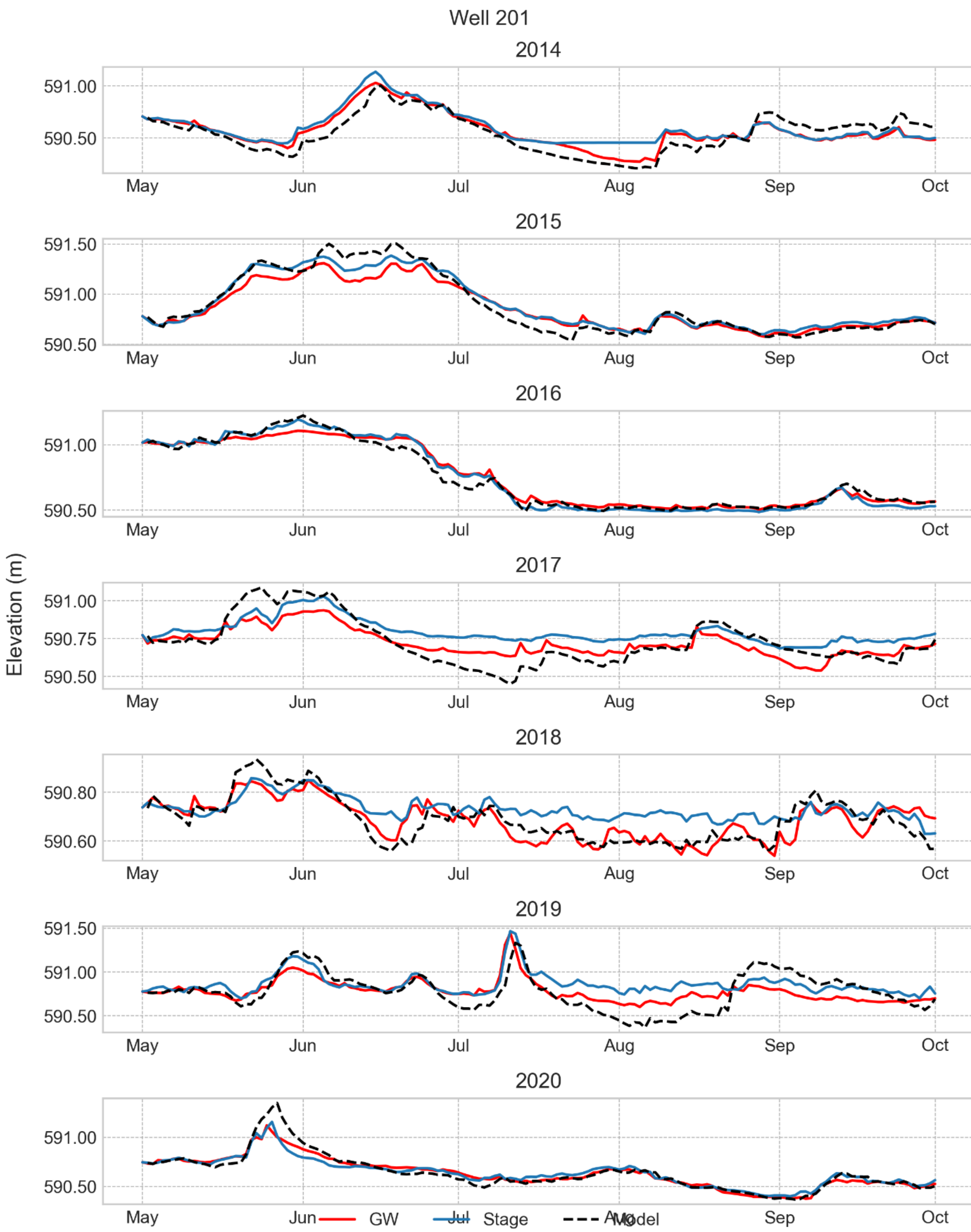

Well 115
2014
2015
2016
2017
2018
2019
Elevation (m)
640.00
640.50
641.00
640.20
640.40
640.60
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

20

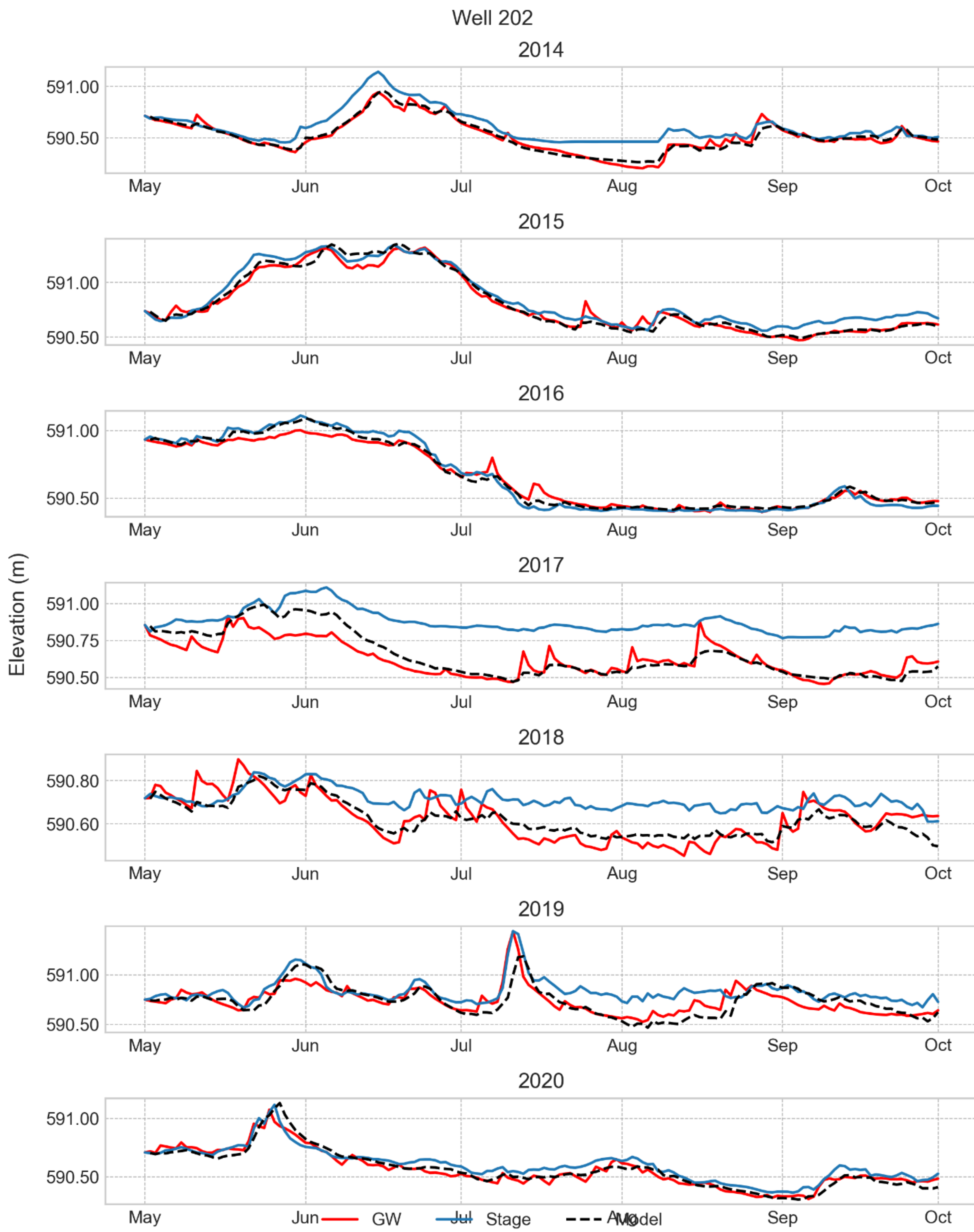


Well 116
2014
640.00
639.50
May
Jun
Jul
Aug
Sep
Oct
2015
641.00
640.00
2016
641.00
640.00
2017
640.50
640.00
2018
640.00
639.80
639.60
2019
640.50
640.00
Elevation (m)
GW
Stage
Model

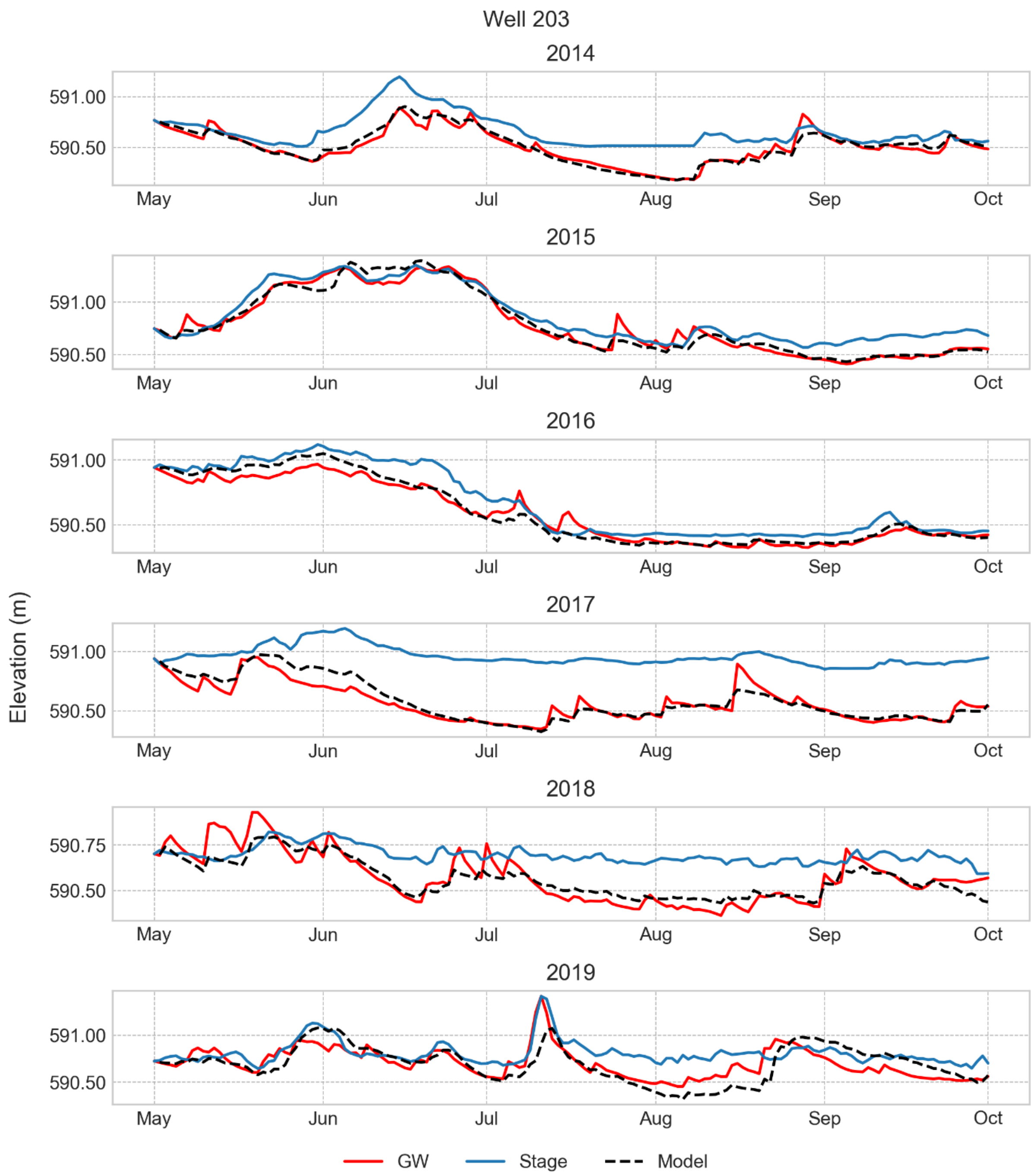
21


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

22

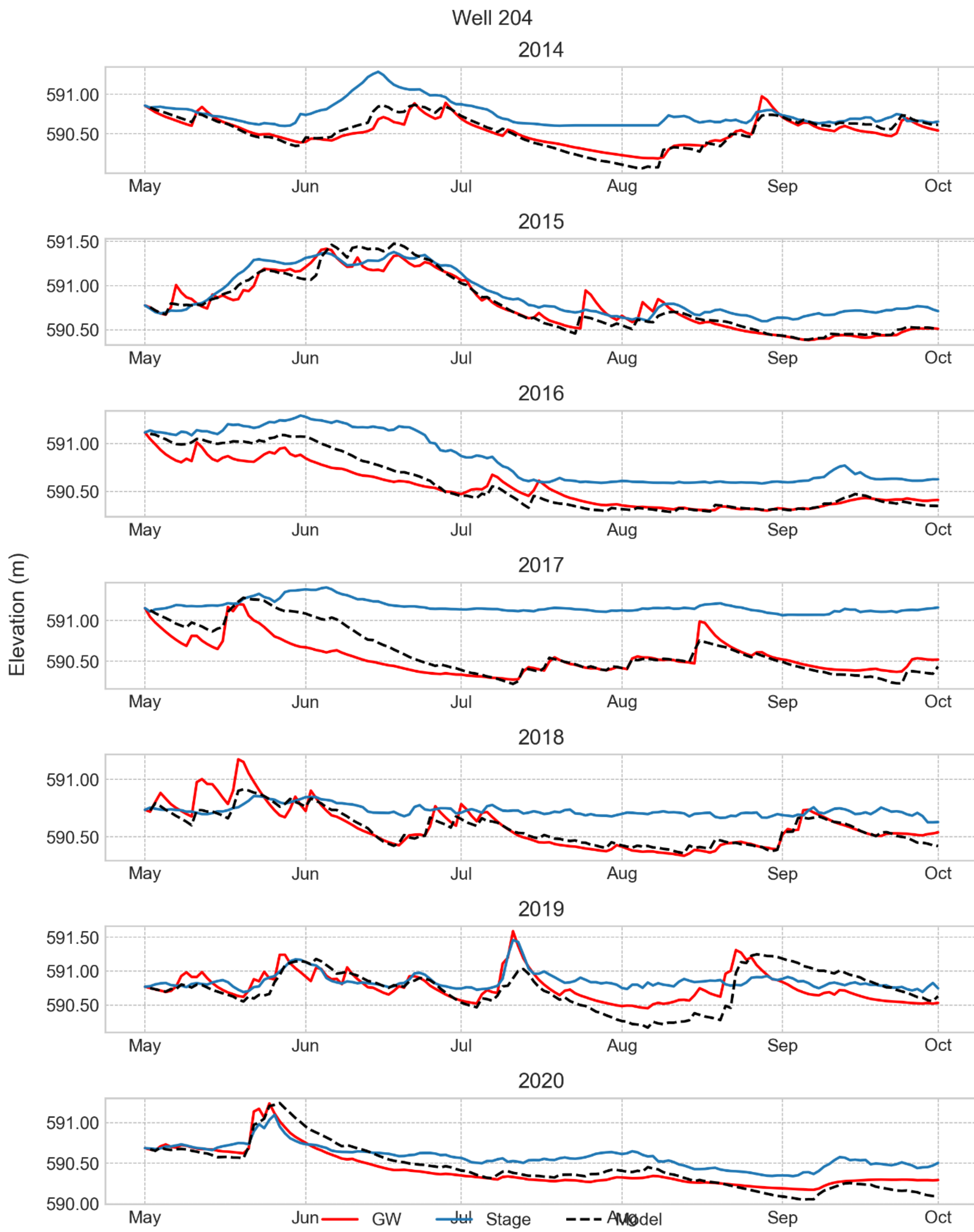

Well 202
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
591.00
590.50
590.75
590.80
590.60
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

23

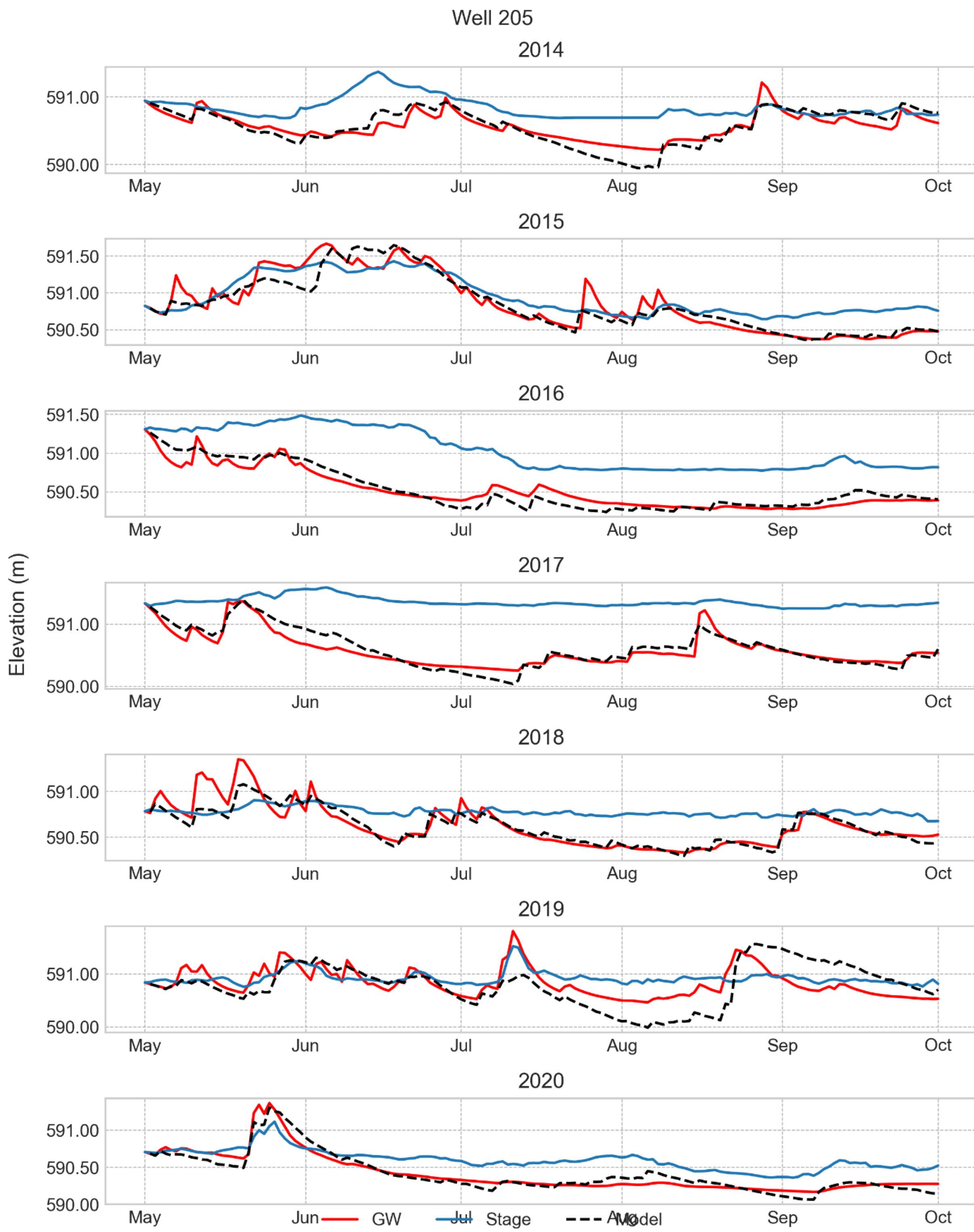

Well 203
2014
2015
2016
2017
2018
2019
Elevation (m)
591.00
590.50
590.75
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

24

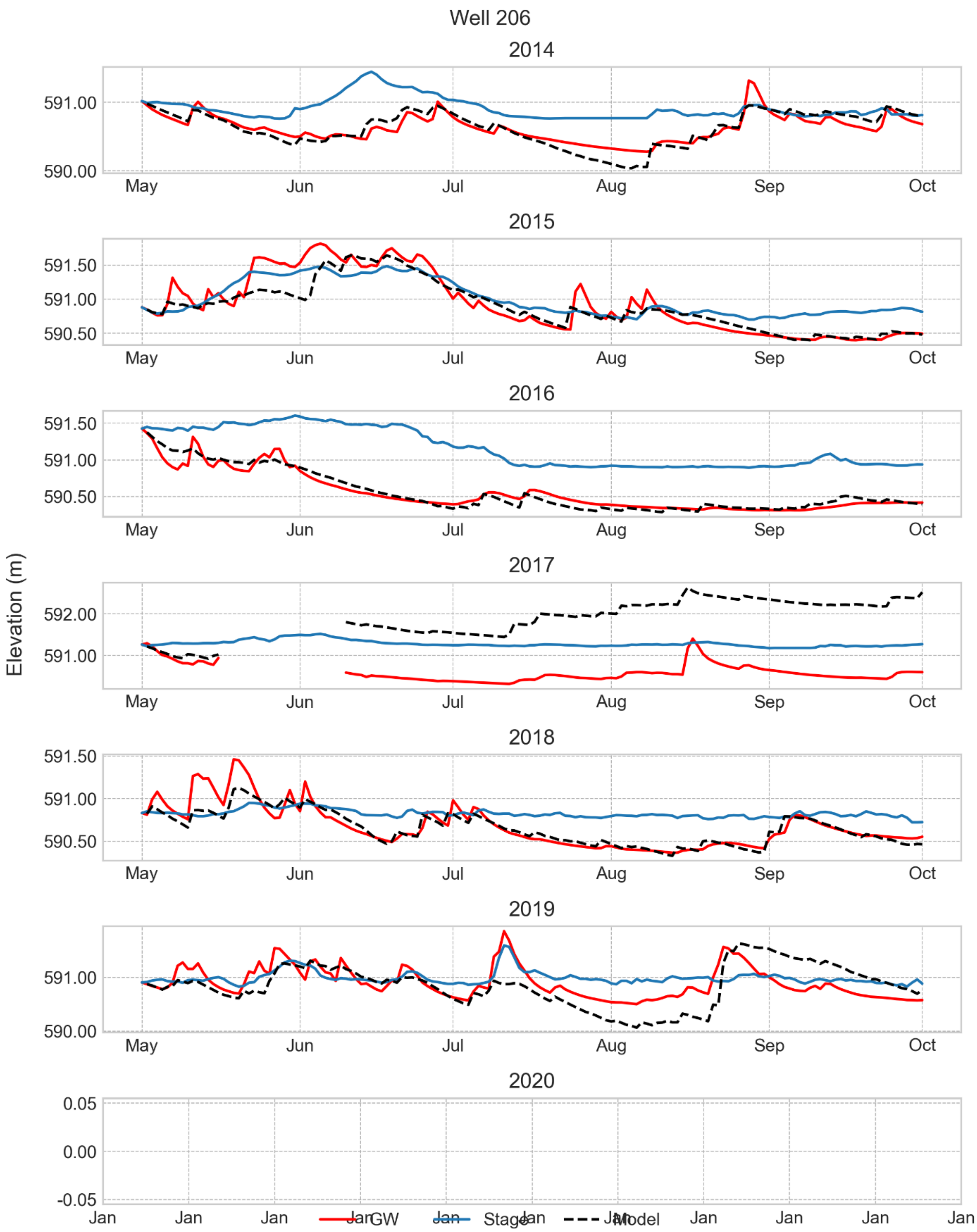
25


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

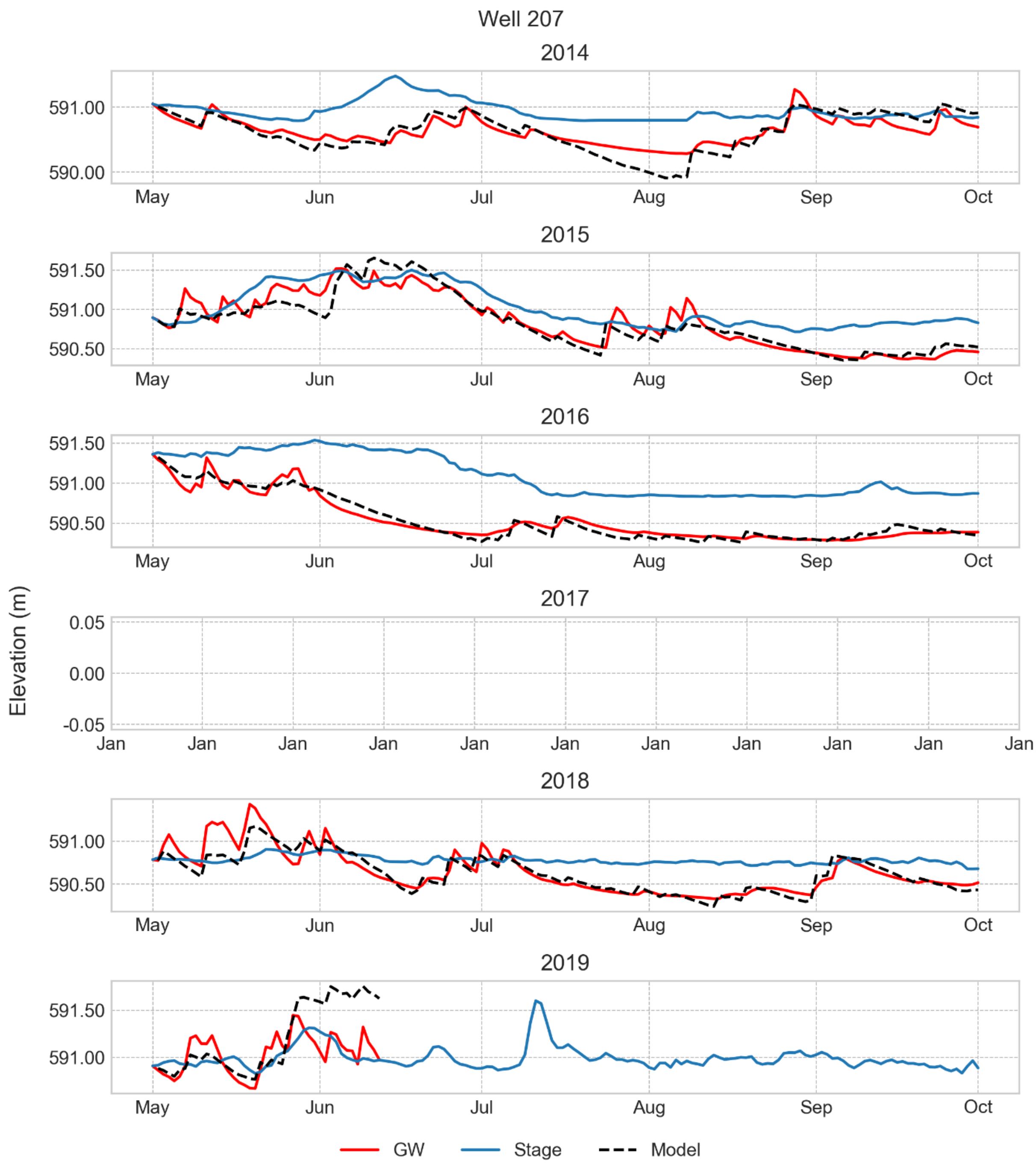
26

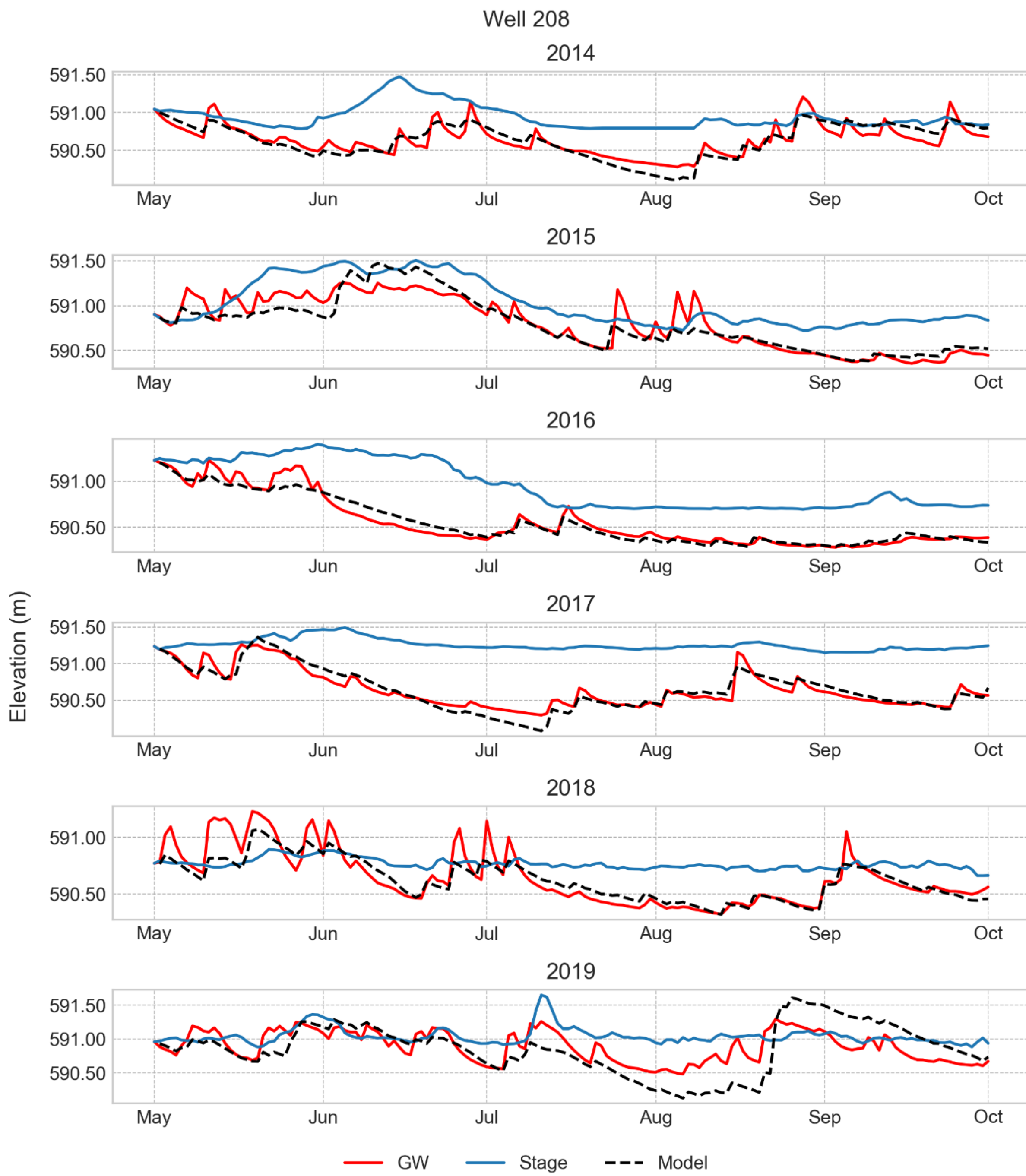

Well 205
2014
591.00
590.00
May
Jun
Jul
Aug
Sep
Oct
2015
591.50
591.00
590.50
2016
2017
2018
2019
2020
Elevation (m)
GW
Stage
Model

27


Well 206
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

28

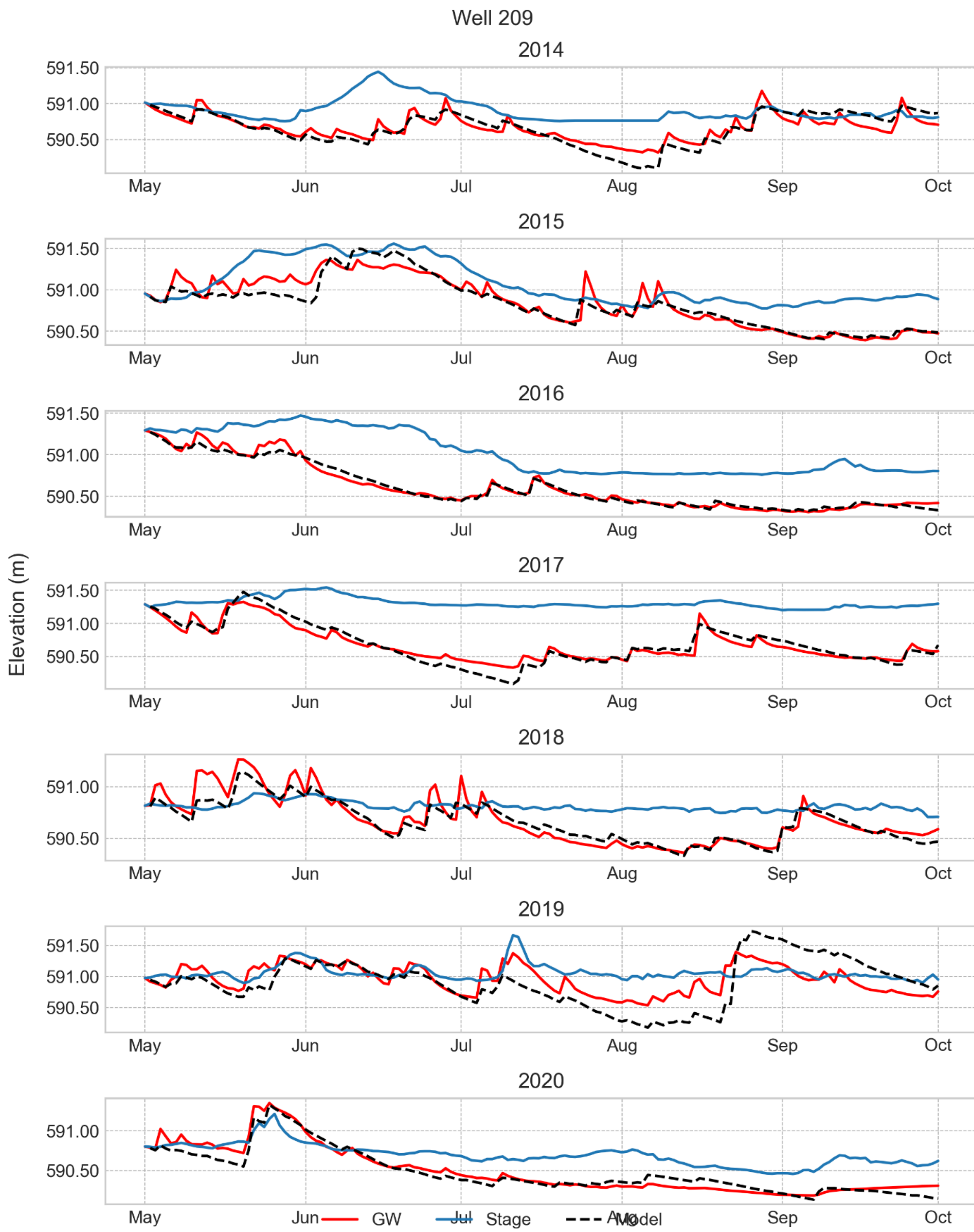

Well 207
2014
2015
2016
2017
2018
2019
Elevation (m)
591.00
590.00
591.50
590.50
0.05
0.00
-0.05
May
Jun
Jul
Aug
Sep
Oct
Jan
GW
Stage
Model



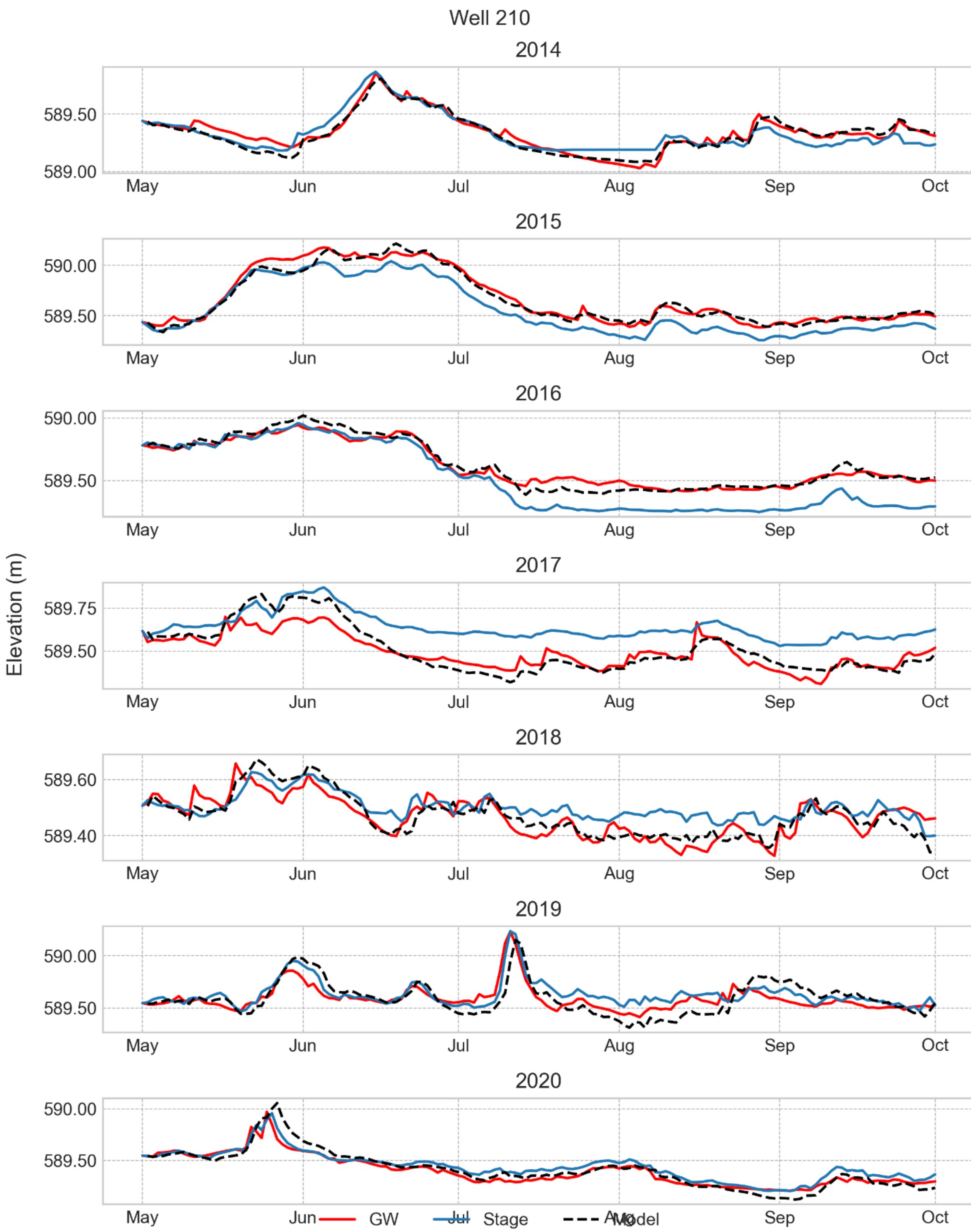
Well 208
2014
2015
2016
2017
2018
2019
Elevation (m)
591.50
591.00
590.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

31

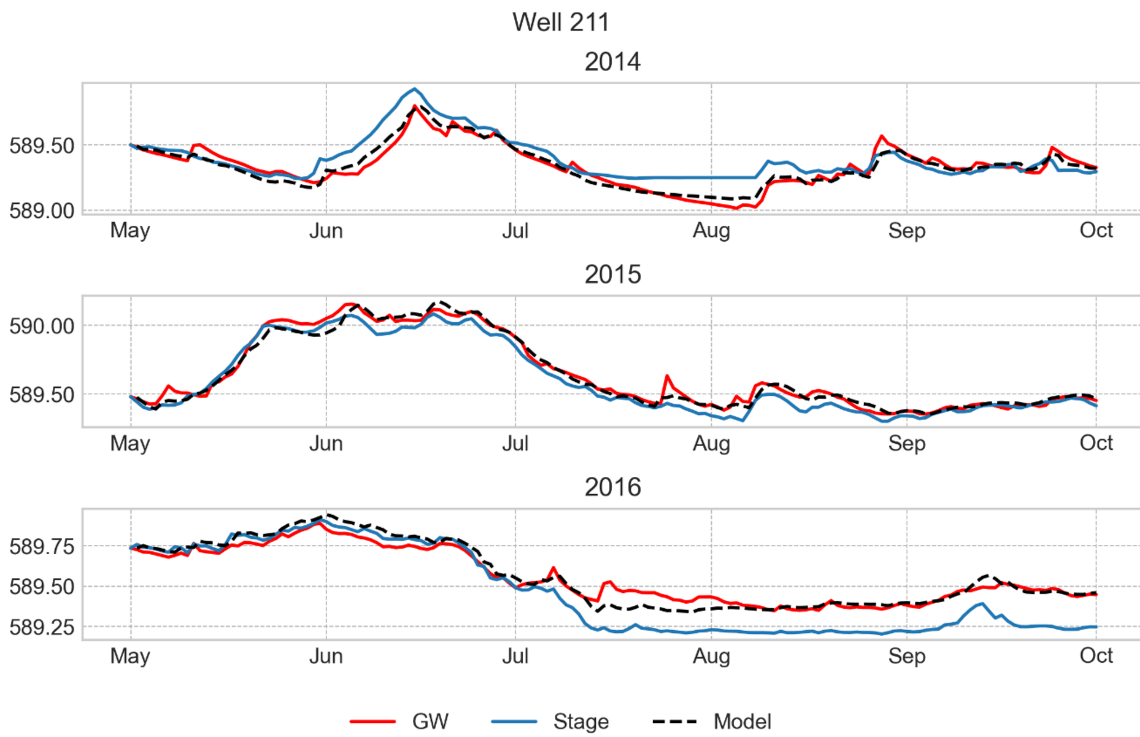
32

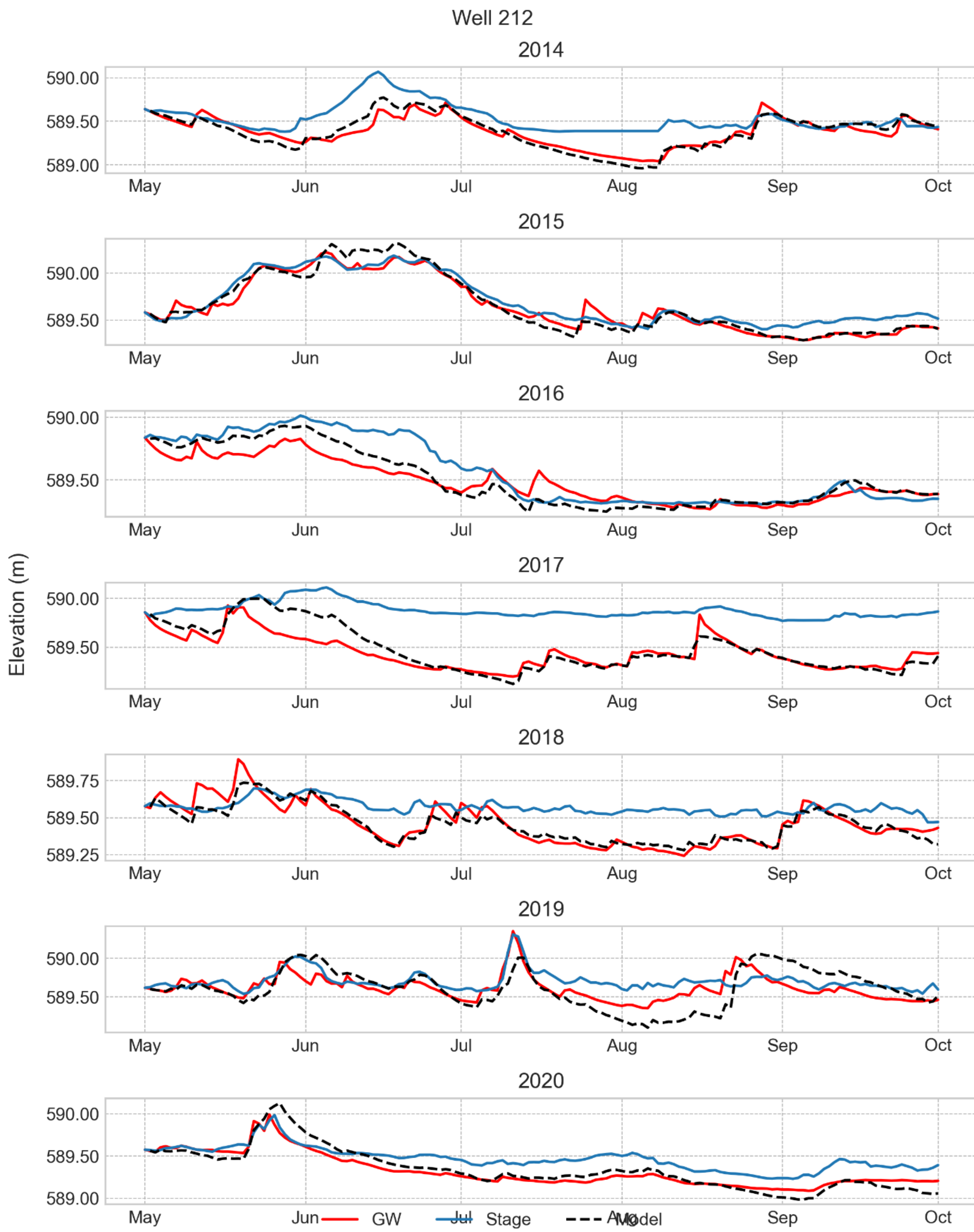

Well 209
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
591.50
591.00
590.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

33

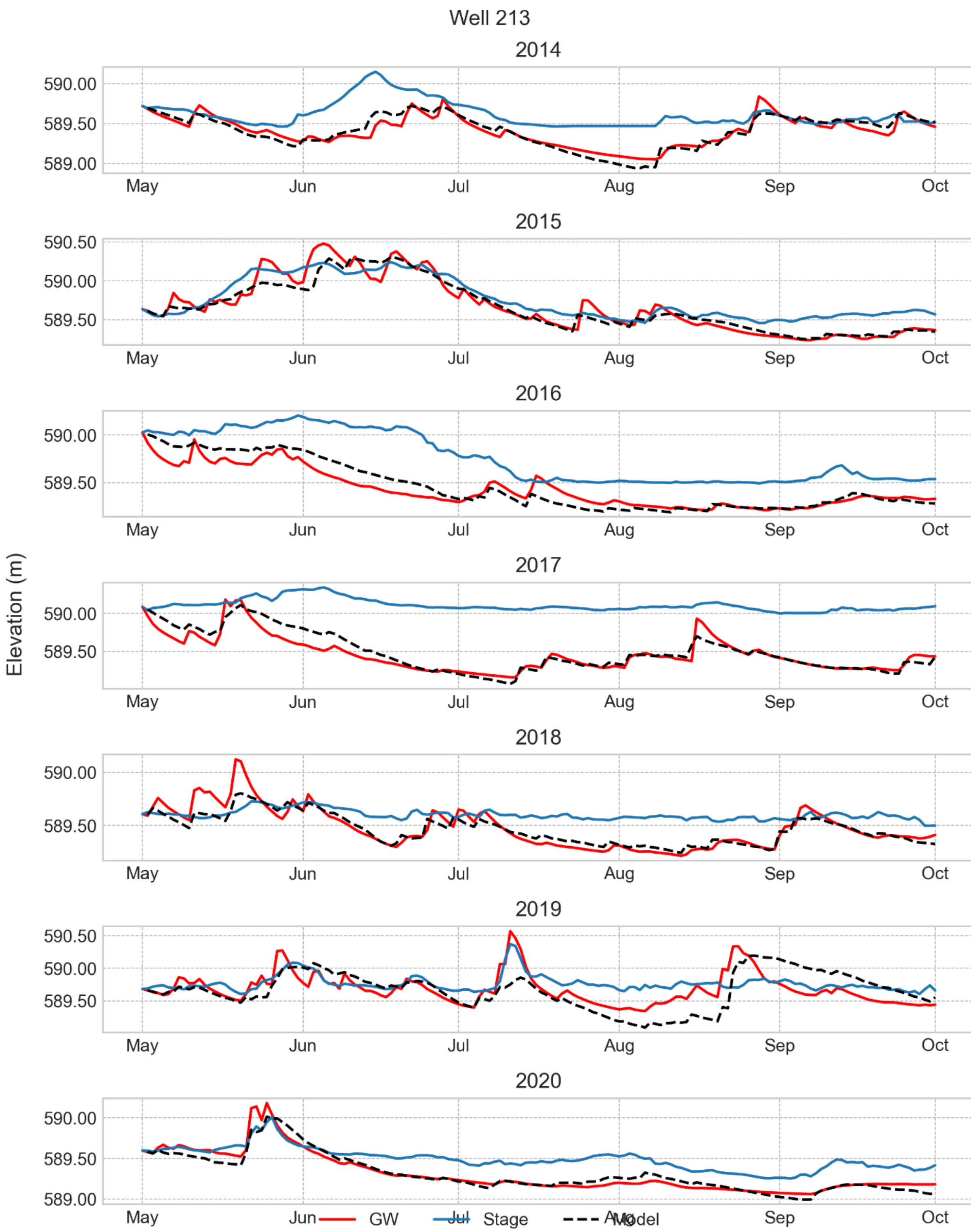

Well 210
2014
589.50
589.00
May
Jun
Jul
Aug
Sep
Oct
2015
590.00
589.50
2016
590.00
589.50
2017
589.75
589.50
2018
589.60
589.40
2019
590.00
589.50
2020
590.00
589.50
Elevation (m)
GW
Stage
Model

34


Well 211
2014
589.50
589.00
May
Jun
Jul
Aug
Sep
Oct
2015
590.00
589.50
May
Jun
Jul
Aug
Sep
Oct
2016
589.75
589.50
589.25
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

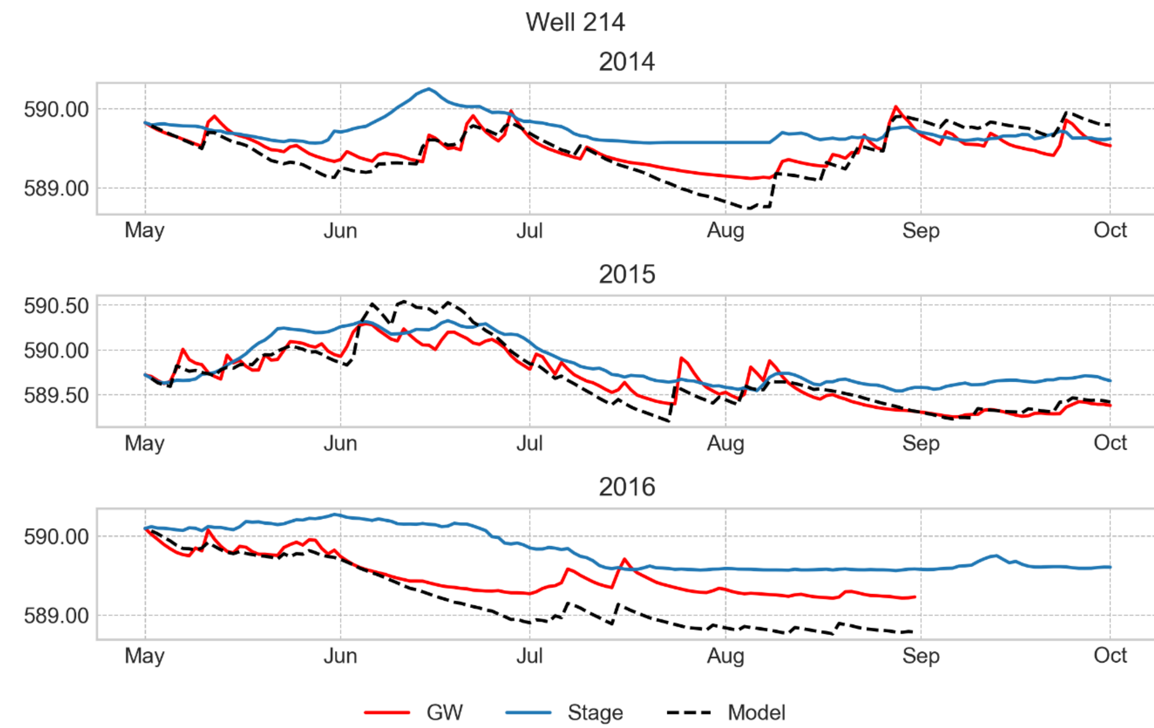

Well 212
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
590.00
589.50
589.00
589.75
589.25
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

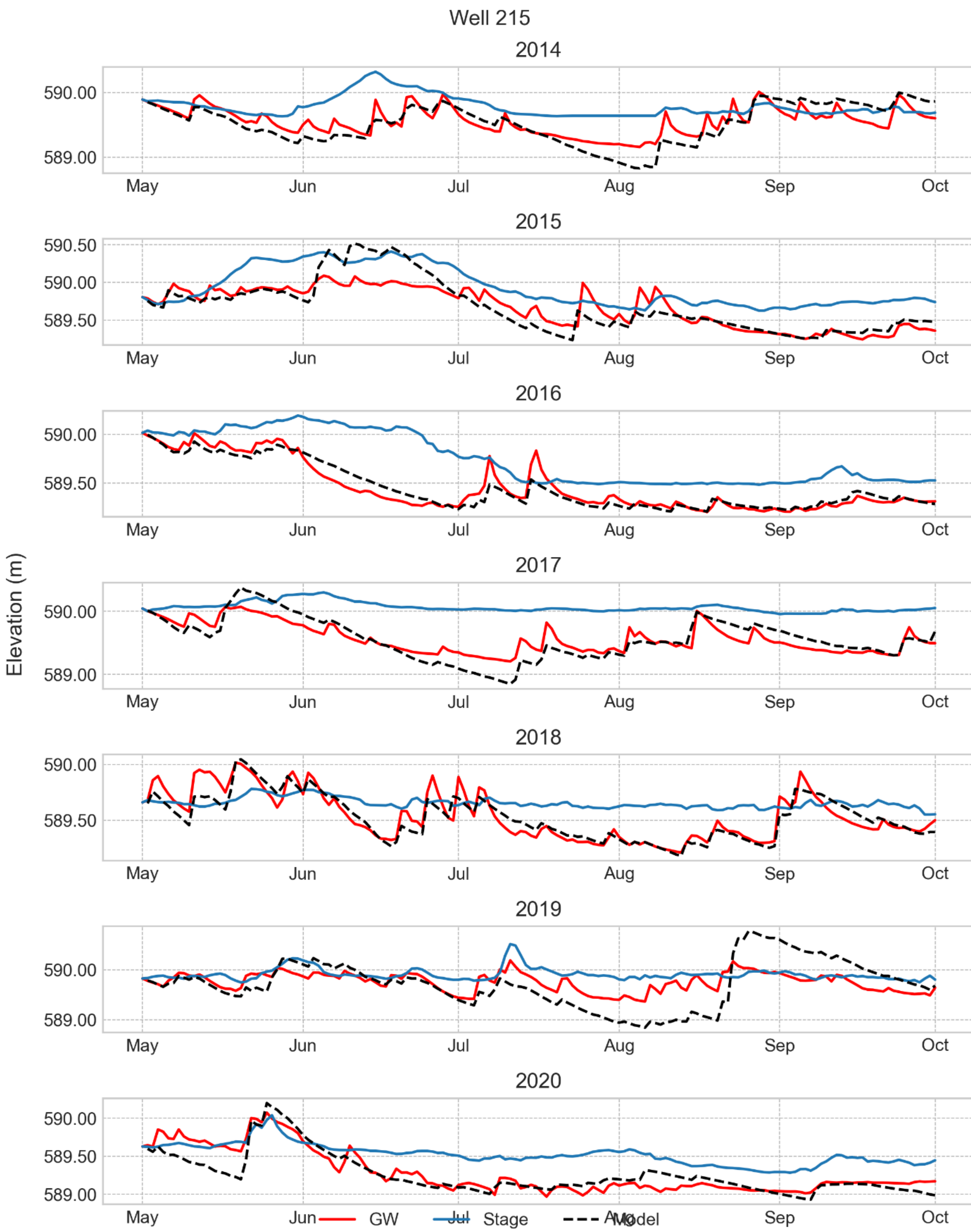
37



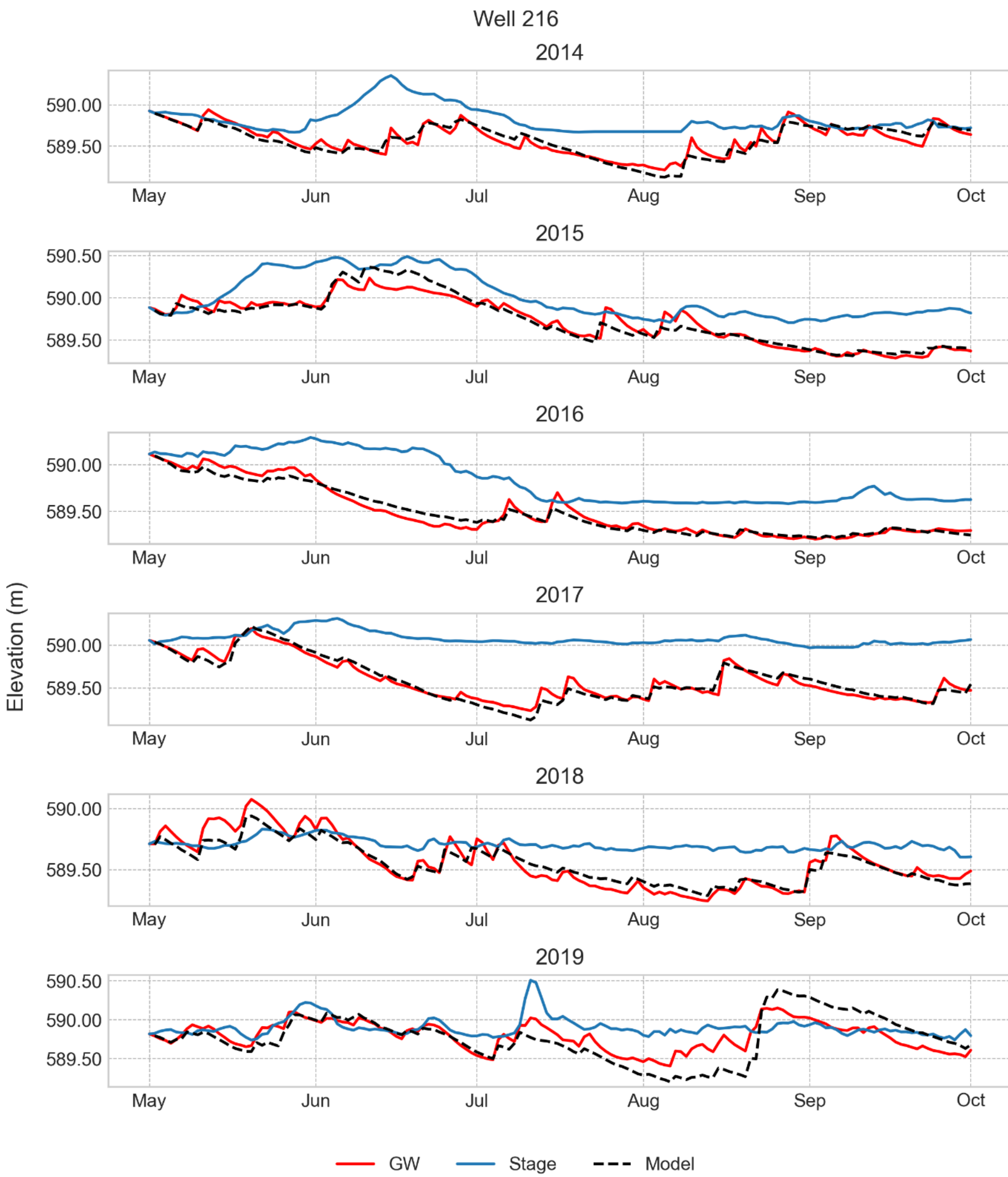
Well 213
2014
2015
2016
2017
2018
2019
2020
Elevation (m)
590.50
590.00
589.50
589.00
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model

38


Well 214
2014
590.00
589.00
May
Jun
Jul
Aug
Sep
Oct
2015
590.50
590.00
589.50
May
Jun
Jul
Aug
Sep
Oct
2016
590.00
589.00
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model


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

41


Well 216
2014
2015
2016
2017
2018
2019
Elevation (m)
590.50
590.00
589.50
May
Jun
Jul
Aug
Sep
Oct
GW
Stage
Model