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Mucking About: Hydrologic Regime and Soil Carbon Storage in Restored Subtropical Wetlands

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MUCKING ABOUT:
HYDROLOGIC REGIME AND SOIL CARBON STORAGE IN RESTORED SUBTROPICAL WETLANDS

by

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ABSTRACT

Wetlands are extremely important ecosystems that have declined drastically worldwide, continue to be lost, and are threatened globally. They perform a number of important ecosystem services such as flood control, provide habitat for many species, and have aesthetic and recreational value. Wetlands are also important to the global carbon (C) cycle. Wetland soils are especially effective C sinks because they have high primary productivity and low decomposition rates due to flooded, anoxic conditions. Increased recognition of wetlands' value has led to more ecological and hydrological restoration of degraded wetlands to mitigate the effects of wetland destruction. Hydrological restoration, which attempts to recreate natural hydroperiod and water levels in wetlands, is expected to increase soil C storage. Many studies have estimated the C stock in different wetland ecosystems across biomes, but few have examined hydrological drivers of soil C variation across wetland types.

This study investigated the relationship between hydrologic variables (hydroperiod and average water depth) and soil C storage in three types of hydrologically restored wetlands (marsh, bay swamp, and cypress swamp) at the Disney Wilderness Preserve (DWP) in central Florida, USA. I collected 150 50-cm soil cores along existing monitoring transects in sampled wetlands where water elevation data had been collected since 1995 to examine the relationship between hydrologic variable and soil C storage. I analyzed a combination of generalized linear mixed models (glmm), evaluated using AICc. Mean water depth was a better predictor than hydroperiod of soil C concentration and stock. Mean water depth had a significant positive relationship with soil C concentration in bay swamps and marshes and soil C stock in marshes. However, this effect was small and often outweighed by other factors such as differences in

vegetative community, soil depth, or local site conditions. Water depth had no significant relationship with soil C concentration in cypress swamps or upland communities or on soil C stock in bay swamps, cypress swamps, or uplands.

Wetland community type had a strong influence on soil C variation, with bay swamp soils having the highest mean soil C concentration followed by cypress swamp, marsh, and upland soils, respectively. Soil C concentration generally decreased with soil depth. Bay swamps also had the highest soil C stock, followed by cypress swamp, marsh, and upland soils, respectively. Together, the sampled wetland communities cover approximately 22% of the sampled communities at DWP, yet store an estimated 47% of the total soil C to a 90 cm depth. The results of this study affirm the importance of inundation for soil C storage in wetlands, but also highlight that there are a number of other complex variables affecting soil C in different types of wetlands such as differences in litter quality and decomposition rates.

ACKNOWLEDGMENTS

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INTRODUCTION

Wetlands are among the world's most ecologically important ecosystems, but remain threatened globally. Commonly referred to as "nature's sponges," wetlands can quickly absorb large quantities of water and gradually release their stores over time. This phenomenon buffers the effects of storm events by reducing storm water runoff and flooding in natural and human-developed areas and recharges groundwater aquifers (Bullock and Acreman 2003; Zedler and Kercher 2005; Mitsch and Gosselink 2007; Min, Perkins, and Jawitz 2010; McLaughlin and Cohen 2013). Wetlands also improve water quality by filtering sediments, pollutants, and excess nutrients from the water column either through sediment deposition or biological transformations (Johnston, Detenbeck, and Niemi 1990; Russell, Guynn, and Hanlin 2002; Zedler and Kercher 2005; Mitsch and Gosselink 2007). Additionally, wetlands moderate local climates (Marshall et al. 2004), support regional biodiversity (Semlitsch and Bodie 1998; Sabo et al. 2006), and are important habitat for many threatened species (Chapman, Chapman, and Chandler 1996; Hamer, Lane, and Mahony 2002). Humans further benefit from wetlands through the harvest of edible and commercially profitable species and through recreational activities (Whitney et al. 2004, Mitsch and Gosselink 2007).

Wetlands are also a significant part of the global carbon (C) cycle, serving as important C sinks (Chmura et al. 2003; Mitra, Wassmann, and Vlek 2005). While covering only 4-6% of global land area, wetlands store approximately 20-30% of all global soil organic carbon (SOC) (Mitsch and Gosselink 2007; Lal 2008; Yu et al. 2012). Wetlands store significant amounts of C in the form of soil organic matter (SOM), which is 40-60% C (Mitsch and Gosselink 2007; Reddy and DeLaune 2008). A combination of high primary productivity and slowed decomposition due to waterlogged, anoxic soils, results in the accumulation and long term storage of SOM (Tate 1987; Batjes 1999; Chmura et al. 2003; Mitra, Wassmann, and Vlek 2005; Bridgham et al. 2006; Mitsch and Goseelink 2007). SOM has a strong influence on aboveground vegetation, ecosystem function, and environmental quality (Kimble et al. 2007; Reddy and DeLaune 2008) by regulating soil temperature, increasing soil moisture holding capacity, enhancing soil aggregate stability, and serving as a nutrient reservoir for plants (Tate 1987; Schlesinger 1991; Mitsch and Gosselink 2007; Reddy and DeLaune 2008). For these reasons, the presence and amount of SOM is commonly used as an indicator of ecosystem health, especially in wetland communities.

Some ecosystem services provided by wetlands can be difficult or costly to mimic with human infrastructure, yet wetlands are still highly endangered (Costanza 1997; Mitsch and Gosselink 2007). An estimated 50% of global historic wetland land cover has been lost and wetlands are continually threatened by hydrologic modification for agriculture, mosquito control, and development (Kimble et al. 2007; Mitsch and Gosselink 2007). Florida, one of the most wetland-dense states in the United States, has lost an estimated 46% or 3.8 million ha of wetlands since European settlement (circa 1780) (Mitsch and Gosselink 2007).

Beginning in the 1970s, wetlands gained recognition for their role in flood control, coastal protection, and water quality enhancement. A series of international meetings and treaties including the 1971 Convention on Wetlands of International Importance (Ramsar Convention), Agenda 21 from the 1992 Rio Earth Summit, and World Summit on Sustainable Development (WSSD) called for the protection of ecologically important wetlands. The United States 1972 Clean Water Act, the adoption of a “no net loss” policy in 1989, and other legislation has increased protections for wetlands across the country. These efforts have slowed wetland losses, and have even led to some even some “gains” in protecting, restoring, or creating wetlands, but we are far from achieving stability in wetland cover and functioning (Mitsch and Gosselink 2007). Not only do human activities such as urbanization impact existing wetlands, but restored or created wetlands do no always provide the same ecological services as natural or undisturbed wetlands (Bishel-Machung et al. 1996; Zedler and Callaway 1999; Moreno-Mateos et al. 2012; McLaughlin and Cohen 2013; Lewis and Feit 2015).

Because soil C sequestration is enhanced by flooded conditions, hydrologic restoration of wetlands is expected to increase soil C accretion and storage (Kimble et al. 2007; Vincent, Burdick, and Dionne 2013). A better understanding of the relationship between hydrology and C storage in wetlands could prove useful in managing wetlands to mitigate C emissions and anthropogenic climate change. Several studies of wetland C stock in a variety of biomes show wide variation in C storage across scales, indicating that wetlands are complex, highly variable systems with C stocks differently affected by climate, hydrology, vegetation, soil conditions, and microbial populations (Mitra et al. 2005; Bridgham et al. 2006; Kayranli et al. 2010). However, few studies have explored the relationship between hydrological conditions and wetland C storage (Sahuquillo et al. 2012; Sulman, Desai, and Mladenoff 2013; Lewis and Feit 2015; Nyamadzawo et al. 2015). Most of these studies focus on a single wetland type and most have limited hydrologic data. This study aims to increase our understanding of the drivers of variation in wetland soil C storage by investigating and modeling the relationship between hydrological variables and wetland soil C storage in different wetland communities in a restored subtropical landscape.

Research Questions

My research examined variation in soil C stock of three types of restored wetlands at the Disney Wilderness Preserve (DWP) in Central Florida in order to provide insights into landscape-scale C storage throughout the upper Kissimmee basin of the Northern Everglades ecosystem. The results will improve our understanding of wetland ecosystem function, specifically by examining the contribution of various drivers to variation in wetland soil C storage.

My three main research questions were:

1. How do long-term (10-year) mean hydroperiod (days inundated per year) and water depth relate to soil C in different types of depressional wetlands in a subtropical landscape?
2. Does the vegetative community type of a wetland interact with hydrological factors to influence C storage?
3. What is the contribution of wetland soils to C storage at the landscape scale?

Based on these questions, I predicted that:

1. Wetlands characterized by greater long-term mean water depths and longer hydroperiods would store more soil C than wetlands with shallower depths and shorter hydroperiods.
2. There would be a strong interactive effect between hydrology and vegetative community on wetland soil C storage.
3. Models based on hydrologic variables such as mean water depth and hydroperiod could be used to predict soil C storage at both the local and landscape scales.

METHODS

Study Site

This study focused on the Disney Wilderness Preserve (DWP), south of Kissimmee, FL, USA. DWP is a 4,654 ha preserve managed by The Nature Conservancy (TNC) in Reedy Creek at the headwaters of the Greater Everglades watershed (Fig. 1). The site includes a 3,440 ha former cattle ranch dominated by wetlands, improved Bahia grass pastures, and degraded flatwoods ecosystems cattle grazed since the late 1800s. In 1992, the Walt Disney Corporation purchased the property for restoration to mitigate the expansion of Walt Disney World and development of Celebration, FL. The Greater Orlando Aviation Authority (GOAA) added an additional 1,200 ha to the preserve in 1995 as mitigation for expansion of the Orlando International Airport (MCO). The region has a subtropical climate with hot summers, mild winters, and cycles of wet (May – October) and dry (November – April) seasons. Annual mean temperatures range from 15°C in January to 27°C in August and mean annual precipitation is 1142 mm per year (South Florida Water Management District Station WRWX). Ground elevation at DWP ranges from 15 to 21 meters above sea level (TNC 1996).

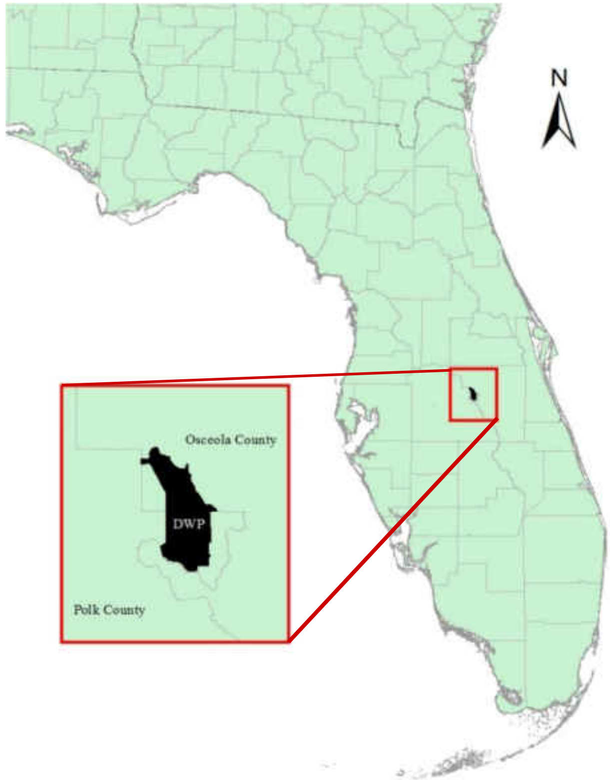


Figure 1. Map showing the location of the Disney Wilderness Preserve (DWP).

Upland and wetland ecosystems at DWP underwent ecological and hydrological restoration between 1994 and 2012. Among the restoration goals for this site was the enhancement of water quality and quantity for the Kissimmee and Northern Everglades ecosystems, in agreement with the Comprehensive Everglades Restoration Plan (CERP) (U.S. Congress 2000; TNC 2004). Hydrologic restoration involved filling drainage ditches to their original ground elevation in order to raise water levels and meet restoration targets for hydroperiod, water level fluctuations, and water level elevation. Hydrologic restoration targets were derived from "literature values and the professional judgment of original project ecologist" (TNC 2004). In drained wetland areas, TNC staff mechanically removed and chemically treated encroaching woody vegetation and invasive plant species to restore natural hydrophilic wetland vegetation. The restoration project met its criteria for success in 2011 (TNC 2011). When completed, this mitigation project was the first large off-site mitigation project of its kind and was a great success for conservation and ecological restoration (Gatewood 1995).

TNC continues to monitor and manage the site by conducting growing season prescribed burns on 2-3 year intervals, treating or removing invasive species, and monitoring threatened or endangered species on site such as the Red Cockaded Woodpecker (*Picoides borealis*) and Florida Scrub Jay (*Aphelocoma coerulescens coerulescens*). There are approximately 1,300 ha of restored wetlands at DWP (28% of the total land area) including freshwater marshes, cypress swamps, bay swamps, wet prairie, and floodplain swamps (Table 1) (Fig. 2).

Table 1. Wetland land cover at the Disney Wilderness Preserve.

Land Use Classes	**Total Area (ha)**	**Percent Cover**	**Mean Size (± SD) (ha)**
Freshwater Marsh	287	21.7	3.0 ± 11.8
Bay Swamp	159	12.0	17.7 ± 21.0
Cypress Swamp	239	18.1	2.8 ± 9.8
Wet Prairie	150	11.4	0.9 ± 1.6
Mixed Hardwood Swamp	108	8.2	4.2 ± 11.6
Floodplain Swamp	377	28.6	41.9 ± 44.2
Grand Total	**1,320**		**3.4 ± 12.71**

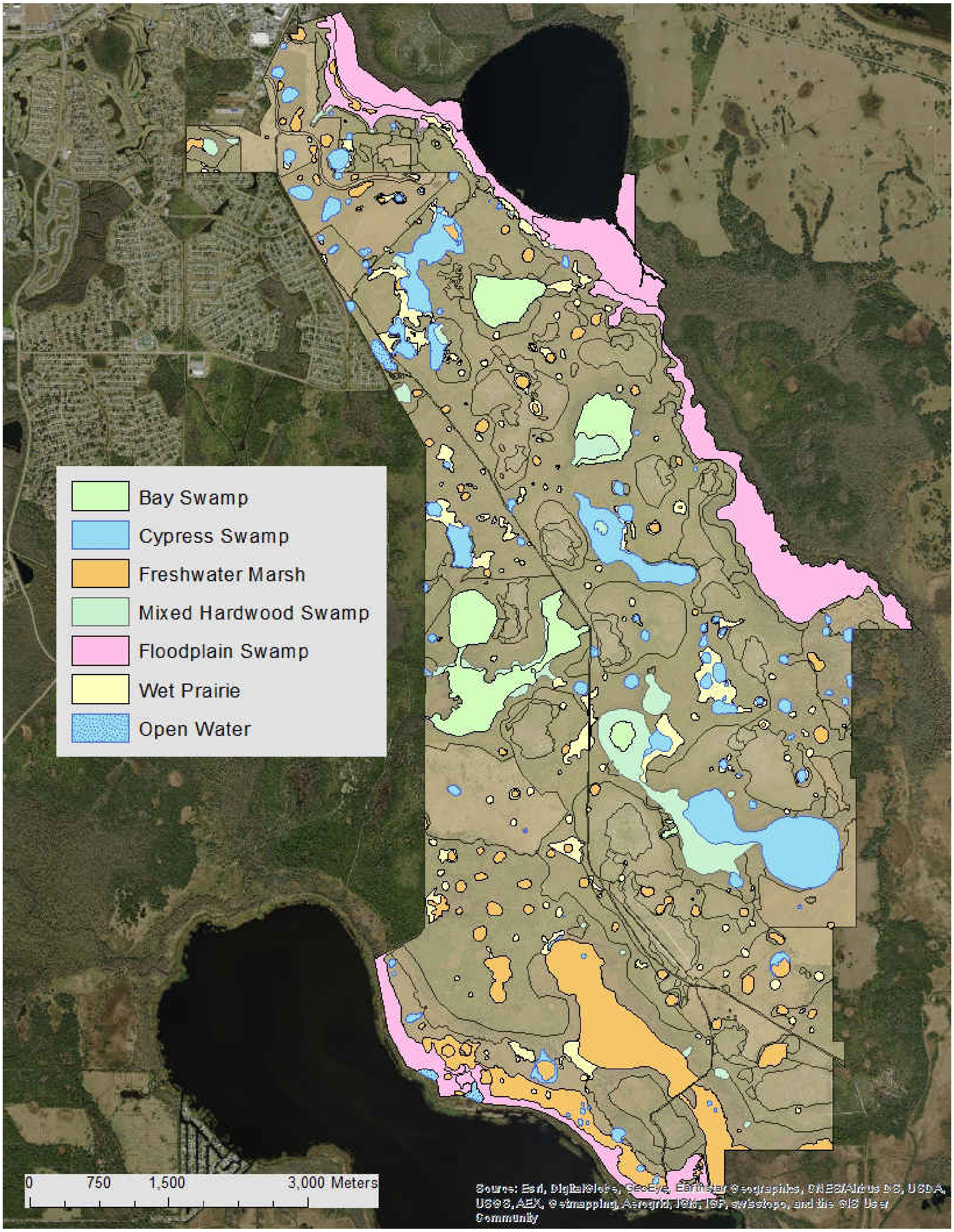


Figure 2. Wetlands at the Disney Wilderness Preserve.

For restoration and monitoring purposes, TNC grouped the wetlands at DWP into 24 hydrologic units (HUs) and established transects for initial site characterization and to monitor wetlands in each HU throughout the restoration process (Fig. 3) (TNC 1994). Transect number and length varied by size of the hydrologic units and number of wetlands in restoration. Transects generally spanned from wetland interiors into adjacent uplands to facilitate survey of ecotone transitional zones, which were expected to experience the most hydrologic change. Along each transect, TNC staff collected data on ground elevation, vegetative characteristics, organic matter depth, and water elevation before or during restoration efforts (TNC 1997).

Since 1995, TNC monitored wetland water level with over 450 manual wells and approximately 25 wells instrumented with continuous water elevation recorders (Telologer 2109e-5 pressure transducers, Telog Instruments, Victor, NY) (Fig. 3). Well installation dates varied by the restoration start dates in each HU. TNC employees recorded water elevations from manual wells, located along monitoring transects, once per month for 1-17 years. Monitoring of all manual wells ended by 2010. Continuous wells, distributed across DWP, recorded hourly means of water elevation from the time they were installed in the 1990s until 2010 or later. Of the original 25 continuous monitoring wells, 18 were still in operation in 2017.

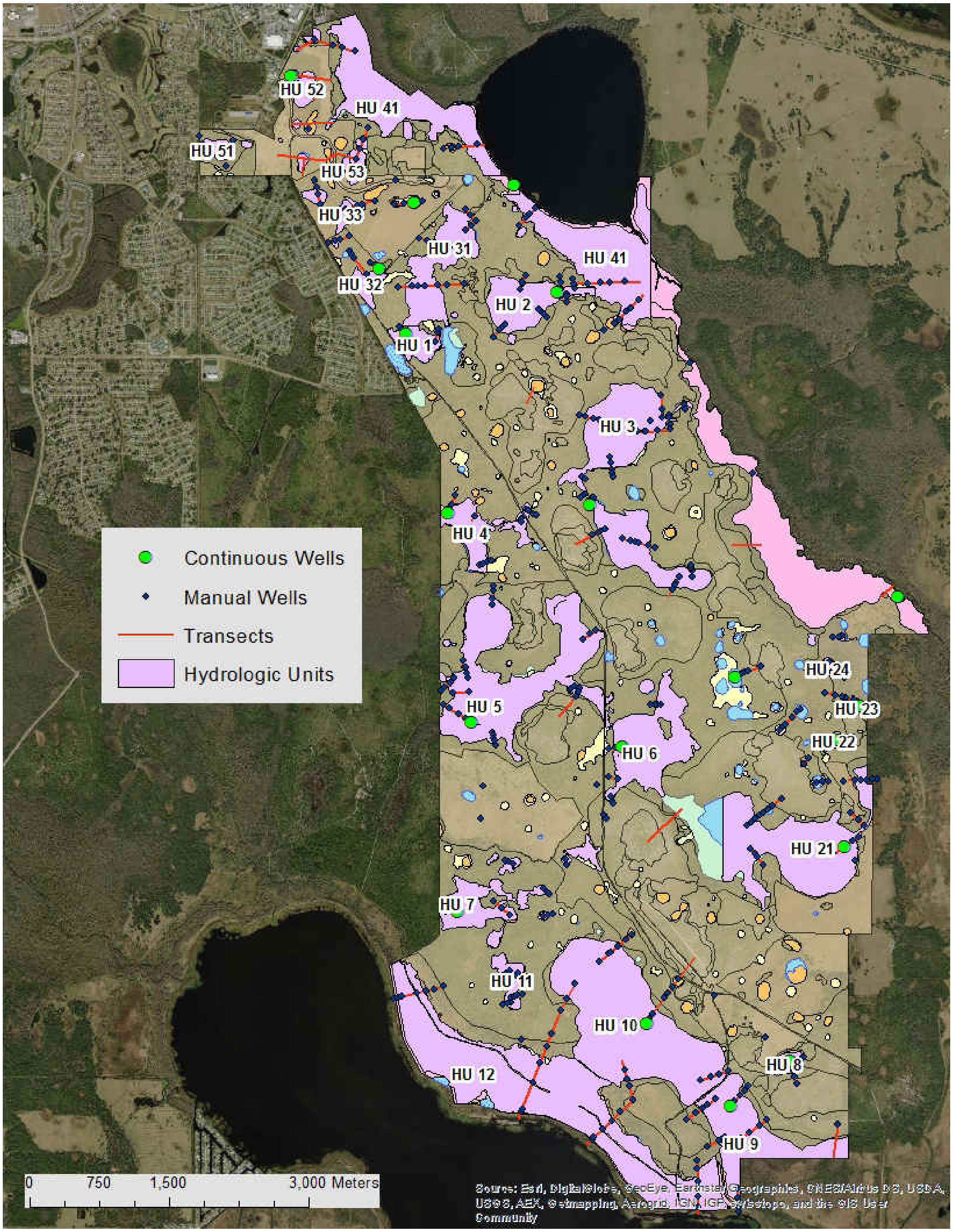


Figure 3. Designated hydrologic units (HUs), monitoring transects, and monitoring wells at the Disney Wilderness Preserve.

Wetland Site Selection

This study included the three dominant wetland types at DWP: freshwater marshes, bay swamps, and cypress swamps (Table 2), which together represent approximately 60% of the wetland cover and 17% of total land cover on site (Table 1). All three wetland types are depressional wetlands that are mostly rain-fed with no or little surface flow into or out of the system. This study does not include non-depressional wetlands at DWP such as riparian wetlands, which are hydrologically dominated by adjacent waterbodies, or the large heterogeneous marsh at the southern end of the site, which is hydrologically more similar to a floodplain than an isolated depressional wetland.

Table 2. Characteristics of wetlands of interest (from Helton 1996).

Community Type	Shape	Dominant Species	Soil
Bay Swamp	Large, Irregular Basin	*Magnolia virginiana* *Gordonia lasianthus* *Persea palustris*	Deep organic layers, Highly variable basal soil: dark salt & pepper to clayey muck
Cypress Swamp	Strand or Isolated dome	*Taxodium ascendens*	Shallower organic layers; Basal soils: organic to mucky clay
Marsh	Basin or Depression	Grasses (*Typha sp., Sagittaria sp.*), sedges (*Panicum sp., Cladium sp.*), floating aquatic plants (*Nymphea sp., Nelumbo sp.*)	Highly variable basal soil: sandy organic to clayey muck

Candidate wetlands included ecologically restored bay swamp, cypress swamp, and marsh communities that had at least one monitoring well within their boundaries and at least one monitoring transect with ground elevation data necessary to calculate hydrologic variables from water elevation data. Of the approximately 186 wetlands at DWP, 49 individual wetlands met these requirements. Twenty-two wetlands, including 10 freshwater marshes, 7 cypress swamps, and 5 bay swamps, met final selection criteria based on the quality and quantity of their available hydrologic data, size (> 0.5 ha), and location in the landscape.

Soil Sample Site Selection

Sampling teams collected a total of 150 soil cores (Fig. 4), which were distributed among the sampled wetlands based on the proportional coverage of each wetland type at DWP (Table 3), and assigned among individual wetlands according to the wetlands' size and number of transects. Mixed hardwood swamp cover was combined with bay swamp since both community types occur together as part of the same individual wetlands. Using ArcGIS (version 10.3.1) and TNC's land class and monitoring data, I established three zones at each wetland site: "upland transect," which included a 25 m buffer upland of the wetland boundary; "wetland transect", which included the TNC monitoring transects within the designated wetland boundaries; and "wetland interior," which was the area of the wetland beyond the monitored transect. I used ArcGIS to randomly place sampling points along transects in the upland transect and wetland transect zones. When the transects did not reach the wetland interior, I selected additional sampling points at approximately 50 m intervals beyond the endpoint of the existing monitoring transects (Table 4). Wetlands in which (a) transect(s) reached the center of the wetland did not have a wetland interior zone.

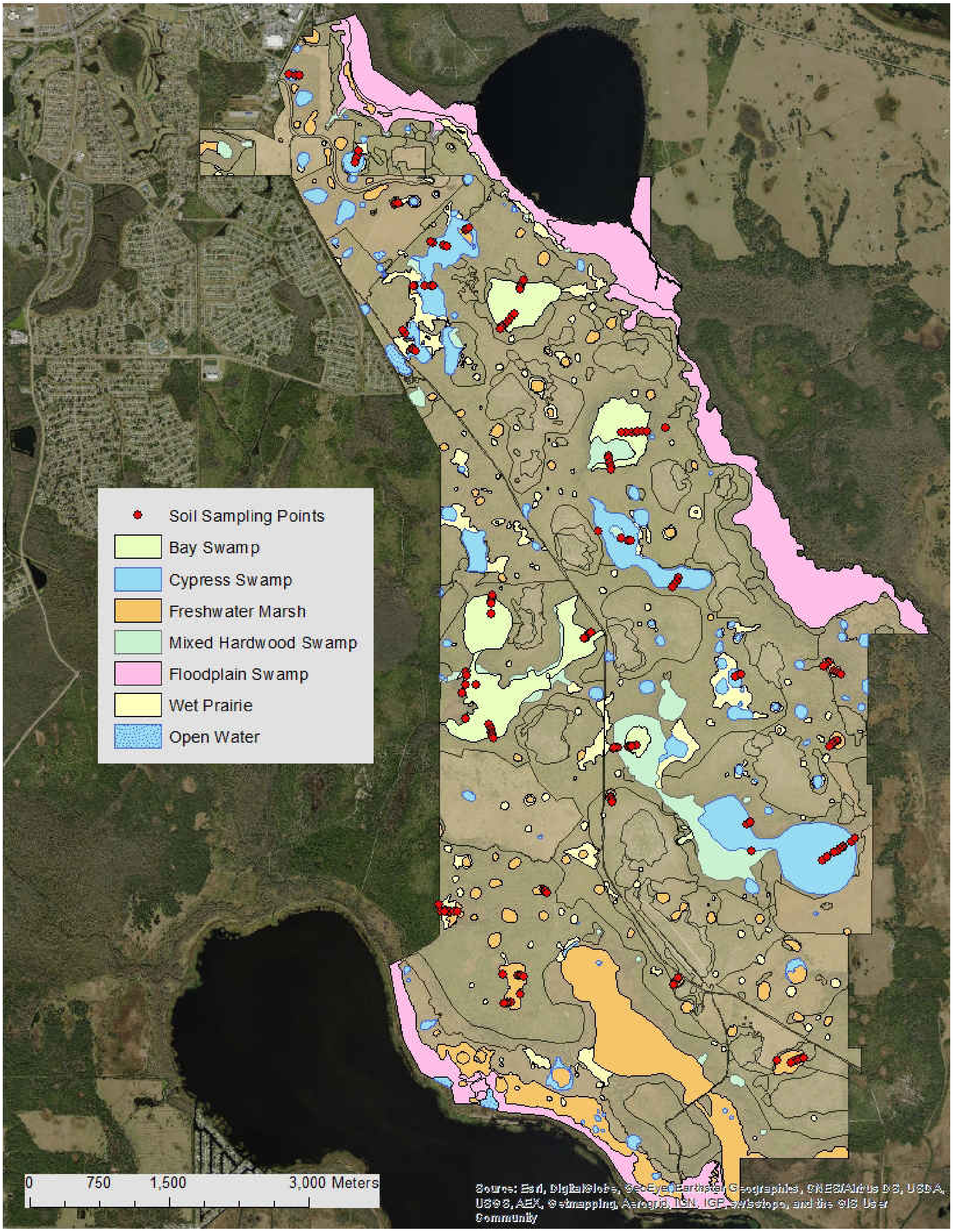


Figure 4. Soil sampling locations at the Disney Wilderness Preserve.

Table 3. Count and relative cover of wetlands of interest at the Disney Wilderness Preserve and soil sampling distribution by wetland type.

Wetland Community	Total Count	Total Cover (ha)	% Cover	Selected Wetlands	Sampling Points
Bay Swamp	5	271	34	5	**51**
Cypress Swamp	86	239	30	7	**45**
Marsh	95	287	36	10	**54**
Total	186	797	100	22	**150**

Table 4. Soil sampling distribution among wetlands zones in the three wetland types included in the study.

Wetland Community	Upland	Transect	Interior	Total
Bay Swamp	11	28	12	**51**
Cypress Swamp	11	32	2	**45**
Marsh	13	41	0	**54**
Total	**35**	**101**	**14**	**150**

<u>Soil Sample Collection and Processing</u>

Sampling teams collected all soil samples between March and November 2016. Soil cores were taken with a three-inch outer diameter clear polyurethane pipe hammered into the ground to a 50 cm depth. Teams extruded cores from the pipe in the field and cut them into six depth segments: 0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-40 cm, and 40-50 cm. Detritus was not collected. Samples remained on ice until brought back to the lab, where they stayed at 4°C until processed.

Mass measurements for each core segment included initial field-wet mass and oven-dried mass (60°C). Soil moisture was the percentage of water mass lost during drying. Soil bulk density was the oven-dried mass of the whole soil core segment sample divided by the volume of the whole core segment. SOM was determined on 1.5-2.0 g subsamples of sieved (2 mm mesh) dried, and mechanically ground (Spex Sample Prep 8000M Mixer/Mill) samples that were ashed for 16 hours at 550°C in a Fisher Scientific Isotemp® muffle furnace. The Wetland Biogeochemistry Analytical Services (WBAS) lab at Louisiana State University (Baton Rouge, LA) determined carbon and nitrogen content (g C kg^{-1} soil) for 50 randomly selected soil core segments (Costech 1040 CHNOSA Elemental Combustion). I used these data to create a regression equation to convert organic matter content to C content for the remaining samples (APPENDIX A). I calculated total C content by multiplying percentage C by the total dry soil mass and calculated soil C stock (S_C, kg C m^{-2}) using the horizon depth (D), percent carbon of materials passing the 2 mm screen (C_S), mass of materials passing the 2 mm screen (M_S), and the total volume of the depth increment (V) as follows:

$$S_c = \frac{D * \left(10{,}000\,\frac{cm^2}{m^2}\right) * (C_S * M_S)}{V}$$

Hydrological Analysis

When possible, I used data from the geographically closest well to each sampling location to calculate hydrologic variables for each point. To estimate daily water elevations at manual well locations where only monthly measurements were available, I fit regressions comparing water elevation readings at manual wells and the nearest continuous well, using readings taken within 30 minutes of each other (APPENDIX B). I then used the regression equation to estimate continuous water elevation the manual well location using data from the continuous well location. I assessed the fit of each regression with R^2 values and residual plots. In the event that regression fit was poor for between manual well data and data from the geographically closest continuous well, I chose another continuous well for the regression. I used a total of 73 manual monitoring wells and 17 continuous recorders for analysis.

I estimated daily relative water depths at each sample location by subtracting the site ground elevation, determined from original site surveys, from estimated daily water elevations, and then calculated the mean water depth between January 1, 2001 and December 31, 2010. I chose this date range because water elevation data was available from all continuous wells for this time period. I calculated hydroperiod as the number of days per year a sampling point was inundated (water depth > ground elevation), including a 2 cm buffer. I then calculated the 10-year mean hydroperiod for the same time frame as water depth.

Statistical Analysis

To model both soil C stock (kg C m^{-2}) and concentration (g C cm^{-3}), I fit a number of generalized linear mixed models (glmms) with the function glmmadmb using the glmmADMB package (Fournier et al. 2012, Skaug et al. 2013) in R version 3.3.0 (R Core Team 2016). I ran the models with a gamma family because the data did not meet the assumptions of normality for maximum likelihood (ML) or restricted maximum likelihood (REML). To account for non-random soil sampling design, each model included the following random effects in the models' random intercepts: individual core ID (when analyzing by soil depth) nested within transect nested within wetland. Distribution plots of the random effects are shown in APPENDIX C. The models also included combinations of these fixed predictor variables: mean water depth, standard deviation of the mean water depth, mean hydroperiod, soil depth segment, and vegetative community type. Over 20 models were fit of logical combinations of the above predictor variables and their possible interactions while excluding collinear predictor variables from appearing in the same model (such as mean water depth and hydroperiod or hydroperiod and vegetative community). I used the corrected Akaike information criterion (AICc) for model selection and evaluated the residuals of the most informative plots. I also ran glmms to analyze water depth, hydroperiod, and soil bulk density by vegetative community type and community type by soil depth. Models including hydrologic variables exclude interior sample points, because accurate ground elevation data were not available for these points. All other models include data from all soil samples.

RESULTS

Wetland Hydrology

Hydroperiod was significantly shorter in upland communities than in the wetland communities (Fig. 5, $p < 0.001$). Among wetland community types, mean hydroperiod in marshes was significantly shorter than in cypress and longer than in bay swamps ($p < 0.05$). There was no significant difference in mean hydroperiod between bay and cypress swamps. Bay swamps had the greatest range of hydroperiod (0-356 days/year), whereas cypress swamps had the smallest range of hydroperiod (220-357 days/year). Hydroperiod was normally distributed in cypress swamps and marshes, slightly skewed towards shorter hydroperiods in bay swamps, and very skewed towards longer hydroperiods in uplands (Fig. 5). Median hydroperiod in the upland communities was 21 days/year, but over 25% of upland sampling locations had hydroperiods over 100 days/year. The random effect variance (1.45), due to individual wetlands and individual transects in a given wetland, is on par with or greater than the effect of community type on hydroperiod, indicating that a significant amount of variability in hydroperiod was due to local site conditions. The model output and residual plots are shown in APPENDIX D.

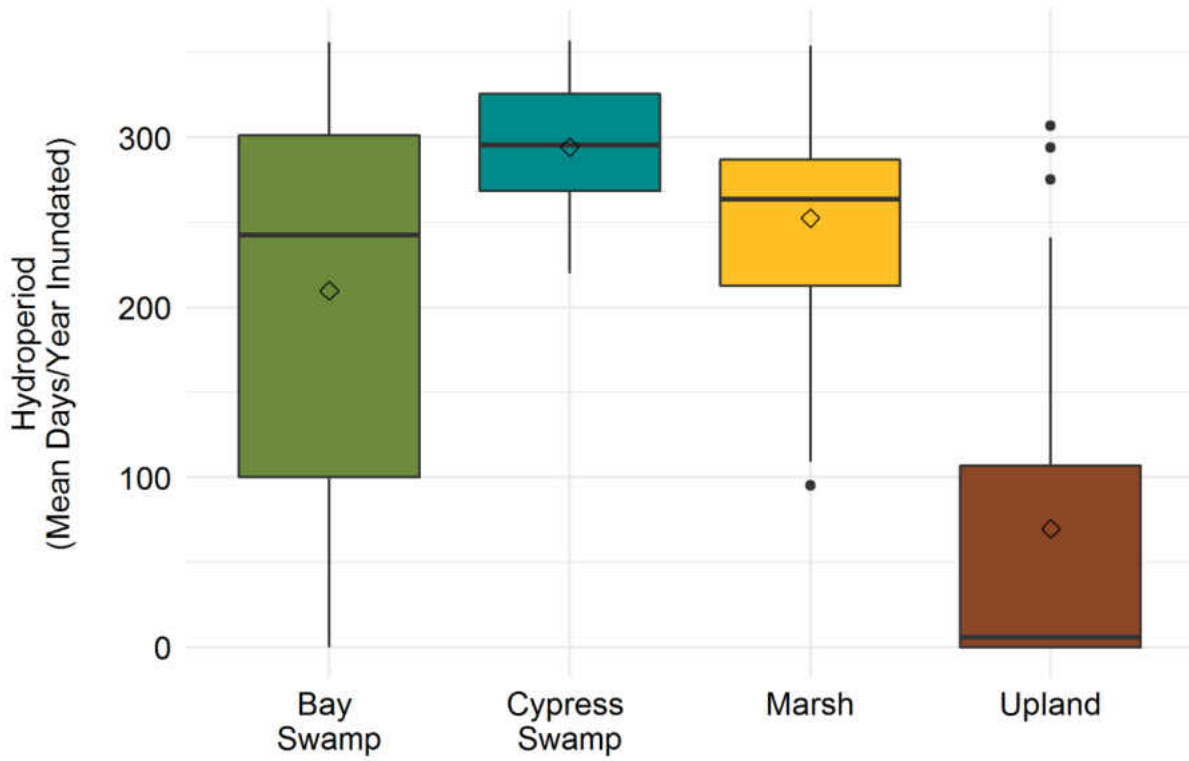


Figure 5. Mean hydroperiod in three wetland communities and adjacent upland communities. Diamonds represent the means, horizontal lines the medians.

Mean water depth was significantly lower in the upland communities than the wetland communities (Fig. 6, $p \approx 0$). There was no significant difference in mean water depth between bay and cypress swamps or between cypress swamps and marshes, but mean water depth was slightly (~17 cm) but significantly lower in bay swamps than in marshes ($p < 0.05$). Water depth was normally distributed in all community types. Mean water depth was below the ground surface in both bay swamp (-3 cm) and upland communities (-52 cm). The random effect variance (0.026), due to individual wetlands and individual transects within wetlands was much lower than the random effect variance on hydroperiod and lower than the effects of community type on mean water depth, indicating that more of the variably in mean water depth is due to differences in vegetative community than to local site conditions. The model output and model residual plots are shown in APPENDIX E.

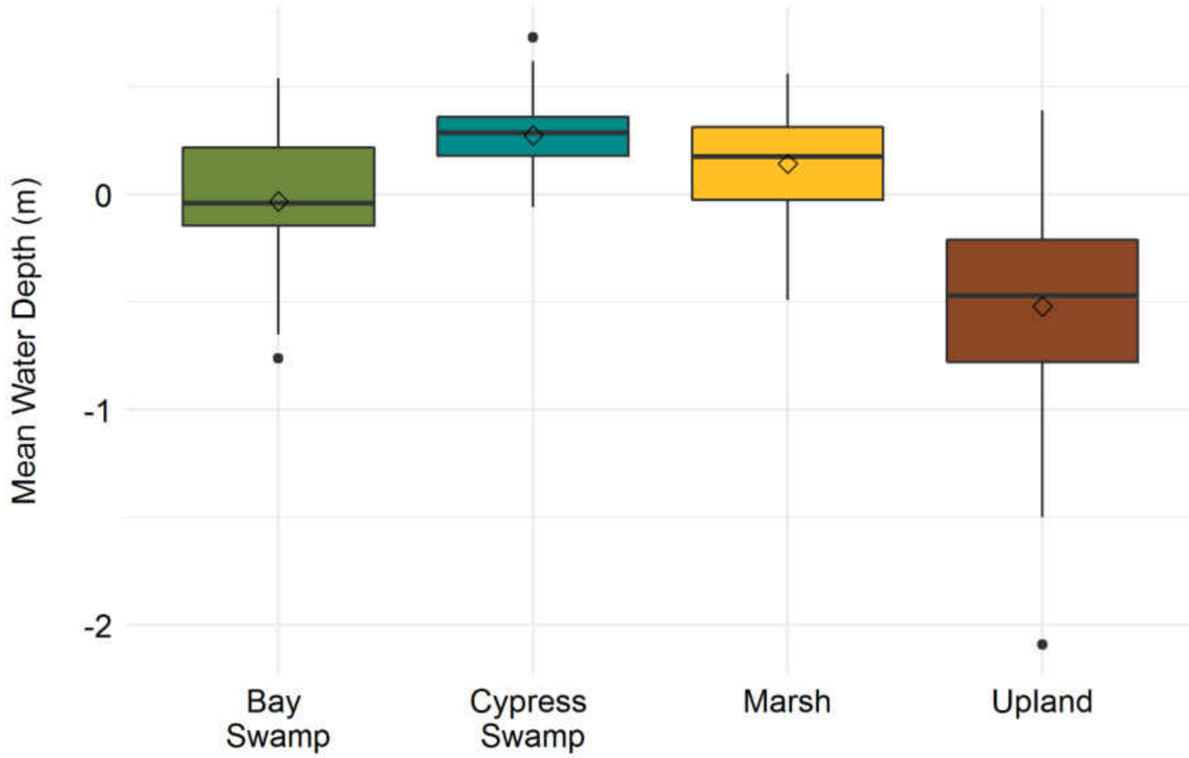


Figure 6. Mean water depth in three wetland communities and adjacent upland communities. Diamonds represent the means, horizontal bars the medians.

Soil Carbon Concentration

Mean soil C concentration for 0-50 cm depth was significantly different among all community types (Fig. 7, $p < 0.01$). Bay swamps had the highest mean soil C concentration, followed by cypress swamp, marsh, and uplands, respectively. The distribution of soil C concentration was greatly skewed in all wetland types, towards the lower C concentrations of the range in bay swamps and towards the higher concentrations of the range in in cypress swamps and marshes. The random effect variance (0.066), due to individual wetlands and individual transects within wetlands is lower than the estimates for community types, indicating a small effect of local site conditions on soil C concentration. The model output and residuals are shown in APPENDIX F.

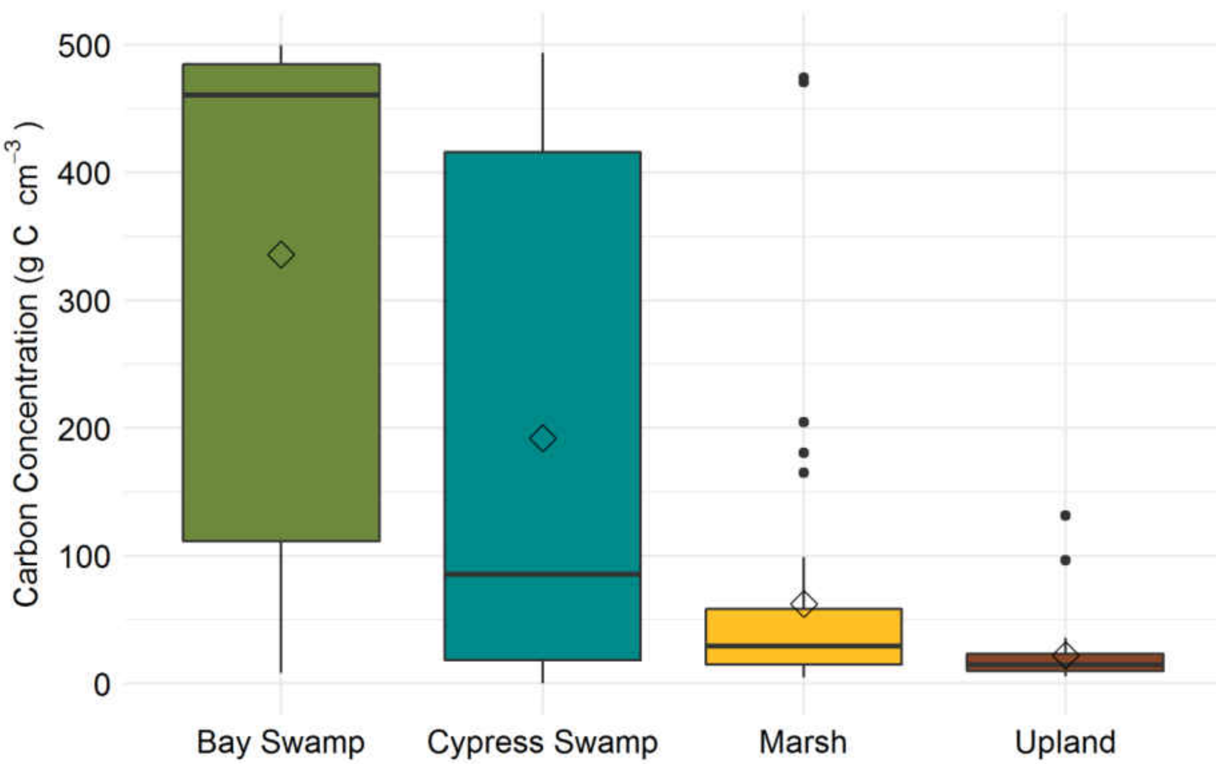


Figure 7. Soil carbon concentration for 0-50 cm depth in three wetland communities and adjacent upland communities. Diamonds represent the means, horizontal bars the medians.

At each soil depth below 5 cm, bay swamp soils had the highest mean C concentration, followed by cypress swamp, marsh, and upland soils (Fig. 8). In the 0-5 cm soil depth, mean C concentration was not significantly different among the wetland communities. At the 5-10 cm depth, there was no significant difference in mean soil C concentration between cypress and bay swamps, but both were significantly higher than marsh and upland. After 10 cm depth, all differences in mean soil C concentration across community types are significant. The total random effect variance due to individual wetlands, transects, and soil cores (1.25) was on par with or greater than the estimates of the fixed effect parameters, indicating significant variability in soil C concentration due to site and sampling location.

Soil C concentration tended to decline with increasing soil depth in all community types. In the both the upland and cypress soils, there was a significant decrease in soil C concentration in each segment for the top 0-20 cm, but no significant change below 20 cm depth. Similarly, the marsh soil exhibited a significant decrease in soil C concentration to a 30 cm depth, but no significant decrease below 30 cm. In bay swamp soils, C concentrations did not change in the upper 20 cm of the profile, but declined significantly below 20 cm, although there were no significant differences among layers from 20-50 cm. The model output and residuals are shown in APPENDIX G.

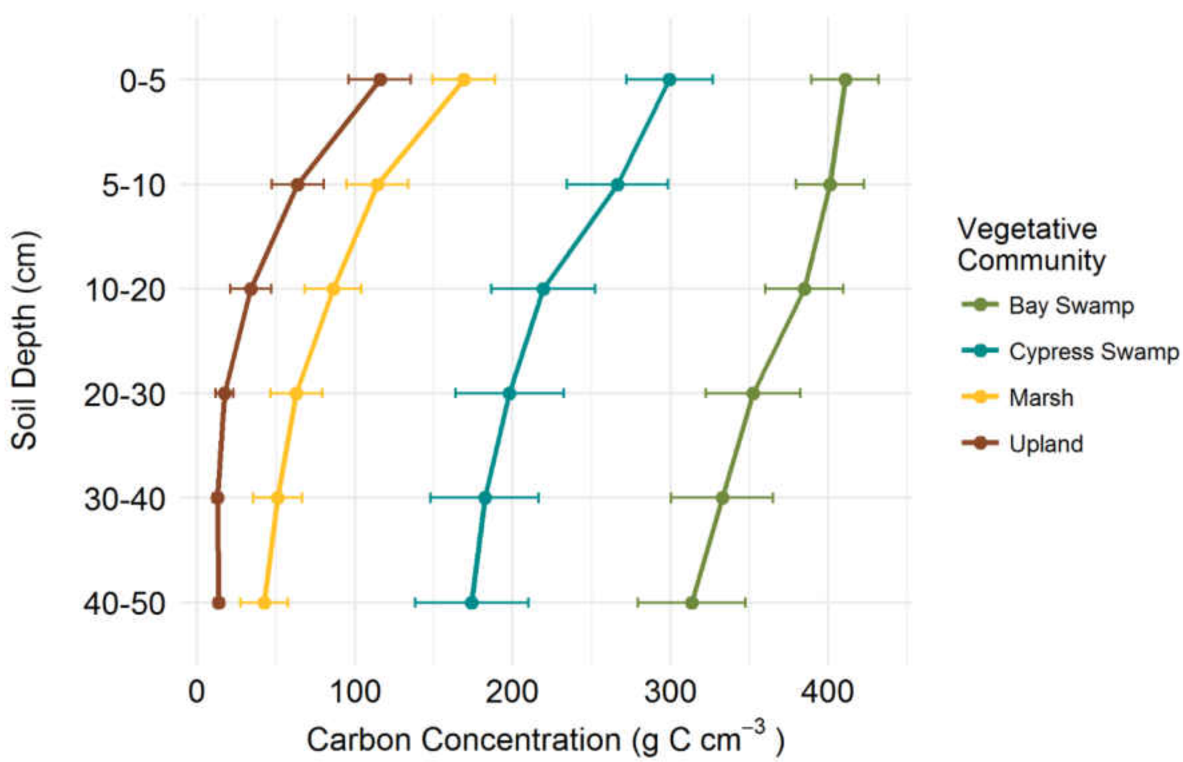


Figure 8. Mean soil carbon concentration by soil depth in three wetland communities and adjacent upland communities. Error bars are standard error of the mean.

The following function was the most plausible model for predicting soil C concentration from wetland ecological and hydrological variables at the study sites (Table 5):

glmmadmb(C_Conc ~ WD_avg*Pt_Com+Seg_Code*Pt_Com + (1|WL_ID/TR_ID/Pt_ID), data = SegData, family = "Gamma")

Where C_Conc is the soil C concentration (g cm^{-3}) of an soil core segment, WD_avg is the mean water depth, Pt_Com is the vegetative community, Seg_Code is the soil depth segment, WL_ID is the individual wetland sampled, TR_ID is the transect ID number, and Pt_ID is the core identification number. The model includes the individual effects of mean water depth, vegetative community, and soil depth as well as interactive effects between mean water depth and vegetative community and between soil depth and vegetative community. Mean water depth was a significant predictor of soil C concentration in bay swamp and marsh soils ($p < 0.001$) but not in upland or cypress swamp soils. The total random effect variance from sampling location embedded within wetlands and sampling transects (1.136) was similar to or greater than many estimates from the fixed effect parameters, suggesting that local site conditions had significant influence on variation of soil C concentration. The model output and residual plots are in APPENDIX H, along with the statistical model for soil C concentration at DWP.

Table 5. AICc output showing the dAICc value, degrees of freedom (df), AICc weight, and percentage deviance explained (Devex) of the top five models for soil carbon concentration. Table truncated to omit all other models with AICc weights < 0.001.

Model (Fixed Effects)	**dAICc**	**df**	**weight**	**Devex**
C_Conc ~ WD_avg*Pt_Com+Seg_Code*Pt_Com	0.0	32	1	13.55
C_Conc ~ WD_avg*Pt_Com*Seg_Code	25.9	52	< 0.001	13.75
C_Conc ~ Pt_Com*Seg_Code	29.2	28	< 0.001	13.16
C_Conc ~ WD_avg*Pt_Com+Seg_Code	51.0	17	< 0.001	12.68
C_Conc ~ WD_avg*Pt_Com*WD_SD+Seg_Code	54.3	25	< 0.001	12.83

In marsh and bay swamp soils, soil C concentration increased with increasing mean water depth (Fig. 9). There was no significant relationship between mean water depth and soil C concentration in cypress swamp or upland soils. There is also no discernable pattern between mean water depth and soil depth.

In bay swamps, there was a dramatic shift in soil C concentration after mean water depth crosses the 0 m (ground surface) threshold towards high C concentration at all depths (with a few exceptions). Although there are some bay swamp segments with low C concentration, these low values are not observed in locations with positive mean water depths. There also appears to be little change in bay swamp soil C concentration with increasing depth after crossing the 0 m threshold.

The increase in soil C concentration with increasing water depth is more gradual in marsh soils, but again the highest C concentrations only occur when mean water levels are above the ground surface. In cypress swamps, soil C concentration does not vary significantly with mean water depth, however almost all of the cypress sampling locations had mean water depths greater than 0 m. contrasting with bay swamps, cypress swamps exhibit a wide range of soil C concentrations with the same and deeper mean water depths. Soil C concentration in upland communities also does not vary significantly with mean water depth, but occupies the different end of the hydrologic spectrum than cypress swamps, with very few upland samples collected from locations with mean water levels greater than the ground surface.

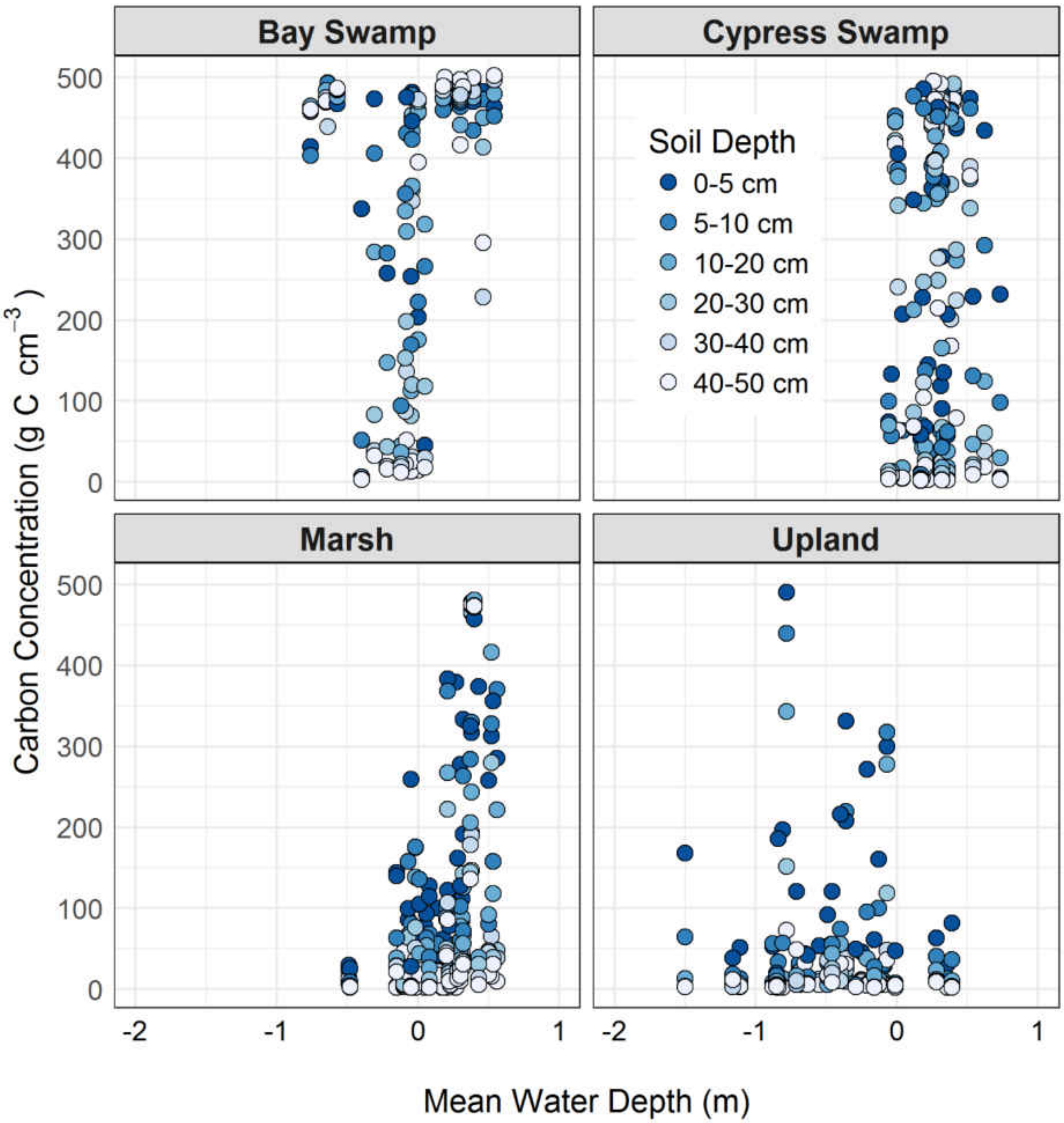


Figure 9. Soil carbon concentration along a hydrologic gradient and by soil depth in three wetland communities and adjacent upland communities.

Soil Bulk Density

Mean soil bulk density for the top 50 cm was significantly different across all wetland communities: marsh soils had the highest mean bulk density, followed by cypress and bay swamp soils, respectively (Fig 10). Bulk density in upland soils was not significantly different from that in marsh soil but was significantly higher than bulk density in both bay and cypress swamp soils. Bulk density was heavily skewed in bay swamps towards higher densities and skewed in the uplands towards lower densities, with a few low density outliers. The combined random effect variance (0.17) is smaller than the community estimates, indicating more variation in soil bulk density is explain by overall differences among vegetative communities rather than variance in local site conditions. The model output and residuals are shown in APPENDIX I.

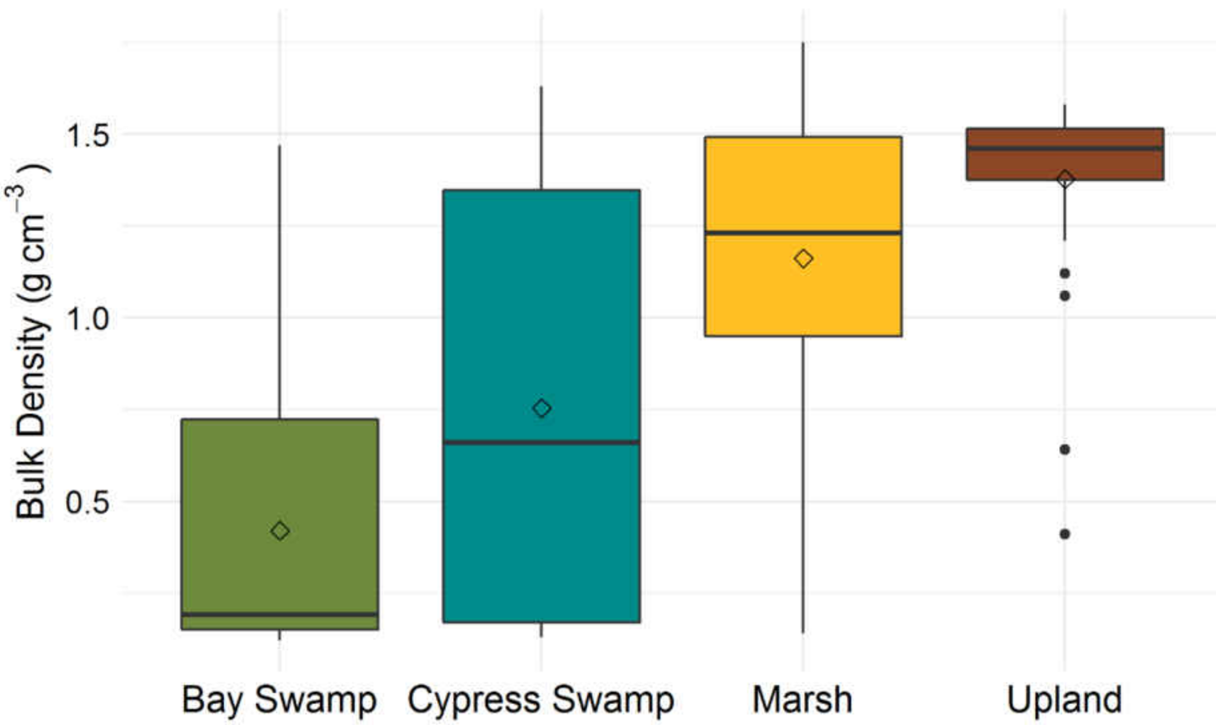


Figure 10. Soil bulk density for 0-50 cm depth in three wetland communities and adjacent upland communities. Diamonds represent the means, horizontal bars the medians.

At each depth segment, mean soil bulk density is greatest in the upland communities, followed by marsh, cypress swamp, and bay swamp (Fig. 11). This pattern is the inverse of that seen with soil C concentration in each community. Mean soil bulk density is significantly different in each community type for the top 0-20 cm. At 20-50 cm, there is no significant difference between marsh and upland soils, but both have significantly higher mean soil bulk densities than the swamp soils. In each community type, mean soil bulk density increased with depth. In bay swamps, there was a significant increase in mean bulk density with increasing depth until 30 cm, but no difference among depths from 30-50 cm. In cypress swamp, marshes, and upland soils, mean bulk density increased significantly with increasing depth in the top 0-20 cm. At greater depths, there was no significant change in bulk density in upland soils but the greatest depths (40-50 cm) in cypress swamp and marsh soils had significantly higher densities than the 20-30 cm depth. The combined random effect variance (0.16) is again lower than the estimates for individual community types and soil depths. The model output and residuals are shown in APPENDIX J.

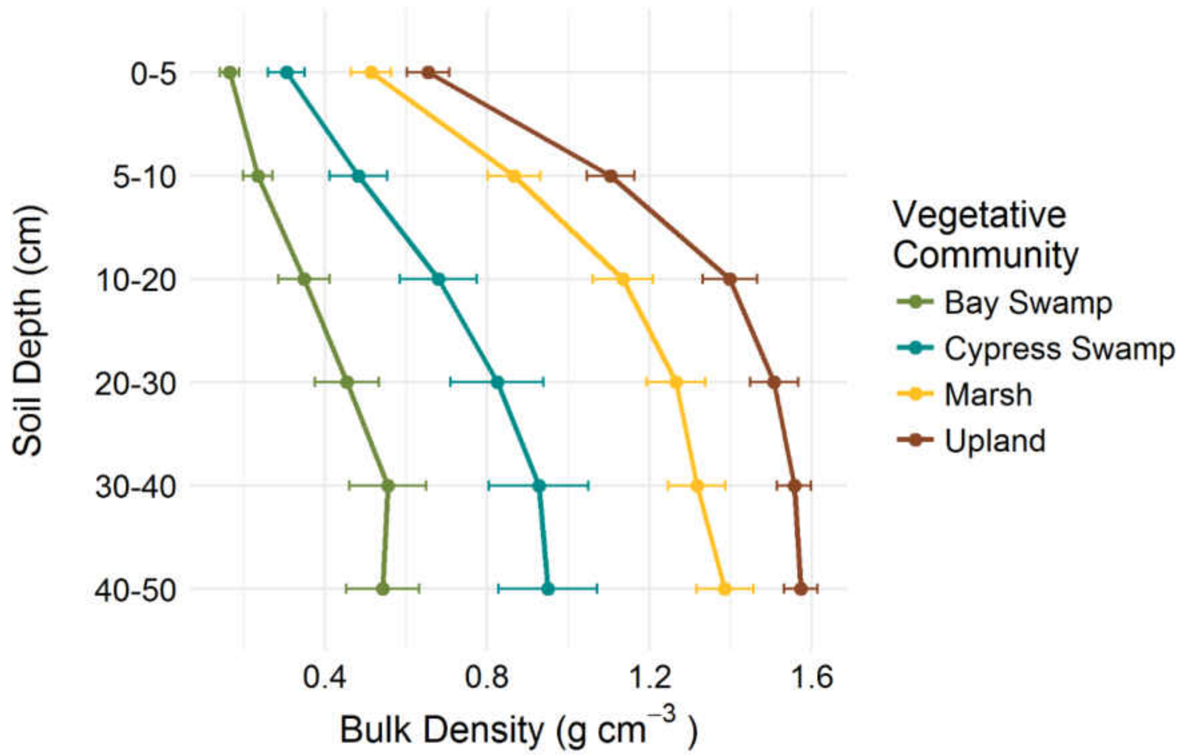


Figure 11. Mean soil bulk density by soil depth in three wetland communities and adjacent upland communities. Error bars are standard error of the mean.

Soil Carbon Stock

Soil C stock in the upper 50 cm of soil was significantly higher in bay swamps than all other communities and lowest in upland communities (Fig. 12). Mean soil C stocks in cypress swamps and marshes falls between the mean stocks in bay swamp and upland but are not significantly different from each other. This pattern is similar to the one seen between soil C concentration and vegetative community. Soil C stock was mostly normally distributed in all communities, but more was more variable in cypress swamps and marshes when compared with bay swamp and upland communities. The combined random effect variance (0.09) is an order of magnitude smaller than the estimates for the individual vegetative communities. The model output and residuals are shown in APPENDIX K.

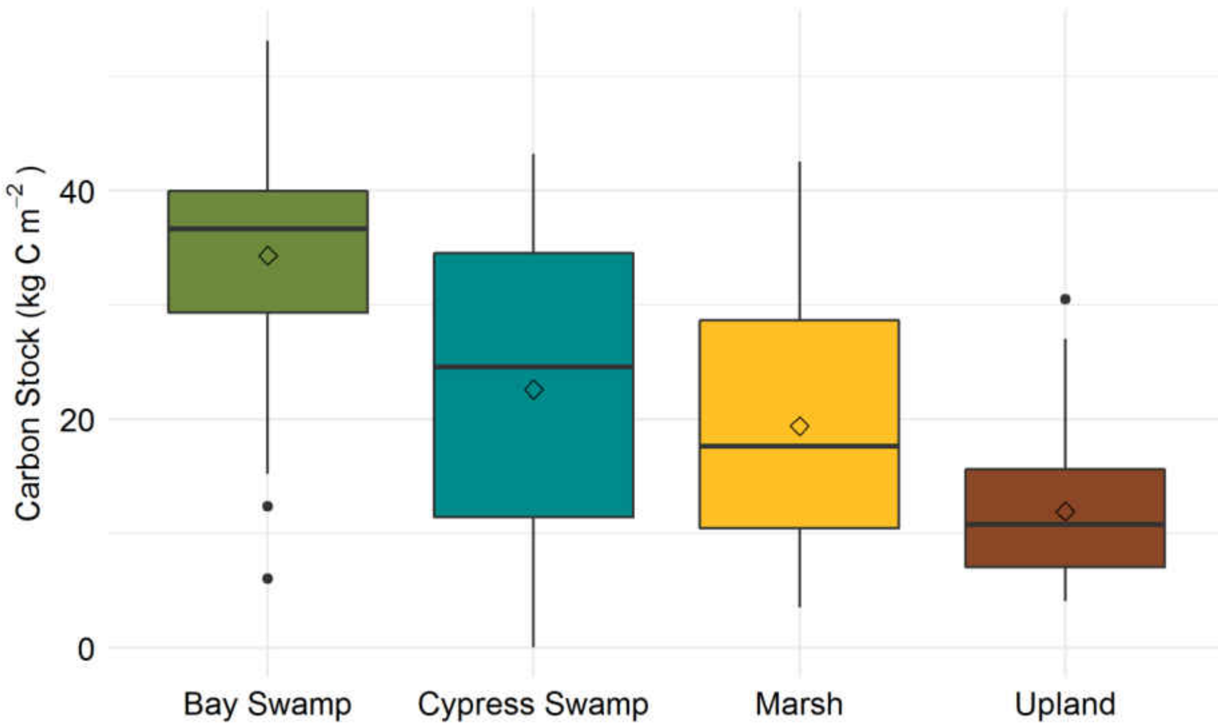


Figure 12. Soil carbon stock for 0-50 cm depth in three wetland communities and adjacent upland communities. Diamonds represent the mean, horizontal bars the median.

At each soil depth except the 0-5 cm depth, bay swamps had the highest mean soil C stock, followed by cypress swamps, marshes, and uplands (Fig. 13). In the top 5 cm, mean C stock was significantly higher in bay than cypress swamps, but did not differ significantly among other communities. In the 5-10 cm depth, mean soil C stock was significantly lower in upland soils compared with wetlands soils, but the wetland soils did not differ from each other. At 10-20 cm, cypress swamps were is not significantly different from either bay swamps or marshes. At 20-50 cm, mean soil C stock was different among all communities except between cypress swamp and marsh soils.

In each wetland type, there was an increase in mean soil C stock between 5-10 cm and 10-20 cm with a decrease in mean soil C stock with increasing soil depth below 20 cm. In bay swamps, there was a significant increase in mean soil C stock from 0-5 cm to 5-10 cm. Mean C stock in the top 10 cm was significantly less than mean C stock at deeper layers, but there is not much significant change in mean C stock in the 10-50 cm segments. In cypress swamps, mean C stock at 10-20 cm is significantly higher than mean C stock at all other depths. After 20 cm, mean C stock resembles the values found in the upper 10 cm. A similar pattern in mean C stock appears in marsh soils. Mean soil C stock in upland communities did not exhibit the increase at 10-20 cm that occurred in the wetlands, but exhibits an over decrease in mean C stock with depth. The combined random effect variance (0.31) is on par with some of the fixed effect parameter estimates, indicating soil C stock varied greatly by local site conditions. The model output and residuals are shown in APPENDIX L.

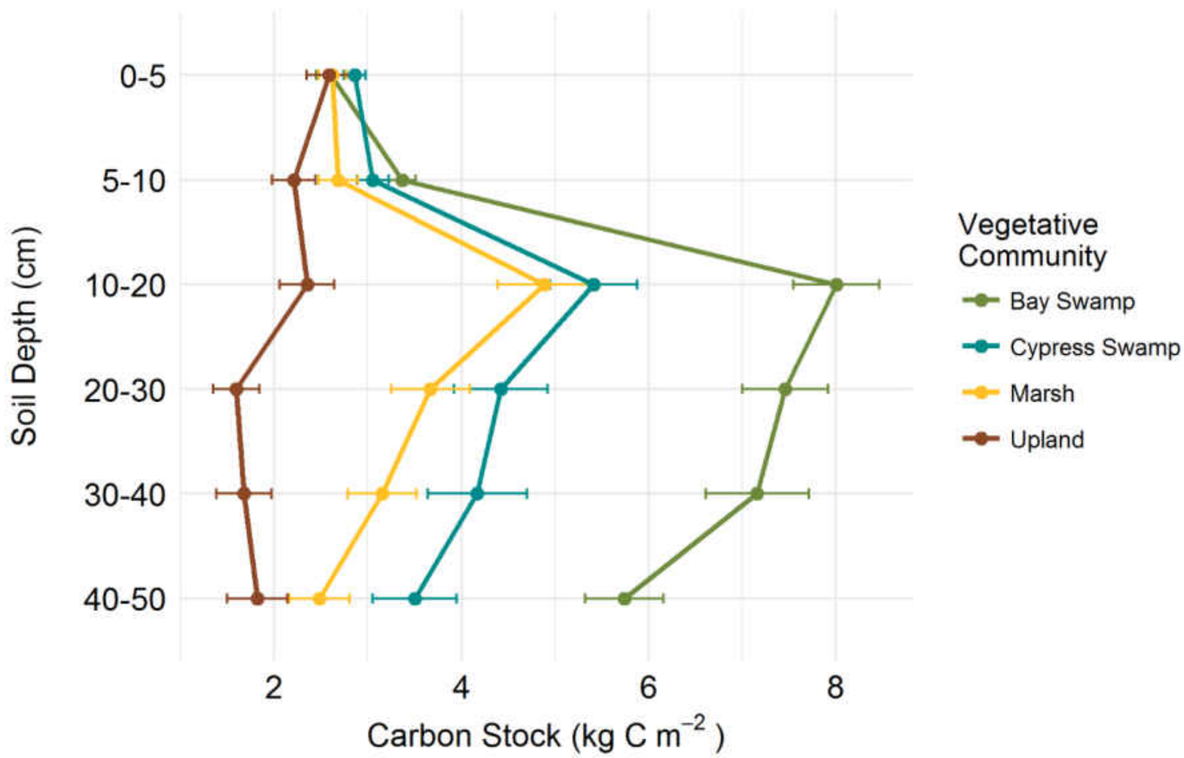


Figure 13. Mean soil carbon stock by soil depth in three wetland communities and adjacent upland communities. Error bars are standard error of the mean.

The function for the most plausible model predicting soil C stock from wetland ecological and hydrological variables was:

```
glmmadmb(C_Stock ~ WD_avg*Pt_Com+Seg_Code*Pt_Com + (1|WL_ID/TR_ID/Pt_ID),
data = SegData, family = "Gamma")
```

Where C_Stock is the total soil C (kg C m^{-2}) of an individual sample (see above for definitions of other variables) (Table 6). This model configuration is identical to that of the model for soil C concentration and includes the individual effects of mean water depth, vegetative community, and soil depth as well as interaction effects between mean water depth and vegetative community and between soil depth and vegetative community. The total random effect variance from individual wetlands, transects, and soil cores (0.120) was an order of magnitude lower than the random effect variance for the glmm predicting soil C concentration (1.136) indicating that the fixed effects of water depth, vegetative community, and soil depth explained more variation in soil C stock and that less variation in soil C stock was attributed to local site conditions. This model also fits the data better than the soil C concentration model, as shown by the model residual plots. The model output, residuals plots, and the statistical model are in APPENDIX M.

Table 6. AICc output showing the dAICc value, degrees of freedom (df), AICc weight, and percentage deviance explained (Devex) of the top five models for soil carbon stock. Table truncated to omit all other models with AICc weights < 0.001.

Model (Fixed Effects)	dAICc	df	weight	Devex
C_Stock ~ WD_avg*Pt_Com+Seg_Code*Pt_Com	0.0	32	1	18.88
C_Stock ~ Pt_Com*Seg_Code	16.7	28	< 0.001	18.17
C_Stock ~ WD_avg*Pt_Com*Seg_Code	19.1	52	< 0.001	19.58
C_Stock ~ WD_avg*Seg_Code	97.3	16	< 0.001	15.24
C_Stock ~ WD_avg*Pt_Com*WD_SD+Seg_Code	100.7	25	< 0.001	15.68

Mean water depth was only a significant predictor of C stock in marsh soils. The pattern of increasing soil C stock with increasing mean water depth in marshes looks very similar to the pattern with soil C concentration (Fig. 14). In the bay swamp, cypress swamp, and upland communities, soil C stock distribution did not vary significantly with mean water depth.

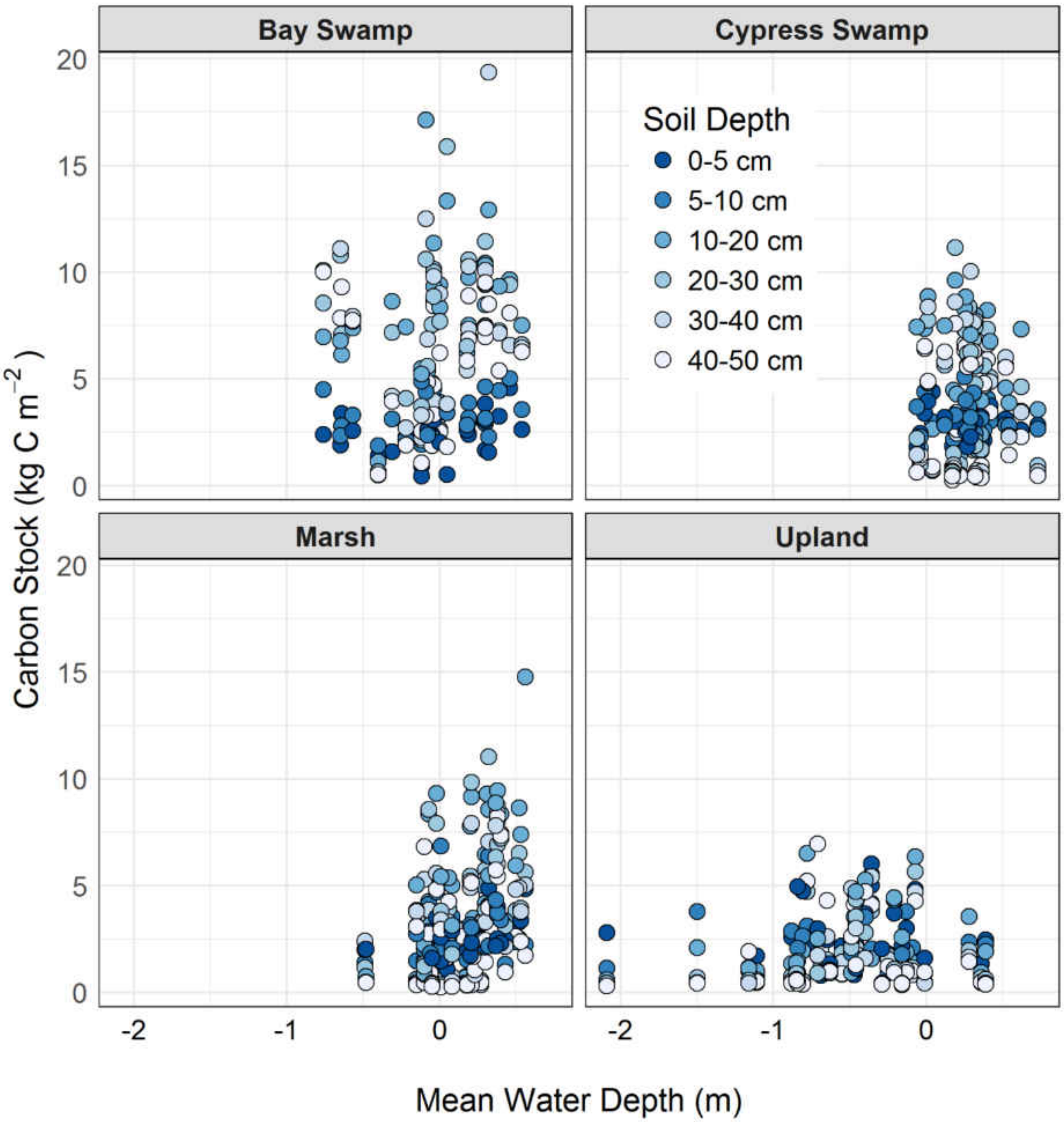


Figure 14. Soil carbon stock along a hydrologic gradient and by soil depth in three wetland communities and adjacent upland communities.

Wetland Contribution to Soil Carbon Stocks at the Landscape Scale

To estimate site-wide soil C storage in the upper 50 cm for each wetland community, I multiplied the mean soil C stock (kg C m^{-2}) by the areal coverage of each community at the Disney Wilderness Preserve (Table 7). Even though bay swamps covered an area approximately equal to the area covered by cypress swamp and marshes, bay swamp soils stored approximately 66% more total C than each of the other wetland types, and had the greatest concentration of soil C in the top 50 cm, 340 Mg ha^{-1}. Total soil C was similar in cypress swamp (230 Mg C ha^{-1}) and marsh soils (195 Mg C ha^{-1}). Combined, these three wetland communities stored approximately 260 Mg ha^{-1} and a total of approximately 204,970 Mg of C in the upper 50 cm of soil at the site.

Table 7. Estimated total carbon stored in the upper 50 cm of soil in three wetland communities at the Disney Wilderness Preserve.

	Total Carbon (kg)		
Soil Depth	**Bay Swamp**	**Cypress Swamp**	**Marsh**
0-5 cm	7,067,680	6,835,400	7,519,400
5-10 cm	8,039,960	7,301,450	7,700,210
10-20 cm	21,701,680	12,939,460	14,017,080
20-30 cm	20,211,180	10,566,190	10,535,770
30-40 cm	19,403,600	9,963,910	9,049,110
40-50 cm	15,555,400	8,367,390	7,120,470
Total	**93,055,980**	**55,973,800**	**55,942,040**

Percent soil C was much higher in the wetland community types sampled in this study than in upland soils sampled at the site by Becker (2011) (Table 9). The wetland-upland ecotones sampled in this study also had higher percentage soil C in the upper 30 cm compared to the uplands sampled by Becker, however mean percentage C is approximately equal for both in the 30-60 cm segment.

Table 8. Percentage soil carbon in communities at the Disney Wilderness Preserve.

	Community	Percentage Carbon	
		0-30 cm	30-60 cm
Becker 2011	**Scrubby Flatwoods**	0.8	0.3
	Longleaf Pine Flatwoods	1.7	1.8
	Pasture in Restoration	2.5	0.9
	Unrestored Pasture	3.1	1.4
	Slash Pine Flatwoods	4.2	0.8
This Study	**Bay Swamp**	38.8	33
	Cypress Swamp	27.0	17.5
	Marsh	10.8	4.5
	Upland Ecotone	5.5	1

To compare my soil C estimates with those for uplands sampled by Becker (2011), I estimated soil C stock in the wetlands from this study by assuming consistent soil C stocks from the last segment (40-50 cm) for the remaining 50-90 cm (Table 10). I then calculated the proportion of land area and total soil C stock for each wetland community and the combined uplands from Becker's study. These calculations exclude approximately 1,000 ha of ecosystems at DWP including floodplain swamps, scrub, and open water. Bay swamps continued to be the greatest soil C pool and alone stored approximately 40% as much soil C in the upper 90 cm as the uplands sampled by Becker. Together, the sampled wetland communities cover approximately 22% of area of sampled communities at DWP, yet store approximately 47% of the total soil C to a 90 cm depth.

Table 9. Coverage and estimated soil carbon storage in the upper 90 cm of soil for three wetland communities (this study) and upland communities (Becker 2011) at the Disney Wilderness Preserve.

Community	**Cover (ha)**	**Proportion of Cover**	**Carbon (kg)**	**Proportion of Total Site Carbon**
Bay Swamp	271	7.5%	155,277,580	22.3%
Cypress Swamp	239	6.6%	89,443,360	12.8%
Marsh	287	7.9%	79,660,400	11.4%
Uplands (Becker 2011)	2823	78.0%	371,861,000	53.4%
Total	3620		696,242,340	

DISCUSSION

Hydrology and Soil Carbon

Results from this study showed mixed support for the prediction that greater mean water depth and longer mean hydroperiod would be associated with increased C storage in wetlands soils. Average water depth was an important explanatory factor for soil C concentration and total C stock, and its effect was comparable to or smaller than the effects of wetland community type. Hydroperiod was likewise a significant, yet weak, explanatory variable for soil C concentration and total C stock even though it was not included in the top models. As expected, wetlands had both higher soil C concentration and stock than the neighboring upland communities, which is consistent with literature findings that hydric wetland soils accumulate and store more soil C than drier upland soils (Mitra et al. 2005; Bridgham et al. 2006; Bernal and Mitsch 2008; Vasques et al. 2010; Ross, Grunwald, and Myers 2013; Xiong et al. 2014). Also, soil C concentration and stock was higher in cypress swamps, which had higher mean water levels and longer mean hydroperiods, than in marshes, with lower mean water levels and shorter mean hydroperiods. However, bay swamps, which had by far the greatest C stocks, did not have higher water levels than the other wetland types. Furthermore, due to high variability in soil C stock within wetlands, there was not always a strong relationship between water depth and soil C storage within wetland types, indicating that local factors other than water depth had an equally or more important influence than average water depth.

Another recent study in Florida showed that shorter hydroperiod swamps held 50-60% less C than longer hydroperiod wetlands (Lewis and Feit 2015). However, the effect of hydroperiod on soil C stock in that study disappeared when hydroperiod was calculated over longer time frames (> 2 years) and water elevation was interpolated to daily values from 1-2 readings per month, which may have introduced significant error into hydroperiod estimates. Results from my study give general support for the finding that more saturated locations within wetlands have higher soil C content (Bernal and Mitsch 2008; Sahuquillo et al. 2012; Hunt et al. 2014; Nyamadzawo et al. 2015), but they also demonstrate that local and site factors in complex landscapes can make it difficult to predict soil C storage from water level or hydroperiod data alone.

For soil C storage, the degree of soil saturation could be as important, if not more important, than duration of inundation based on the fact that water depth was a better predictor of soil C concentration and stock than hydroperiod. Oxygen diffuses about 10,000 times more slowly through water than through air and under saturated conditions soil pore space becomes waterlogged and anoxic (Reddy and DeLaune 2008). Deeper water at a site creates a larger barrier for oxygen to reach the soil-water interface compared to shallower water, an effect that could be as important as prolonged inundation. Mean water depth also represents water level relative to the soil surface instead of a duration of inundation. Even if the soil is not completely inundated, the soil column could still be partially or C saturated, which may sufficiently impede oxygen diffusion and thus aerobic decomposition. Additionally, water depth may be a more robust variable than hydroperiod for this site. Variation in water depth is more evenly distributed across community types than variation in hydroperiod and local site conditions more greatly influenced variation in hydroperiod than in water depth.

Vegetative Community and Soil Carbon

There was strong support for the prediction that there would be a significant effect of wetland plant communities and the interactive effect of on hydrology and vegetative community on soil C storage. While the relationship between water depth and soil C storage was not significant in every community type, vegetative community alone was a significant predictor of both soil C stock and soil C concentration. The strong effect of vegetative community on soil C related to the fact that the current distribution of vegetative communities in the landscape is a response to long term ecological conditions, including hydrologic regime, soil conditions, and fire regime (Mitsch and Gosselink 2007; Reddy and DeLaune 2008; Taggart et al. 2011; Hu et al. 2016; Krishnaraj et al. 2016).

Soil C sequestration in wetlands is the result of a number of complex factors, many of which are influenced directly or indirectly by hydrology, that affect the input of organic matter into the soil system and the decomposition of this organic matter (Turetsky 2004; Wang et al. 2010; Bernal and Mitsch 2012; Peralta et al. 2014; Jiao et al. 2014; Medvedeff, Inglett, and Inglett 2015). Litter input is an important driver of soil C storage, which varies by community type and hydrology (Day and Megonigal 1993; Grant, Desai, and Sulman 2012; Sulman et al. 2013; Wang, Li, and Zhang 2015) and litter quality can directly affect biomass decomposition rates (Chimney and Pietro 2006; Fanin and Bertrand 2016; Gao, Kang, and Han 2016; Hu et al. 2016).

Decomposition is often slower for woody plants, with more recalcitrant chemical and physical composition, compared to herbaceous plants (Godshalk and Wetzel 1977; Gallagher 1978; DeBusk and Dierberg 1984; Janssen and Walker 1999). Older plants and perennial species also decompose more slowly than younger or annual plants (Brock et al. 1982; Morris and Lajtha 1986). Marshes, which had the lowest soil C concentration and stock of the observed wetlands, are dominated by varied herbaceous cover that likely decomposes faster than woody swamp litter, leading to higher rates of organic matter turnover in marshes. A cypress community in the North Atlantic region produced significantly less litter compared to a mixed hardwood swamp dominated by broadleaf trees (*Quercus sp.*), and the cypress litter decayed faster than the mixed hardwood litter (Day 1979, 1982). Cypress litter had both higher P and N concentrations and a lower C:N ratio compared with mixed hardwood litter dominated by oak species. Oak litter also had higher concentrations of tannic acid and lignin, which are associated with slower decomposition. Bay tree species (*Magnolia virginiana, Gordonia lasianthus, Persea palustris*), which dominated bay swamps at DWP, also have waxy, broad leaves that may be more recalcitrant than litter in the cypress swamps or marshes. A study on different types of litter decomposing under common field conditions showed that decomposition rates were very similar for leaves *M. virginiana* (sweetbay) and *Q. virginiana* (live oak), another subtropical evergreen from more upland communities (Monk 1971). This slow litter decomposition could have contributed to more buildup of organic matter in the bay swamp systems, especially under flooded conditions.

Differences in decomposition rate are also supported by the varying organic soil depths in the different types of wetlands, because slower litter decomposition should translate to slower SOM turnover and accumulation of deeper organic soils over time. Marshes had the shallowest layers of organic matter, showing faster SOM turnover in these systems compared with other wetlands. Cypress swamps had more variable SOM depths that were often intermediary in depth between marsh and bay swamp soils. Bay swamps had the deepest organic layers and the highest soil C concentration and stock despite having the lowest mean water levels and shortest mean hydroperiod of the three wetland types. Accumulation of SOM could also have elevated the ground surface over time, so that using old surveying data to relativize water elevation to the soil surface resulted in errors in water depth calculations, potentially explaining the weak connections between mean water depth and soil C storage in bay swamps.

Estimating Soil Carbon at the Landscape Scale

My final prediction that models based on hydrologic variables such as average water depth and hydroperiod could be used to predict soil C storage at both the local and landscape scales within wetland types was not well supported by this study. The model results suggest that vegetative community type is sufficient for predicting soil C concentration and stock in this landscape and that long-term, detailed water level data was not a good predictor of C storage within wetland types. More complicated hydrologic models could potentially show a stronger relationship between hydropattern and soil C, but most sites lack the detailed hydrologic data that would be needed for such modelling. To estimate total wetlands soil C storage at the study site, I used the mean soil C stock values and the total cover of each community type on site with the assumption that incorporating hydrologic data into these estimates would not drastically change the soil C estimates.

The wetland soil C stock estimates in this study are slightly higher than, but comparable to, results from other Florida wetlands studies, which also showed that forested wetlands held more soil C than non-forested wetlands (Ross et al. 2013; Marín-Muñiz et al. 2014; Xiong et al. 2014). Estimates of soil C stock to 90 cm in wetlands in this study are higher than soil C stock estimates to 1 m depth for tropical wetlands by Köchy, Hiederer, and Freibauer (2015). Estimated soil C stock in DWP marshes (27.8 kg m^2) was slightly higher than the 95th percentile of tropical freshwater marshes, floodplain marshes, and other marsh wetlands (24.2 kg m^2). Estimated soil

C stock for DWP cypress swamps (37.4 kg m^2) was also higher than the 95th percentile of tropical swamp forest, flooded forest, and wooded wet swamps (33.8), but estimates for DWP bay swamps far exceeded this measure (57.3 kg m^2). However, projecting soil C stock further than a 50 cm depth may overestimate C storage to a certain extent. Observations from this study also support global observations of wetland storing a disproportionate amount of soil C compared to upland ecosystems (Mitra et al. 2005; Mitsch and Gosselink 2007; Bernal and Mitsch 2008; Vasques et al. 2010; Ross et al. 2013; Xiong et al. 2014).

Together, upland soils held the majority of soil C at DWP, but wetland soils stored a significant amount (47%) disproportionate to their cover. However, this study only measured soil C to a 50 cm depth and estimated soil C stock site-wide to a 90 cm depth. Mean SOM depths exceeded 50 cm in bay swamps (1.72 m ± 5.70), cypress swamps (0.67 m ± 0.47), and marshes (0.73 ± 1.02). Peat in swamps at DWP can be over 5 m deep, while upland soil organic layers tend to be relatively shallow (< 50 cm) (Helton 1996). Accounting for the C stored in deeper organic soils could show that bay swamps (and cypress swamps to a lesser extent) alone represent an even larger pool of C in this landscape.

Wetland Restoration and Soil Carbon

Restoring degraded wetlands is becoming a more common approach to protect wetland cover and reinstate more natural ecological structure and functioning. Wetland restoration often involves re-flooding drained wetlands to meet historic or target hydropattern regimes. Drained wetland soils can lose soil C more easily through aerobic microbial respiration and therefore often store less soil C long term (Bridgham, Updegraff, and Pastor 1998; Zedler and Kercher 2005; Fenner and Freeman 2011; Gao et al. 2014; Mastný et al. 2016). Given the link between hydrology and soil C storage in wetlands, re-flooding drained wetlands is often expected to restore soil C accumulation and storage.

Restoration ecology is a relatively young field and there are still many questions about the responses and success rates of wetland restoration. Responses of wetlands to restoration are highly variable, even at the regional scale (Bullinger-Weber et al. 2014). Increasing soil saturation can depress soil C mineralization rates (Lewis, Brown, and Jimenez 2014) and also protect soil C from mineralization through fire (Wade, Ewel, and Hofstetter 1980; Gunderson and Snyder 1994). In the short term, restored wetlands usually have less soil C than natural "reference" wetlands (Bishel-Machung et al. 1996; Kluber et al. 2014), but many show higher soil C stocks than degraded wetlands (Streeter and Schilling 2017).

Many studies observe relatively quick response of wetlands to restoration, noting slower C turnover and higher soil C storage within a few months or years (Tuittila et al. 1999; Waddington and Price 2000; Cagampan and Waddington 2008) or even restored wetland soil C matching reference wetlands within 8-10 years (Gao et al. 2014). But other studies observe that soil C in restored wetlands may not change significantly or catch up to natural wetlands in the short term (Theriot et al. 2013; Hunt et al. 2014). Longer time frames (> 10 years) may be needed to see significant changes in SOM quality and quantity (Mitsch and Wilson 1996; Moreno-Mateos et al. 2012; Wang et al. 2015; Mastný et al. 2016). Streeter and Schilling (2017) observed that SOC in restored wetlands still did not match SOC in natural wetlands 30 years after restoration. Soil C in the top 0-20 cm increased over 20 years after restoration, but SOC did not change in the deeper layers, demonstrating the slow process of accumulating organic soils in wetlands. Soil recovery also may not follow a linear trajectory towards reference conditions. Several studies observed significant soil OM increases in the first few years following restoration, with slower rates of

SOM increase after 15-20 years (Nair et al. 2001; Campbell, Cole, and Brooks 2002; Spieles, Coneybeer, and Horn 2006; Hernandez and Mitsch 2007; Ballantine and Schneider 2009) Although restoration project success or satisfactory results are not guaranteed (Zedler and Callaway 1999; Zedler 2000; Ballantine et al. 2012).

The restoration of DWP was in agreement with the goals for CERP. Some have speculated that increasing water storage in the landscape will lead to an increase in soil C storage. Based on the fact that water depth had a stronger relationship with soil C storage in marshes than swamps, increasing water storage in marshes at DWP could have resulted in a significant increase in soil C storage, but the impact of altered hydrology on swamp soils is less clear. Wetlands clearly store a disproportionately large amount of soil C at the landscape scale, however predicting soil C outcomes from altering water levels in any wetland system may prove difficult. The lack of pre-restoration data at DWP also makes assessing how much SOM was lost during drainage difficult. This site was drained and ranched for over 100 years, yet land use was not intensive compared to other degraded wetlands that are ranched or farmed and therefore may not have deviated greatly from current conditions in some less impacted wetlands. However, hydrologic restoration could have also increased the footprint of wetlands at DWP, increasing inundation of wetland-upland ecotone regions, leading to more soil C storage across the site.

Wetland Soil Carbon and Climate Change

The role of wetlands in the global C cycle continues to be elucidated and the fate of wetlands and their soil C stocks is uncertain in the face of global climate change. Wetlands are large C pools that can act as both sinks of CO_2 in vegetation and soil and sources of CH_4 from anaerobic respiration (Mitsch and Gosselink 2007, Reddy and DeLaune 2008). Climate change brings a suite of changes to drivers of soil C dynamics in wetlands including increasing temperatures, altered precipitation patterns, sea level rise and salt water intrusion, increased CO_2 concentrations, and increasing evapotranspiration (Wagner et al. 1996; Mitsch and Gosselink 2007; Chambers et al. 2014; Shand et al. 2017; Yao et al. 2017; Zhao et al. 2017). These drivers could indirectly impact water table levels, fire dynamics, and species composition patterns, which in turn will feed back into the other drivers. The response of wetlands across the globe to these varying drivers will be highly variable (Mitra et al. 2005; Erwin 2009).

Wetland soil C reserves are highly labile and will quickly decompose under drier conditions, meaning higher temperatures and lower water tables can result in faster SOC loss (Sulman et al. 2013; Lewis et al. 2014; Xiong et al. 2014; Zhang et al. 2016). Increased primary productivity from could initially compensate for soil carbon loss, possibly even leading to net carbon gains in wetland ecosystems or at the landscape scale in the short term (up to 100 years) (Sulman et al. 2013). Xiong et al. (2014) found that Florida soils have been a net C sink over past 40 years and estimate that Florida soils will continue to be a C sink given climate projections. However, if precipitation decreases, the combination of less rainfall and higher evapotranspiration (from increase primary productivity) could cause “catastrophic” soil C loses, working against goals to store more C in wetland soils (Orem et al. 2014).

CONCLUSIONS

Average water depth was an important explanatory factor for soil C concentration and total C stock, and its effect was comparable to or smaller than the effects of wetland community type. However, due to high variability in soil C stock within wetlands, there was not always a strong relationship between water depth and soil C storage within wetland types, indicating that local factors other than water depth had an equally or more important influence than average water depth. Water depth was a better predictor variable of soil C than hydroperiod, suggesting soil saturation is important for soil C storage and not just length of inundation. The influence of vegetative community type was comparable to or more powerful than that of water depth, likely due to differences in litter inputs among other factors, yet vegetative community distribution is also not independent of hydrologic regime. Wetlands at the study site stored approximately 47% of C to 90 cm depth while representing 22% of analyzed land cover. Among the wetland types, bay swamps stored the most C to 50 cm depth, holding over 20% DWP's soil C while representing approximately 7.5% of the area of interest. This does not account for the fact that bay swamps have the deepest average peat layers of the communities in this study and therefore are likely to hold a much larger proportion of the total soil C than was represented in this study. Future studies and better accounting for soils stocks in these systems should consider total depths of organic layers. Re-flooding the drained wetlands at DWP likely increased soil C storage within wetland boundaries, especially in marsh and bay swamps (which showed significant relationship between water depth and soil C), and also could have increased soil C storage across the site by expanding wetland boundaries and increasing wetland coverage across DWP, relative to pre-restoration wetlands.

APPENDIX A:
ORGANIC MATTER TO CARBON CONVERSION REGRESSION AND EQUATION

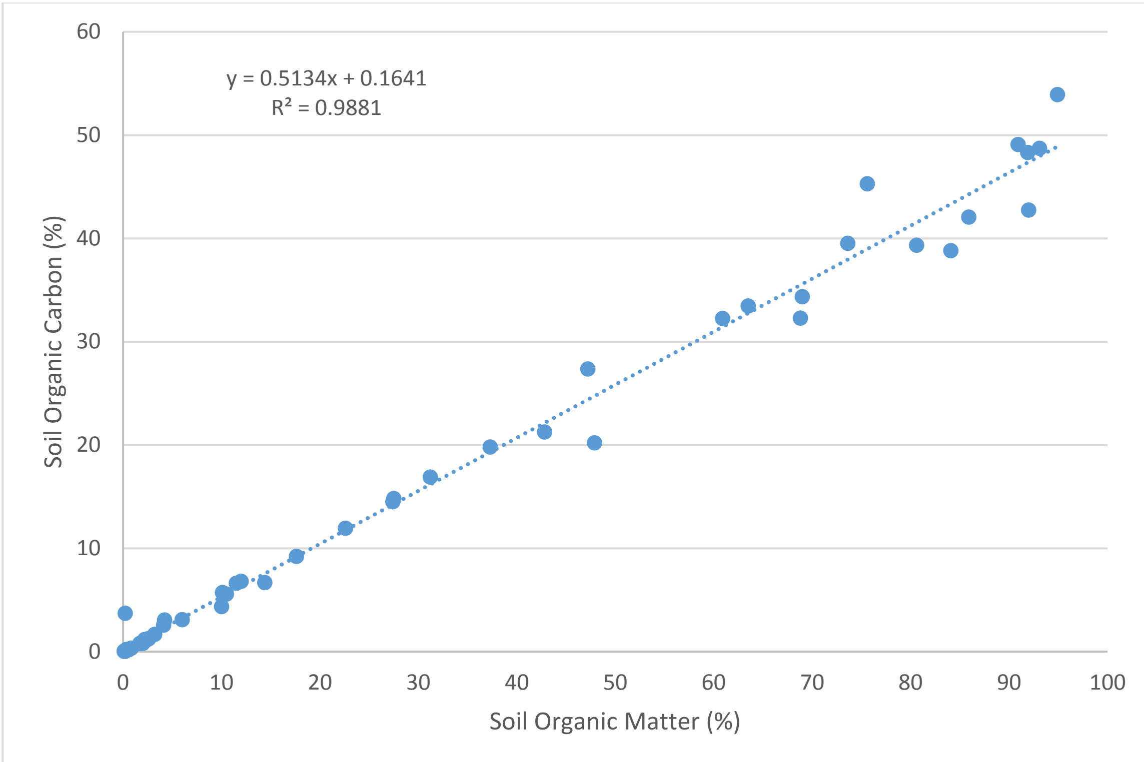


Figure A- 1. Relationship between soil organ matter and soil carbon and conversion equation.

APPENDIX B:
MANUAL MONITORING WELL AND CONTINUOUS RECORDER REGRESSIONS AND CONVERSION EQUATIONS

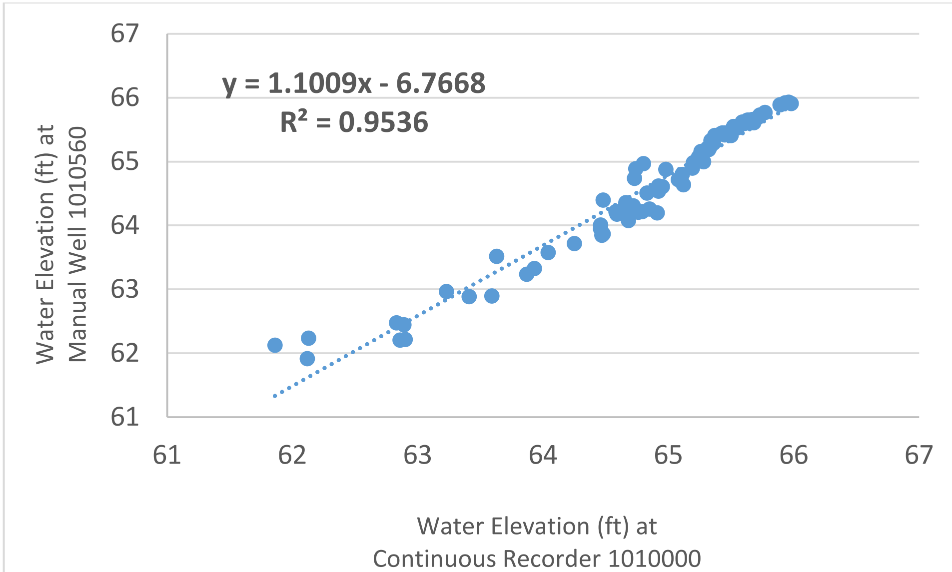


Figure B- 1. Water elevation regression of manual well 1010560 vs. continuous recorder 1010000.

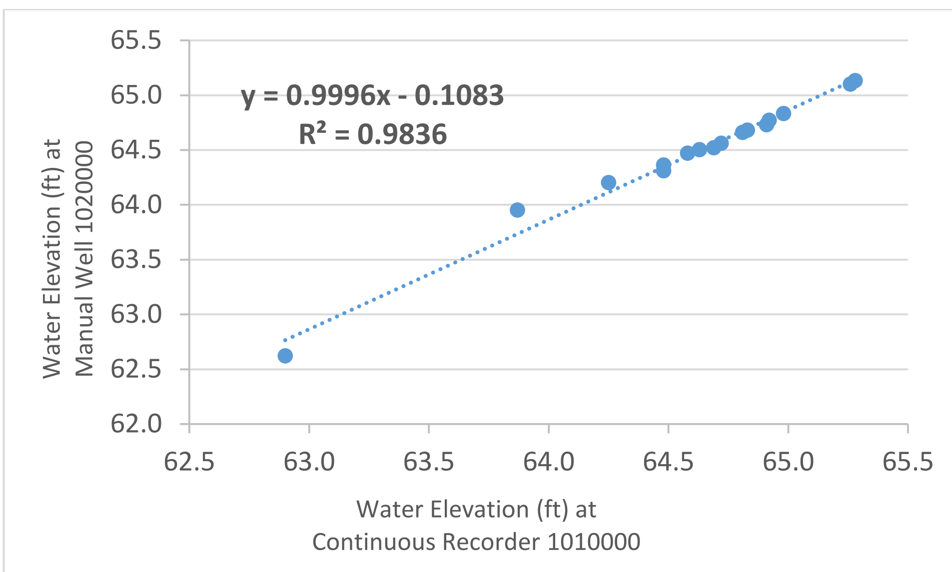


Figure B- 2. Water elevation regression of manual well 1020000 vs. continuous recorder 1010000.

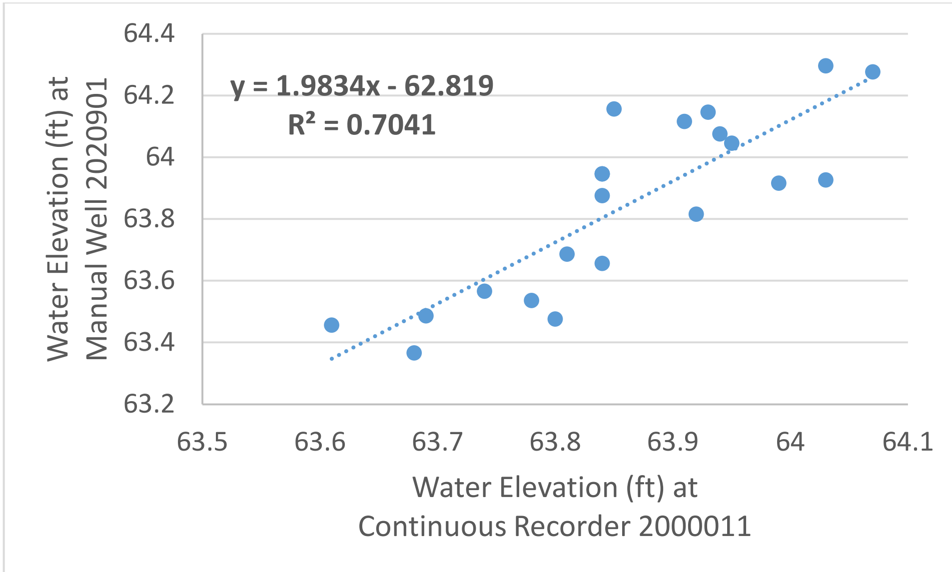


Figure B- 3. Water elevation regression of manual well 2020901 vs. continuous recorder 2000011.

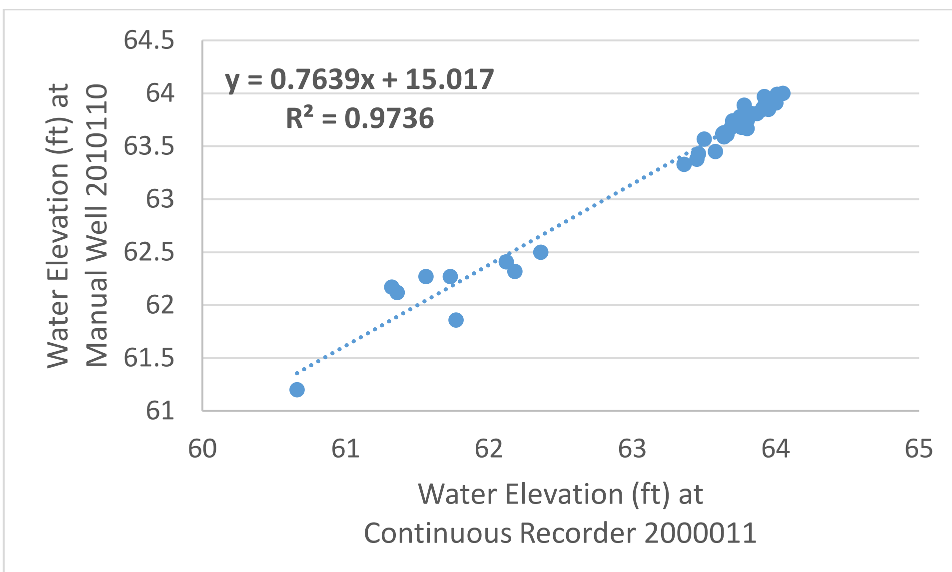


Figure B- 4. Water elevation regression of manual well 2010110 vs. continuous recorder 2000011.

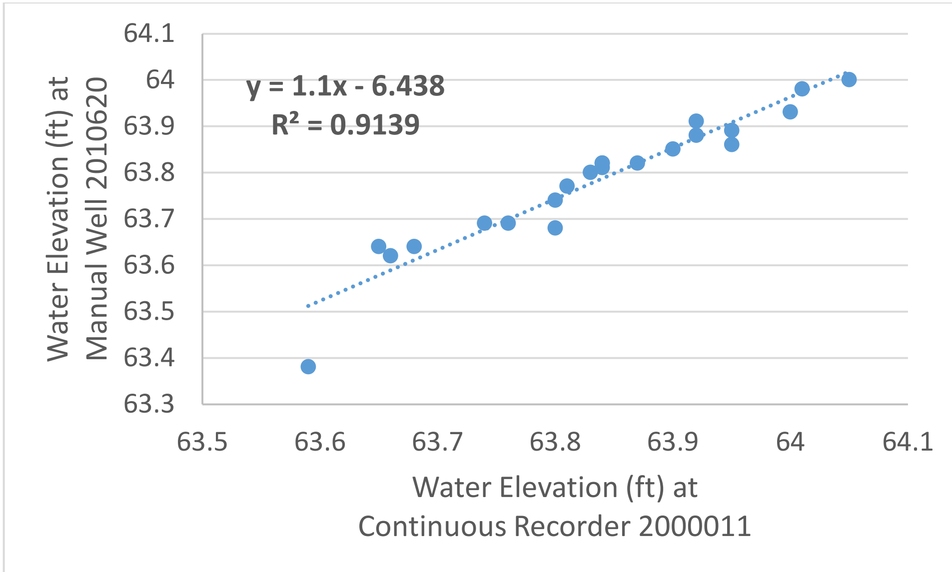


Figure B- 5. Water elevation regression of manual well 2010620 vs. continuous recorder 2000011.

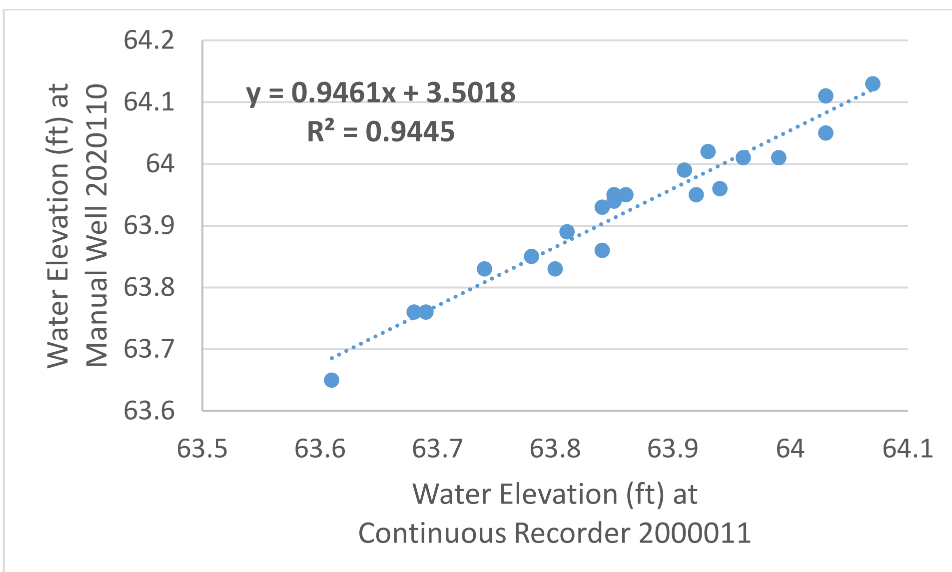


Figure B- 6. Water elevation regression of manual well 2020110 vs. continuous recorder 2000011.

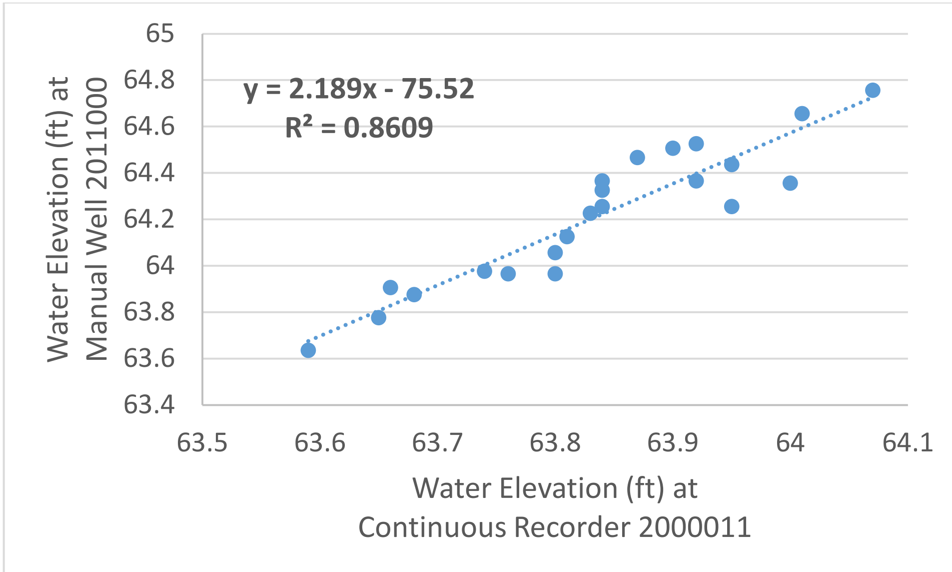


Figure B- 7. Water elevation regression of manual well 2011000 vs. continuous recorder 2000011.

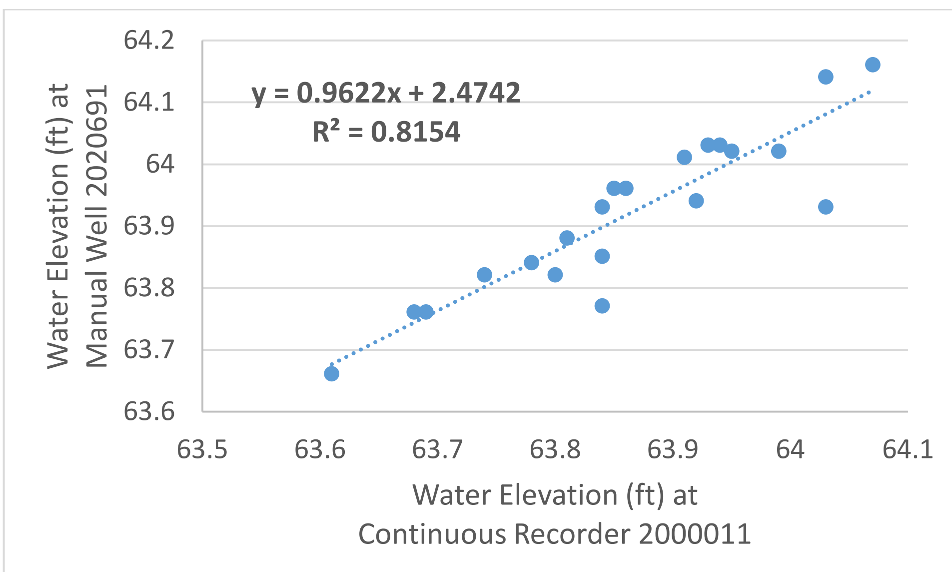


Figure B- 8. Water elevation regression of manual well 2020691 vs. continuous recorder 2000011.

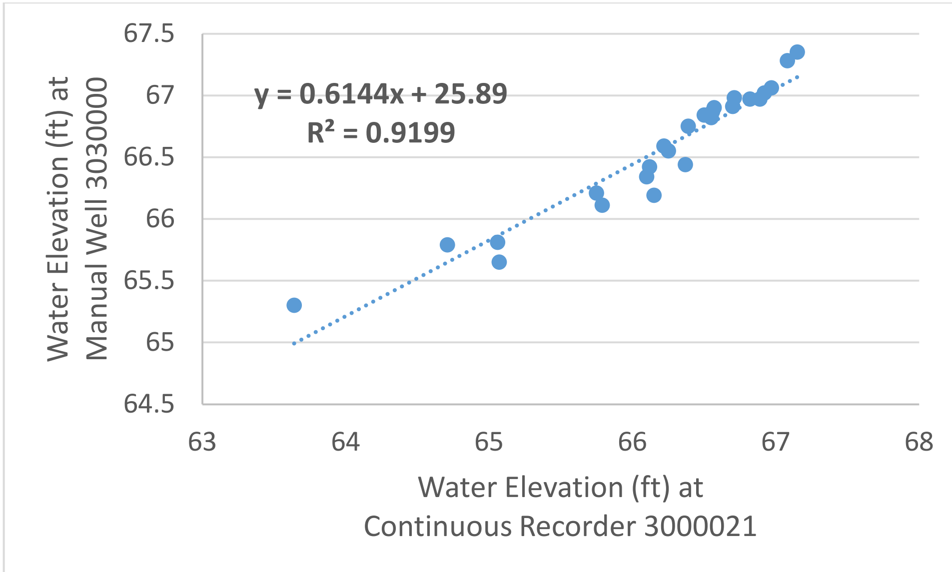


Figure B- 9. Water elevation regression of manual well 3030000 vs. continuous recorder 3000021.

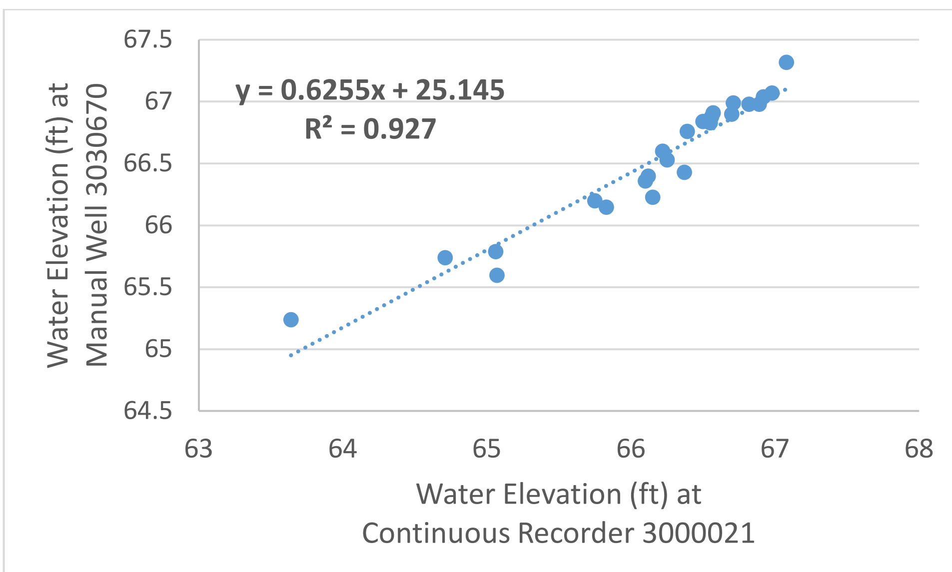


Figure B- 10. Water elevation regression of manual well 3030670 vs. continuous recorder 3000021.

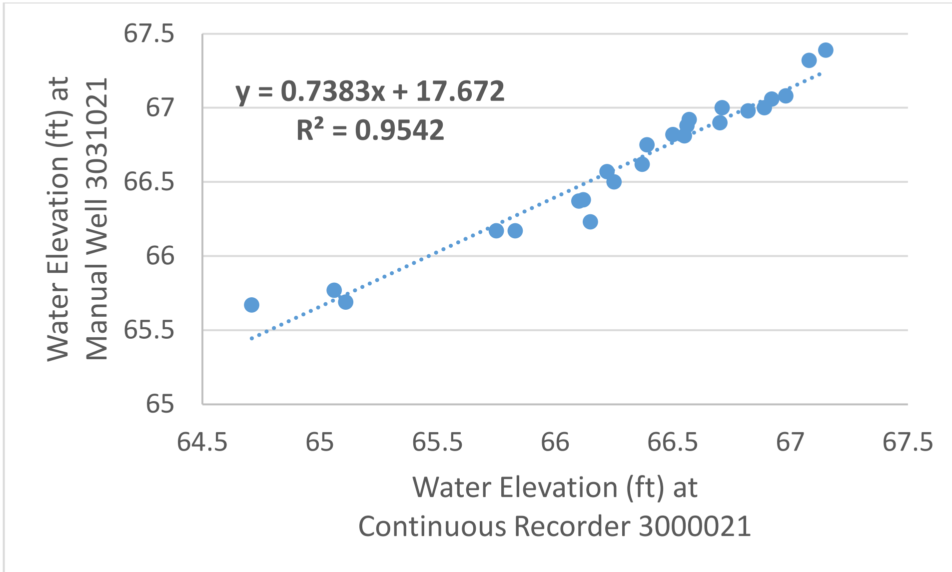


Figure B- 11. Water elevation regression of manual well 3031021 vs. continuous recorder 3000021.

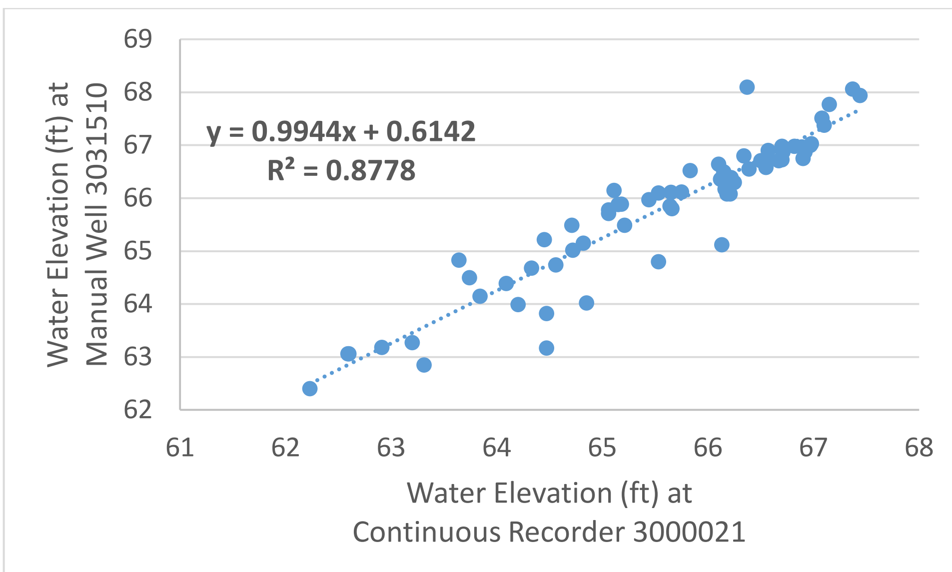


Figure B- 12. Water elevation regression of manual well 3031510 vs. continuous recorder 3000021.

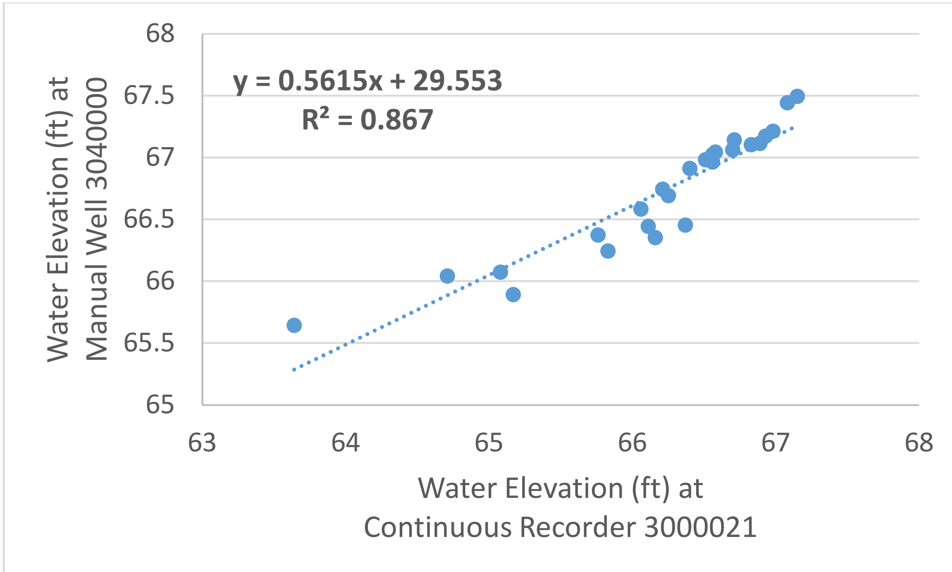


Figure B- 13. Water elevation regression of manual well 3040000 vs. continuous recorder 3000021.

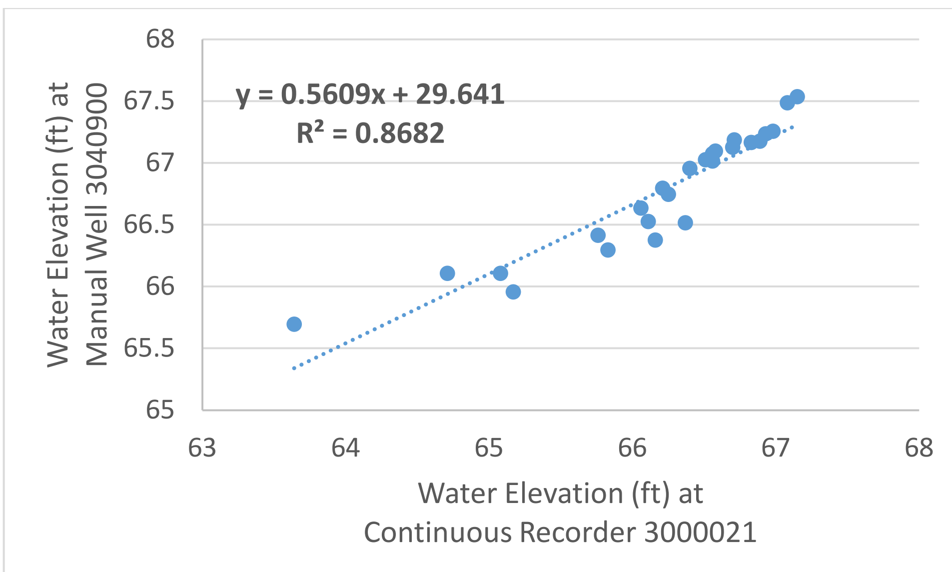


Figure B- 14. Water elevation regression of manual well 3040900 vs. continuous recorder 3000021.

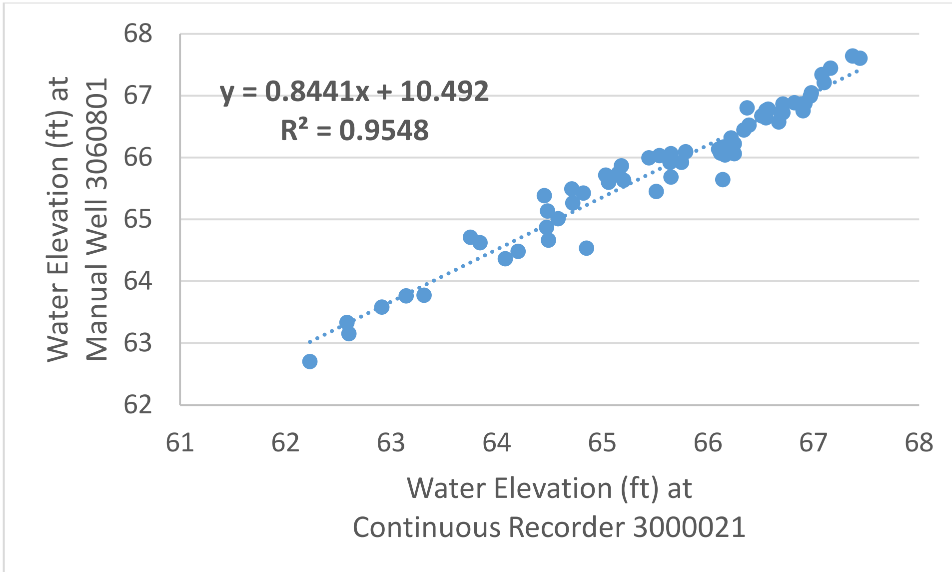


Figure B- 15. Water elevation regression of manual well 3060801 vs. continuous recorder 3000021.

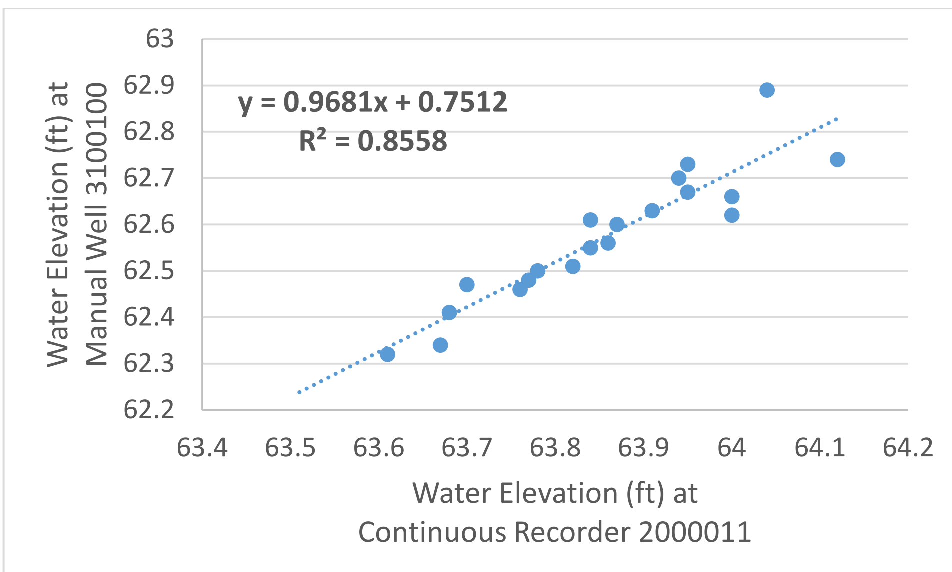


Figure B- 16. Water elevation regression of manual well 3100100 vs. continuous recorder 2000011.

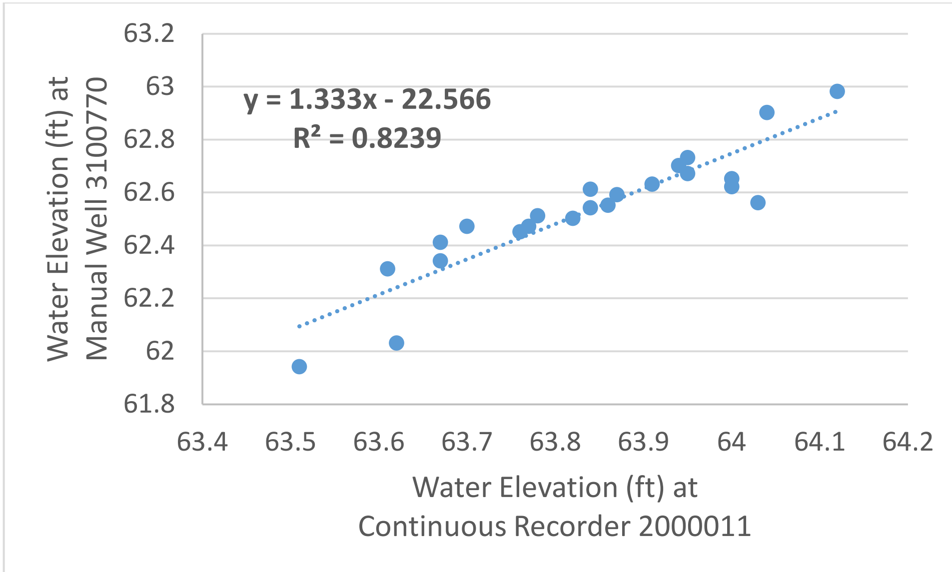


Figure B- 17. Water elevation regression of manual well 3100770 vs. continuous recorder 2000011.

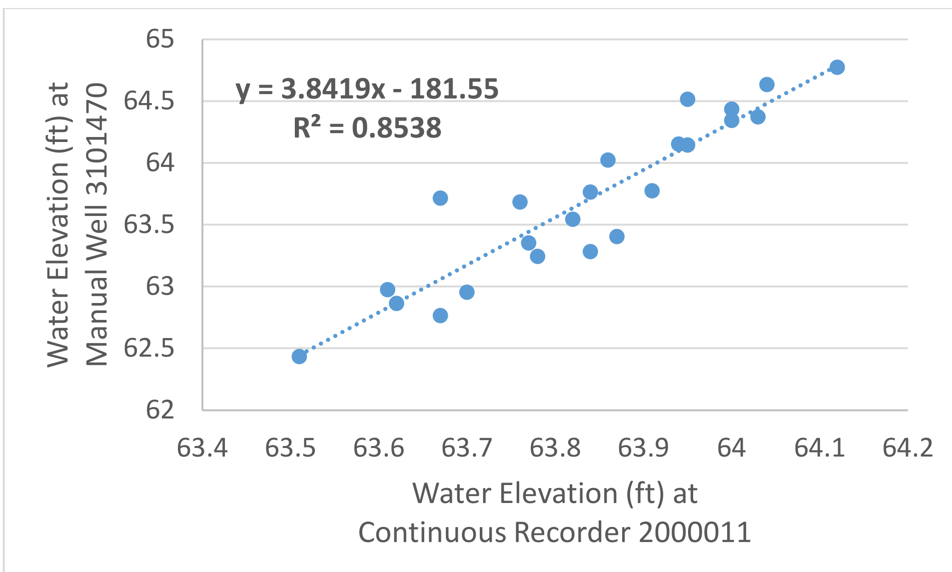


Figure B- 18. Water elevation regression of manual well 3101470 vs. continuous recorder 2000011.

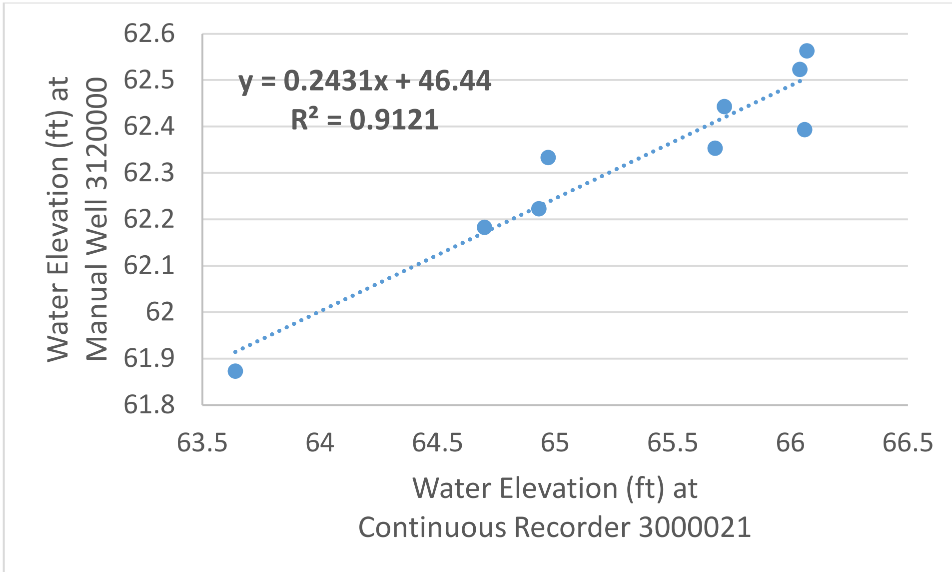


Figure B- 19. Water elevation regression of manual well 3120000 vs continuous recorder 3000021.

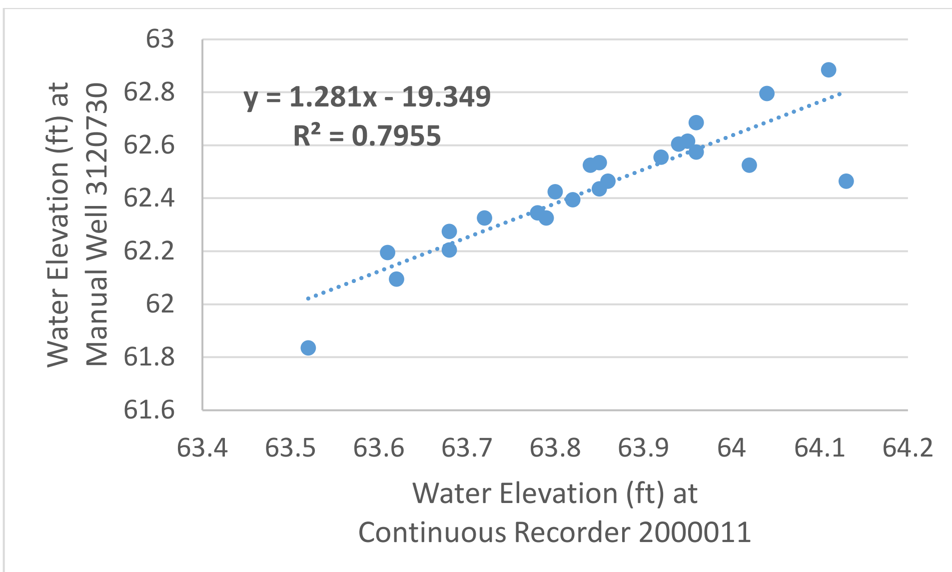


Figure B- 20. Water elevation regression of manual well 3120730 vs. continuous recorder 2000011.

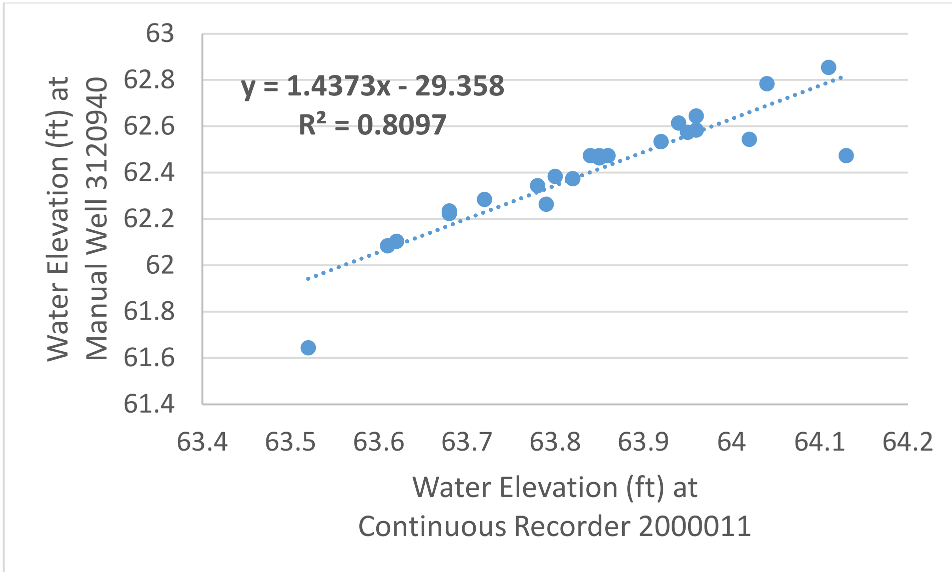


Figure B- 21. Water elevation regression of manual well 3120940 vs. continuous recorder 2000011.

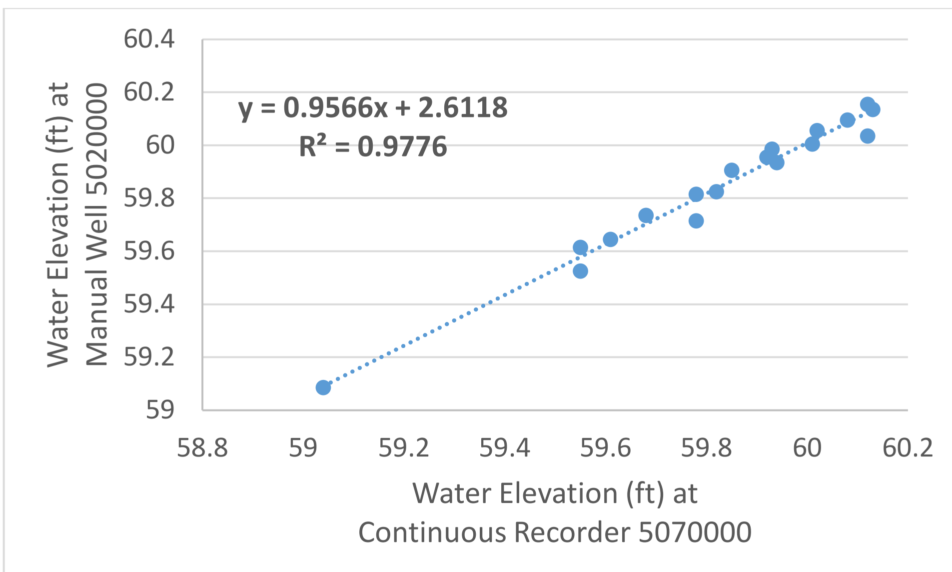


Figure B- 22. Water elevation regression of manual well 5020000 vs. continuous recorder 5070000.

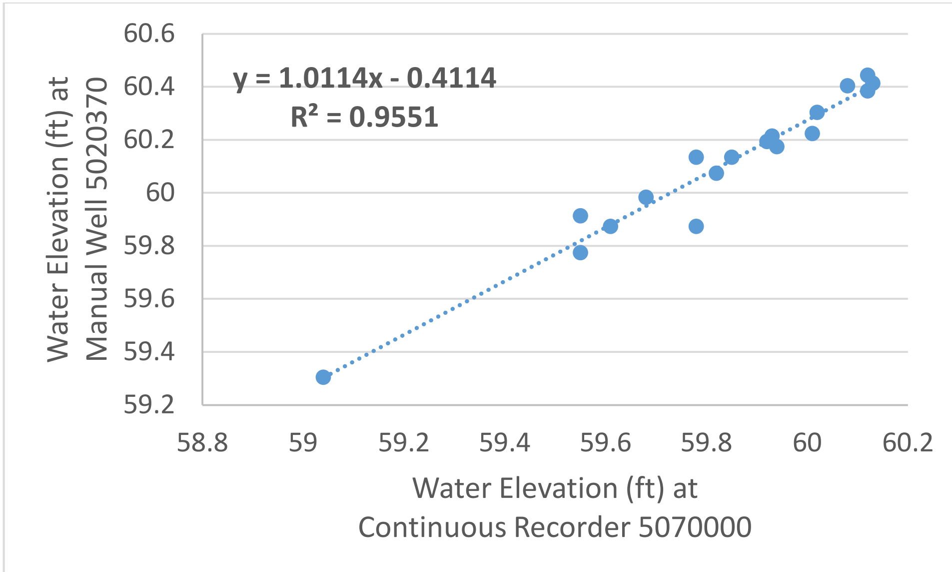


Figure B- 23. Water elevation regression of manual well 5020370 vs. continuous recorder 5070000.

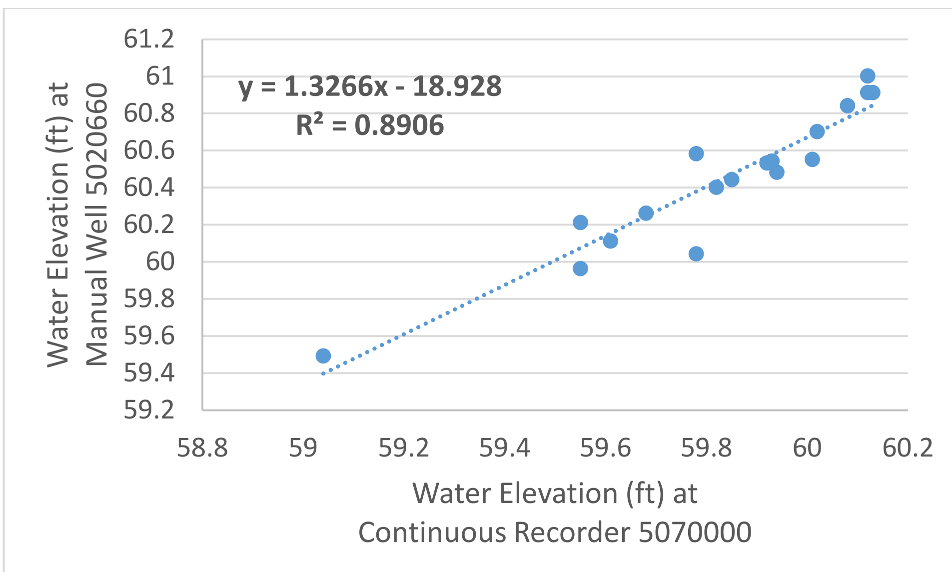


Figure B- 24. Water elevation regression of manual well 5020660 vs. continuous recorder 5070000.

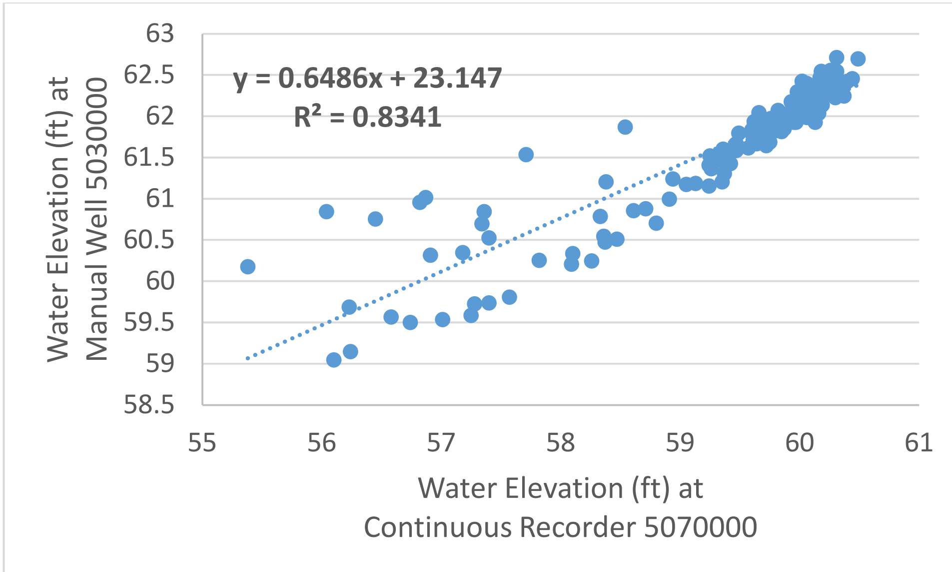


Figure B- 25. Water elevation regression of manual well 5030000 vs. continuous recorder 5070000.

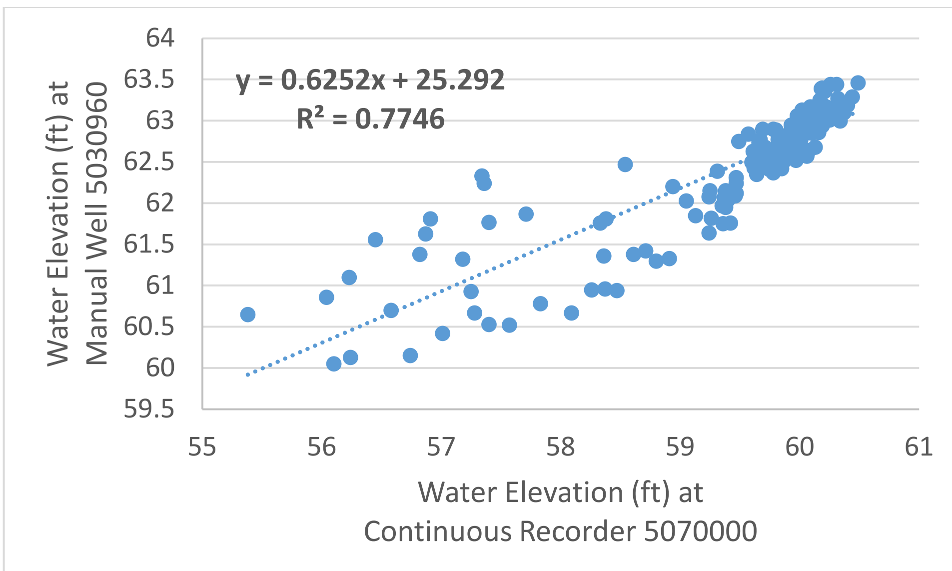


Figure B- 26. Water elevation regression of manual well 5030960 vs. continuous recorder 5070000.

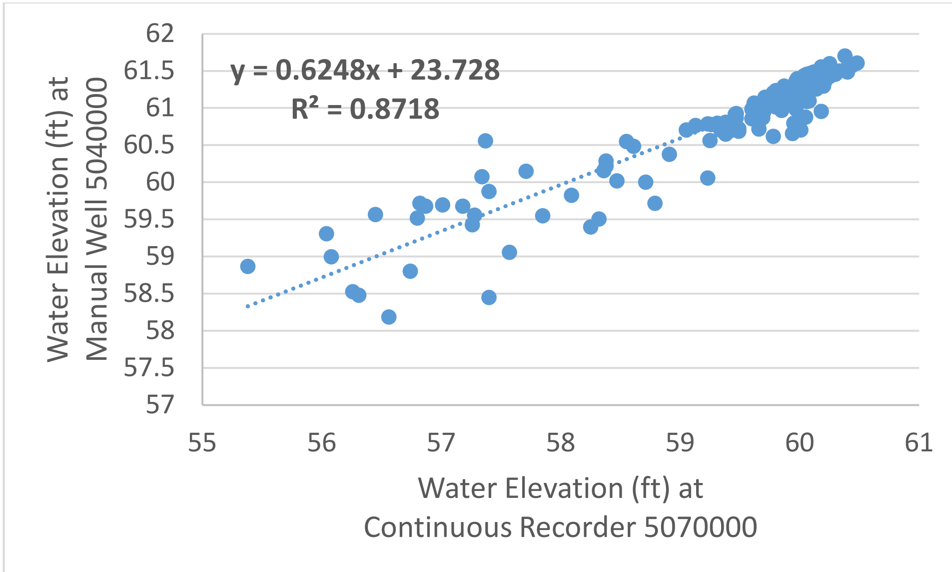


Figure B- 27. Water elevation regression of manual well 5040000 vs. continuous recorder 5070000.

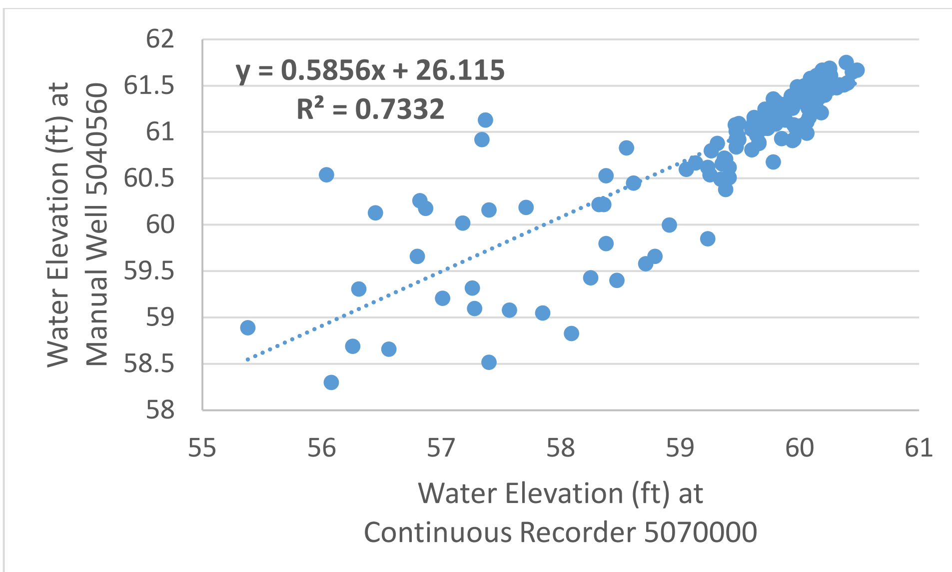


Figure B- 28. Water elevation regression of manual well 5040560 vs. continuous recorder 5070000.

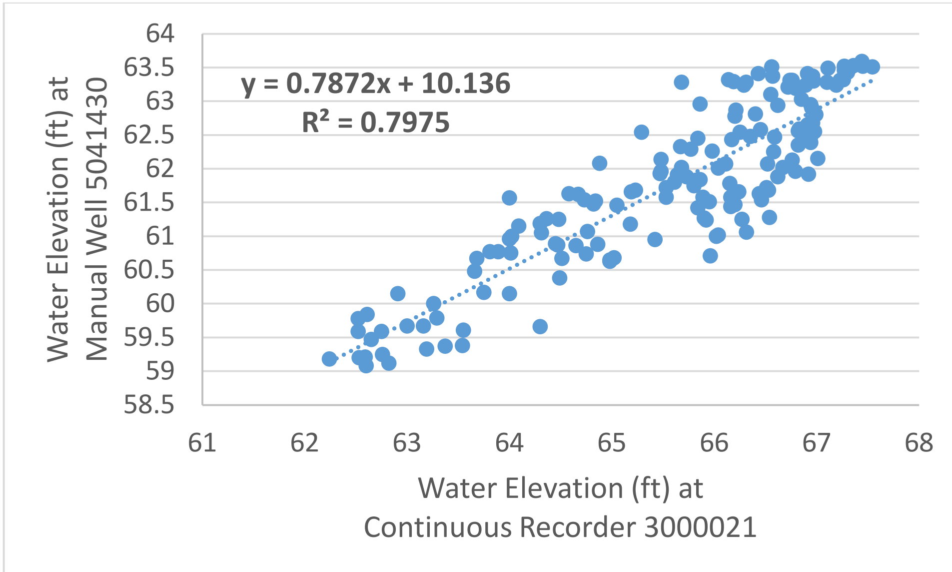


Figure B- 29. Water elevation regression of manual well 5041430 vs. continuous recorder 3000021.

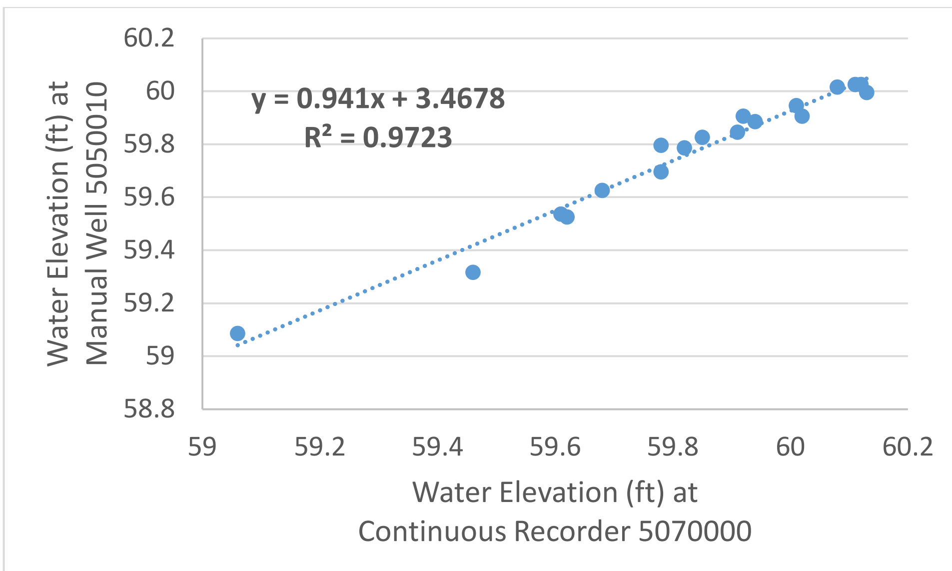


Figure B- 30. Water elevation regression of manual well 5050010 vs. continuous recorder 5070000.

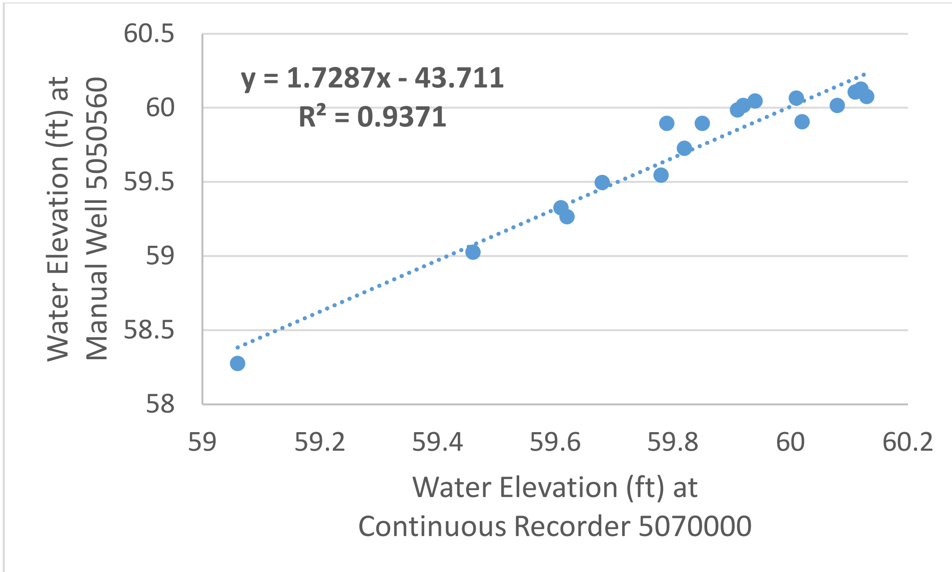


Figure B- 31. Water elevation regression of manual well 5050560 vs. continuous recorder 5070000.

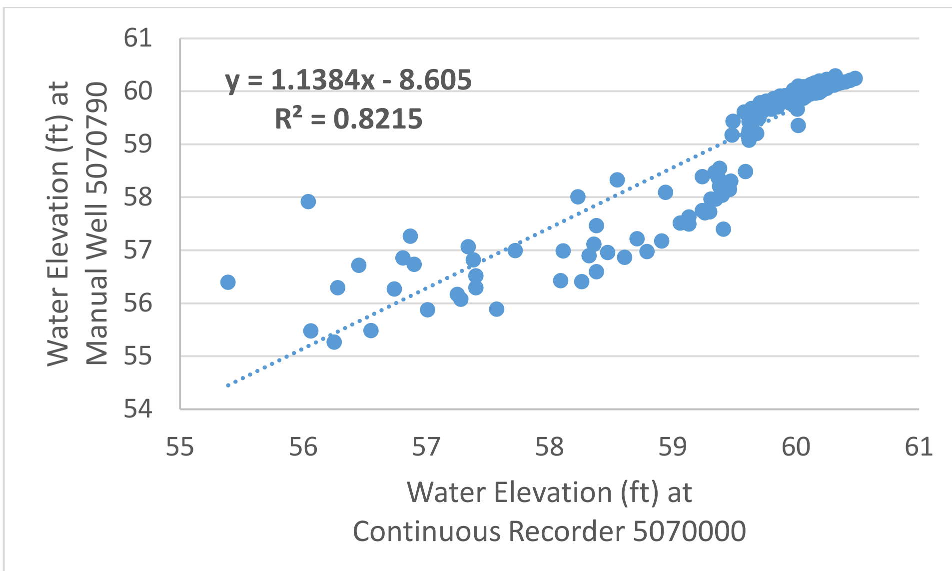


Figure B- 32. Water elevation regression of manual well 5070790 vs. continuous recorder 5070000.

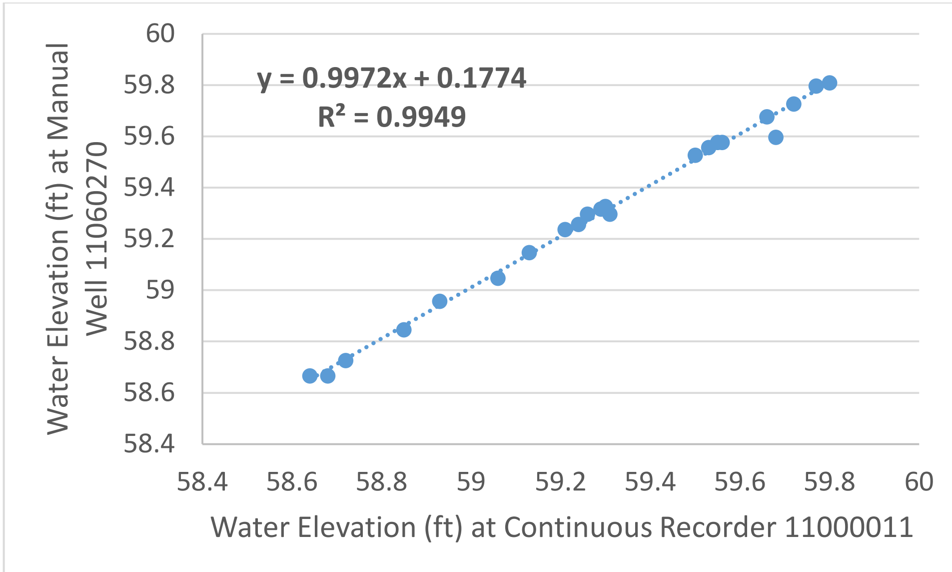


Figure B- 33. Water elevation regression of manual well 11060270 vs. continuous recorder 11000011.

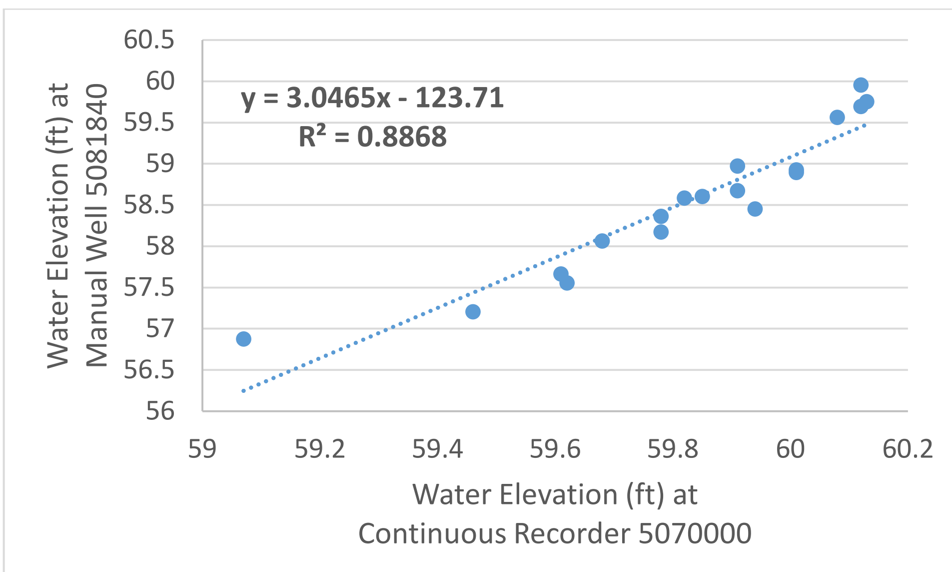


Figure B- 34. Water elevation regression of manual well 5081840 vs. continuous recorder 5070000.

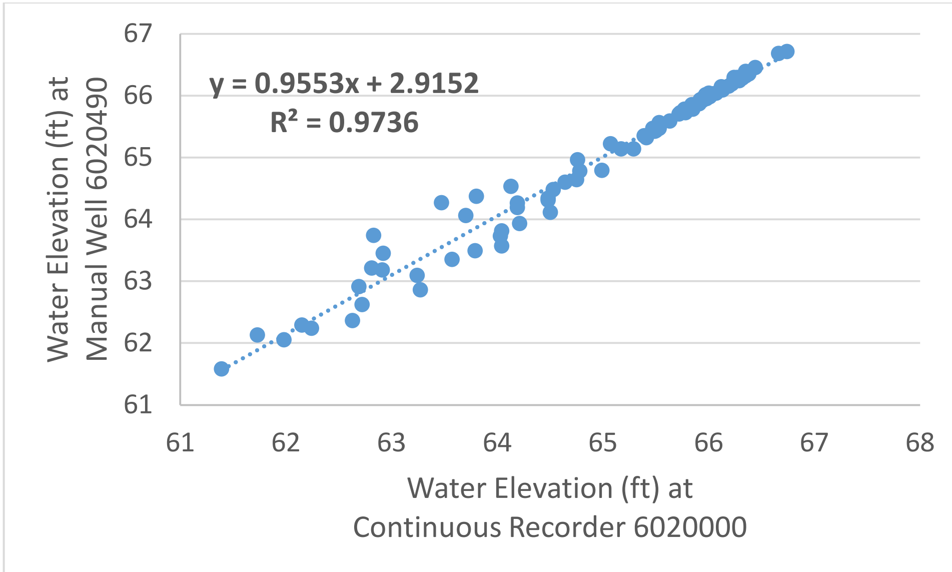


Figure B- 35. Water elevation regression of manual well 6020490 vs. continuous recorder 6020000.

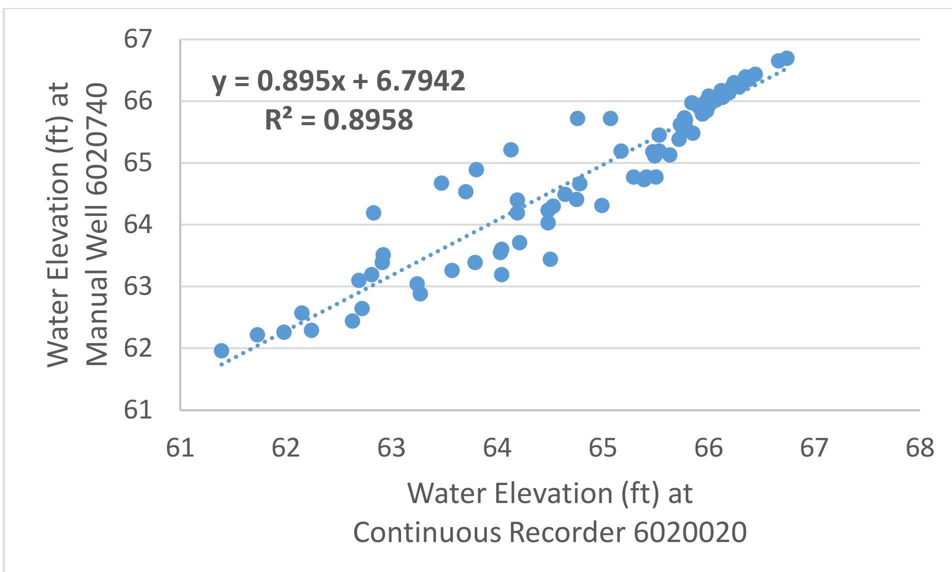


Figure B- 36. Water elevation regression of manual well 6020740 vs. continuous recorder 6020020.

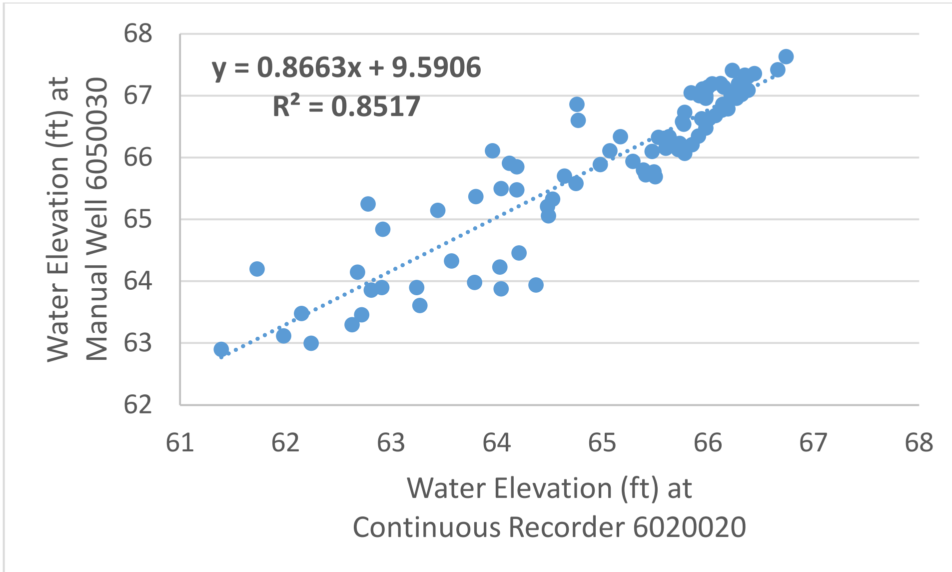


Figure B- 37. Water elevation regression of manual well 6050030 vs. continuous recorder 6020020.

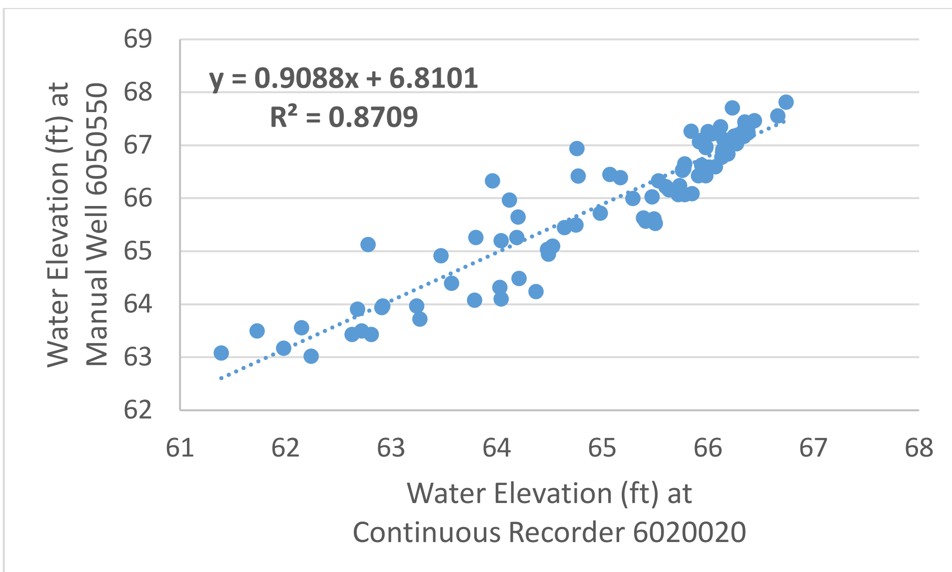


Figure B- 38. Water elevation regression of manual well 6050550 vs. continuous recorder 6020020.

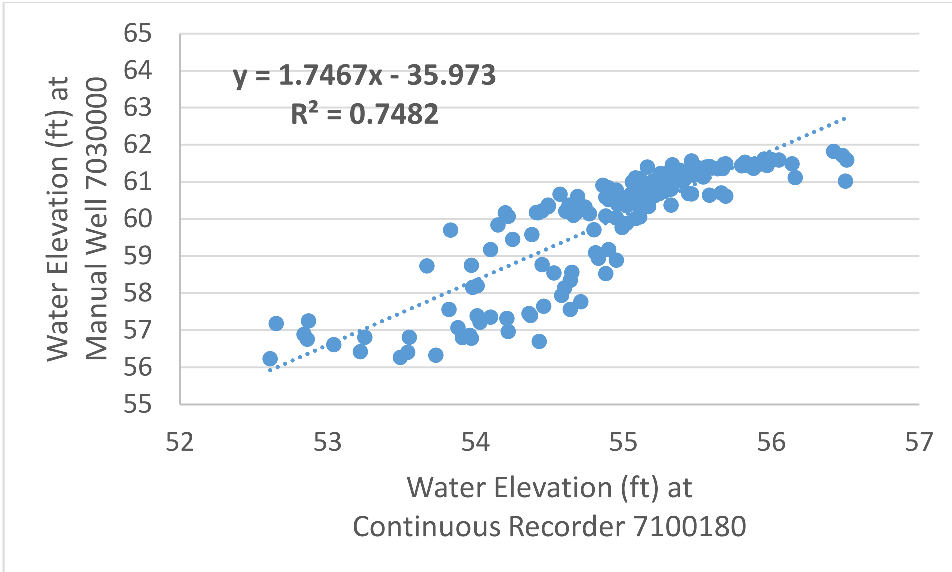


Figure B- 39. Water elevation regression of manual well 7030000 vs. continuous recorder 7100180.

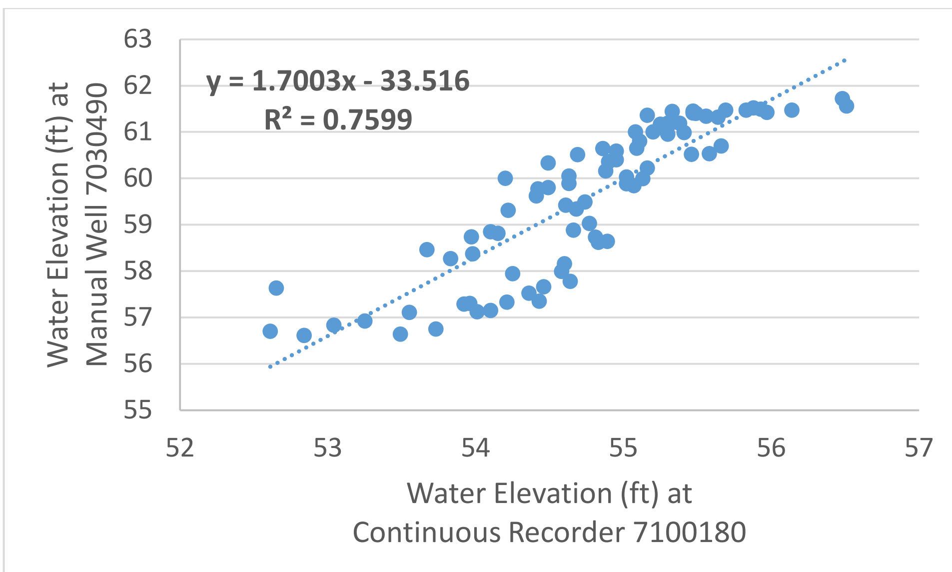


Figure B- 40. Water elevation regression of manual well 7030490 vs. continuous recorder 7100180.

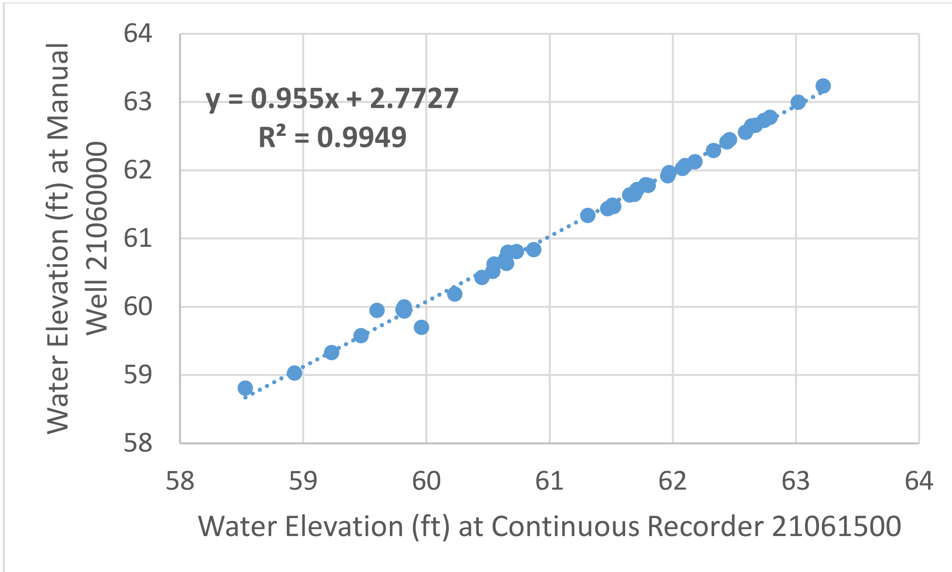


Figure B- 41. Water elevation regression of manual well 21060000 vs. continuous recorder 21061500.

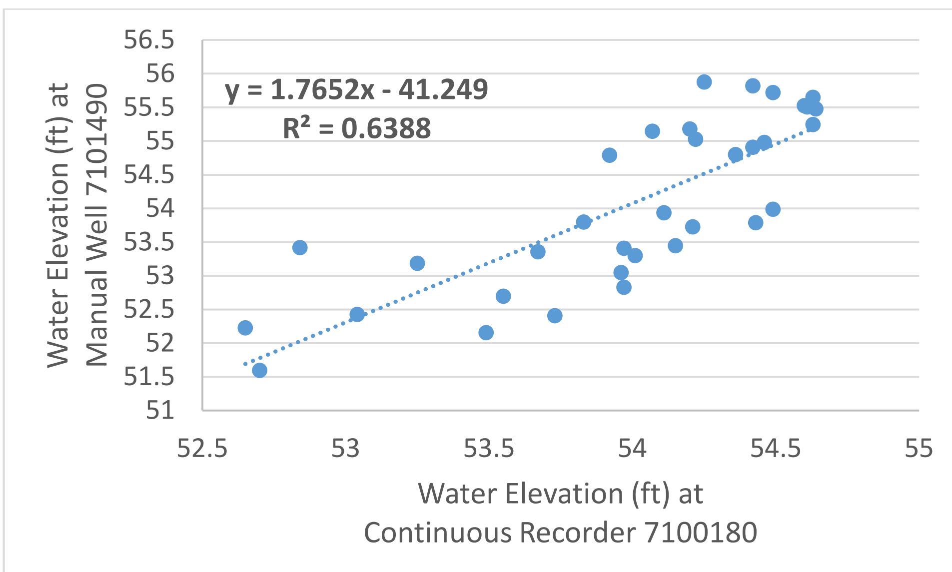


Figure B- 42. Water elevation regression of manual well 7101490 vs. continuous recorder 7100180.

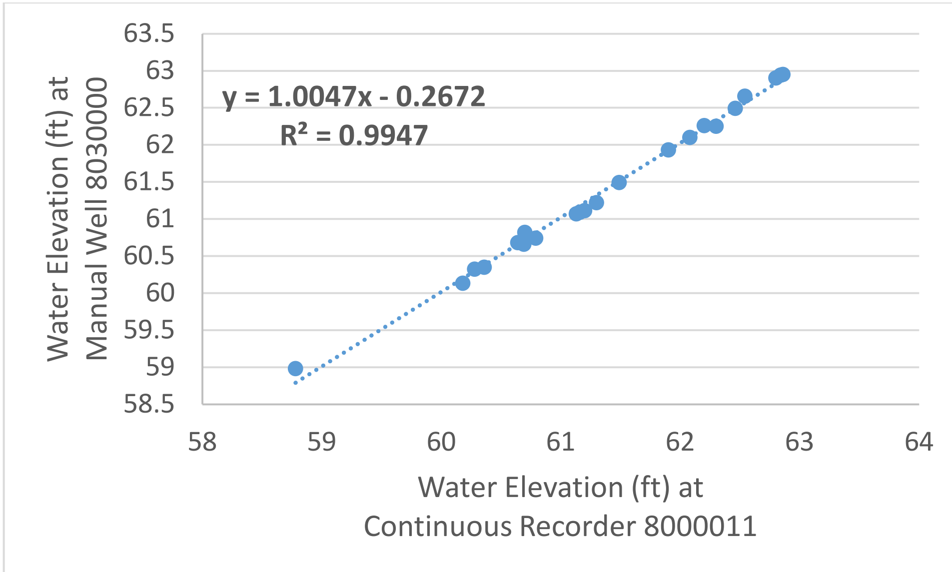


Figure B- 43. Water elevation regression of manual well 8030000 vs. continuous recorder 8000011.

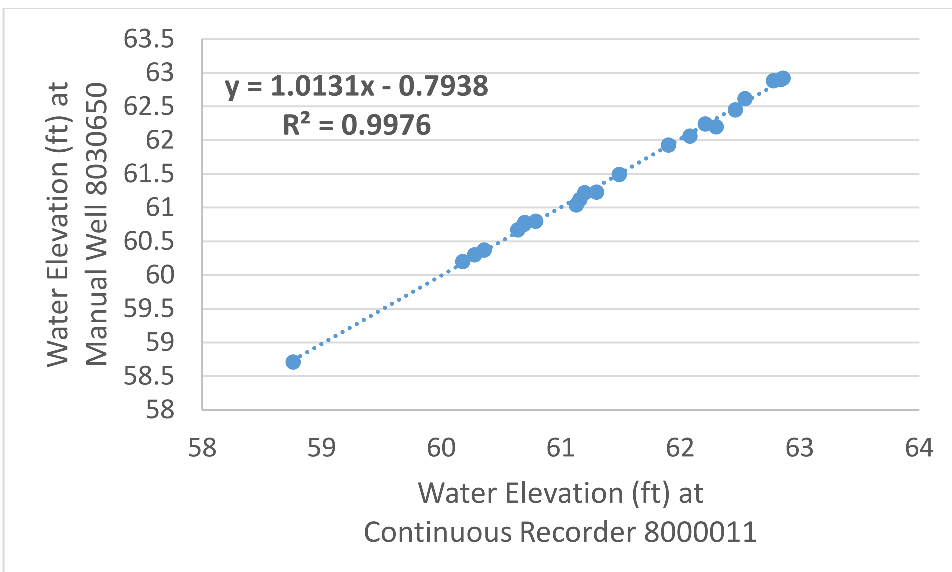


Figure B- 44. Water elevation regression of manual well 8030650 vs. continuous recorder 8000011.

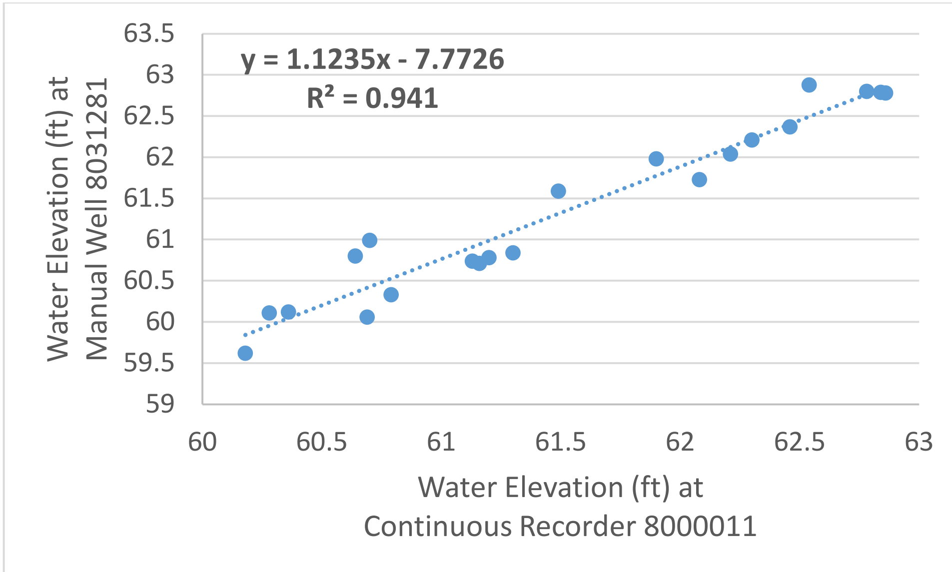


Figure B- 45. Water elevation regression of manual well 8031281 vs. continuous recorder 8000011.

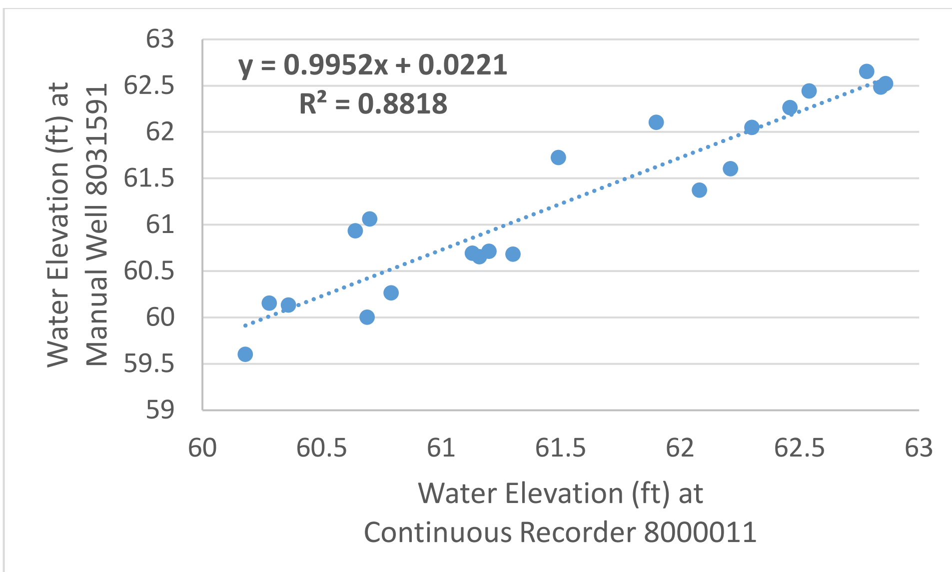


Figure B- 46. Water elevation regression of manual well 8031591 vs. continuous recorder 8000011.

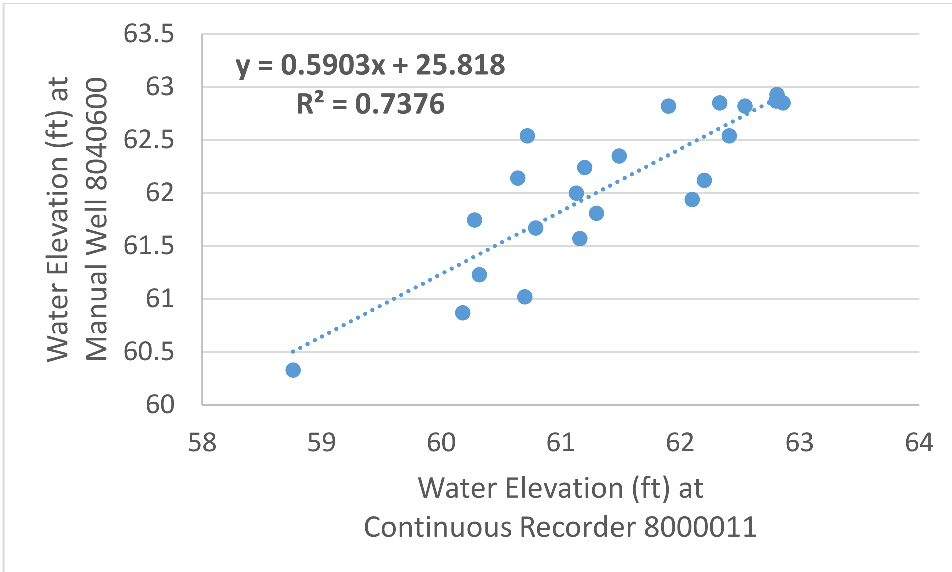


Figure B- 47. Water elevation regression of manual well 8040600 vs. continuous recorder 8000011.

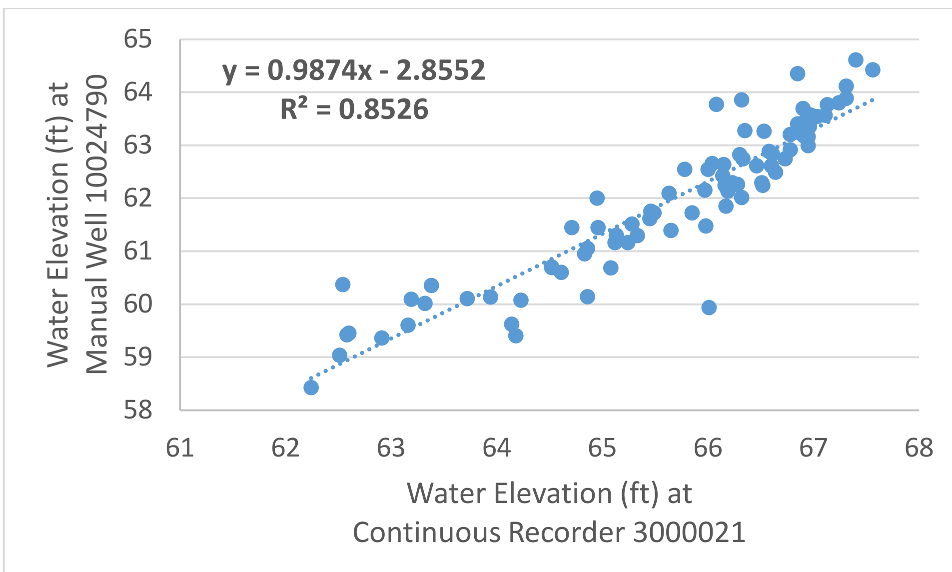


Figure B- 48. Water elevation regression of manual well 10024790 vs. continuous recorder 3000021.

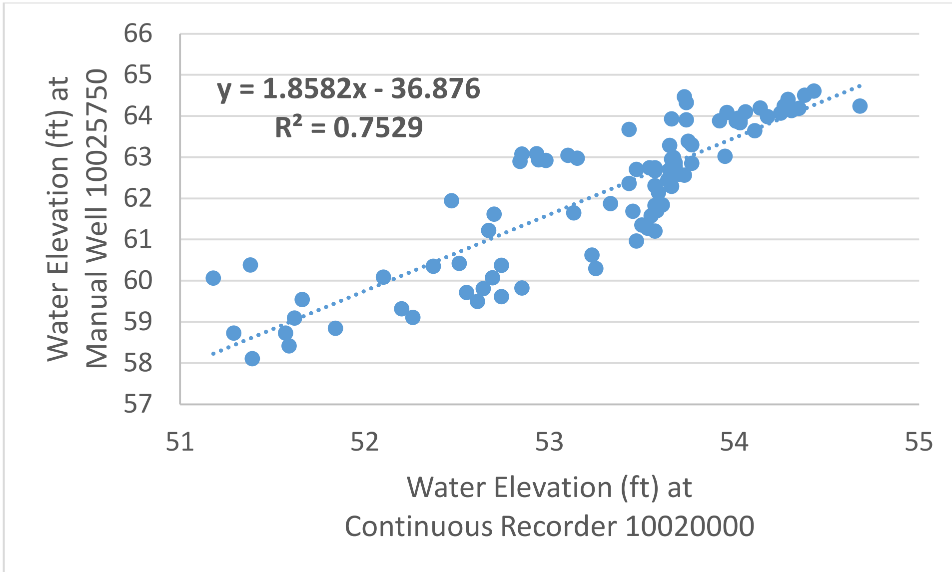


Figure B- 49. Water elevation regression of manual well 10025750 vs. continuous recorder 10020000.

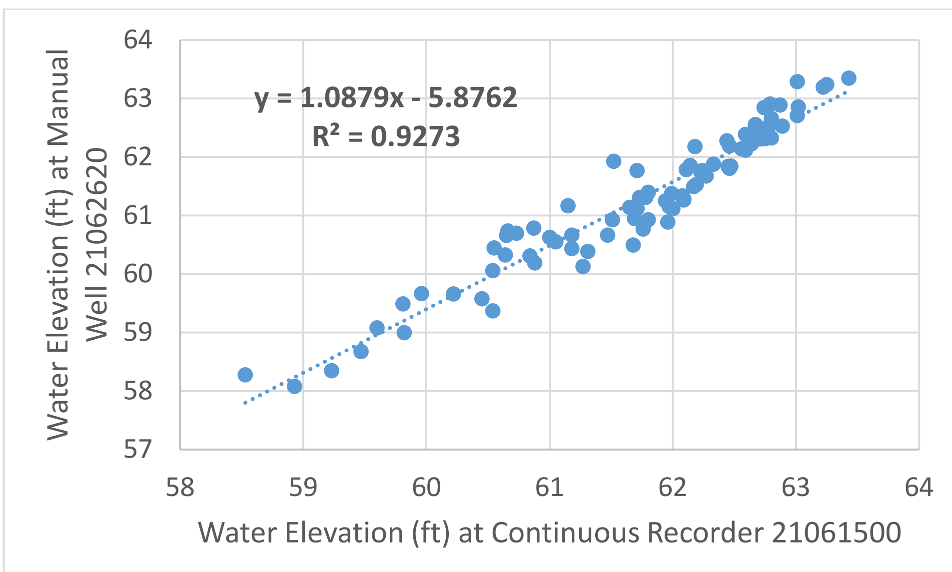


Figure B- 50. Water elevation regression of manual well 21062620 vs. continuous recorder 21061500.

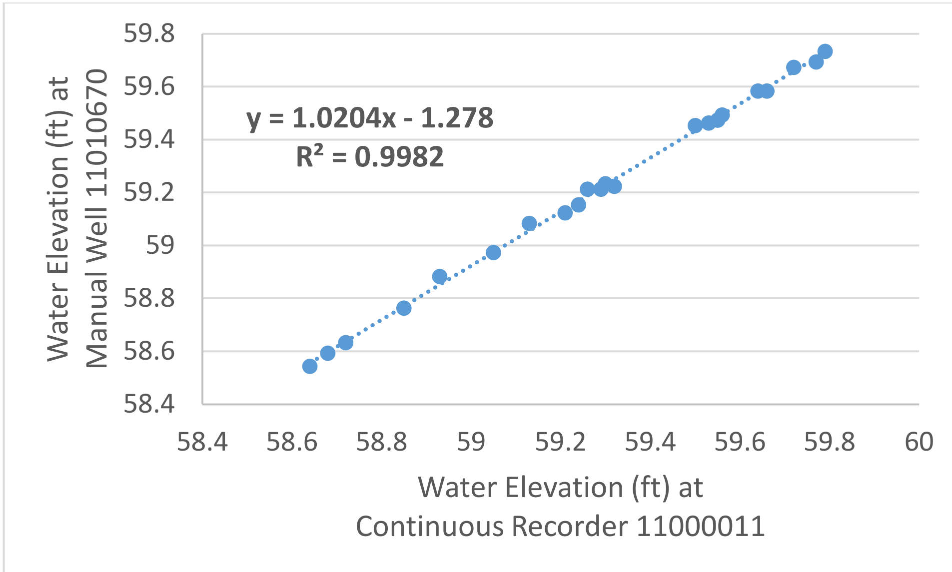


Figure B- 51. Water elevation regression of manual well 11010670 vs. continuous recorder 11000011.

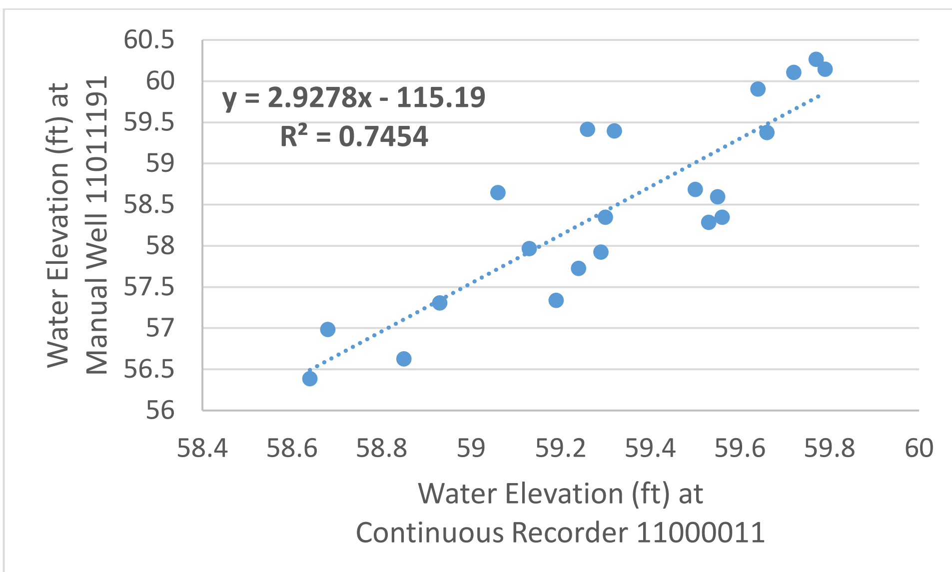


Figure B- 52. Water elevation regression of manual well 11011191 vs. continuous recorder 11000011.

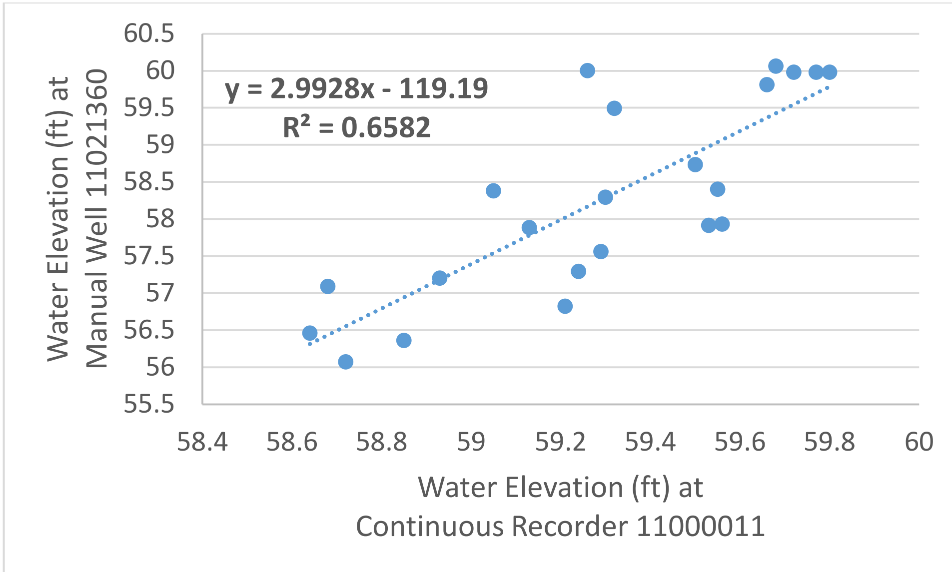


Figure B- 53. Water elevation regression of manual well 11021360 vs. continuous recorder 11000011.

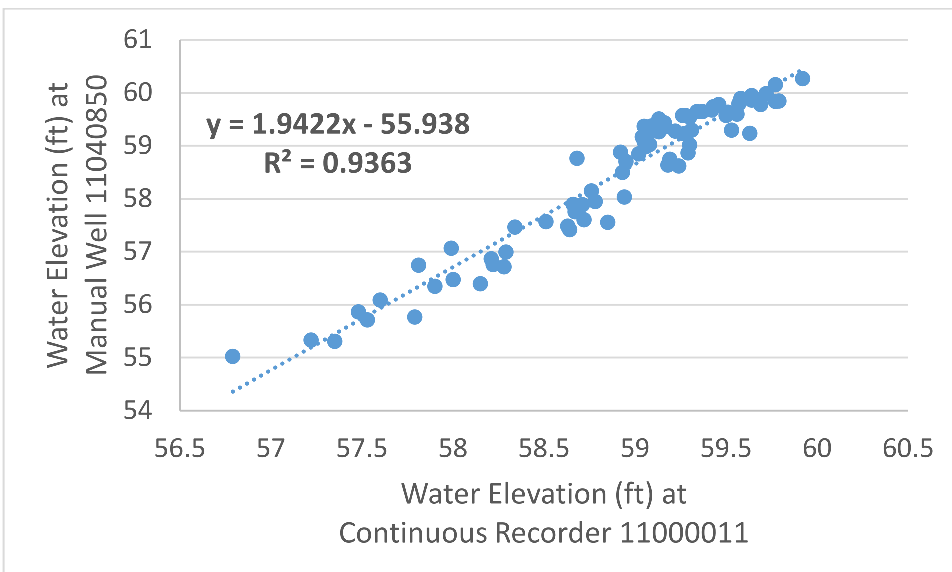


Figure B- 54. Water elevation regression of manual well 11040850 vs. continuous recorder 11000011.

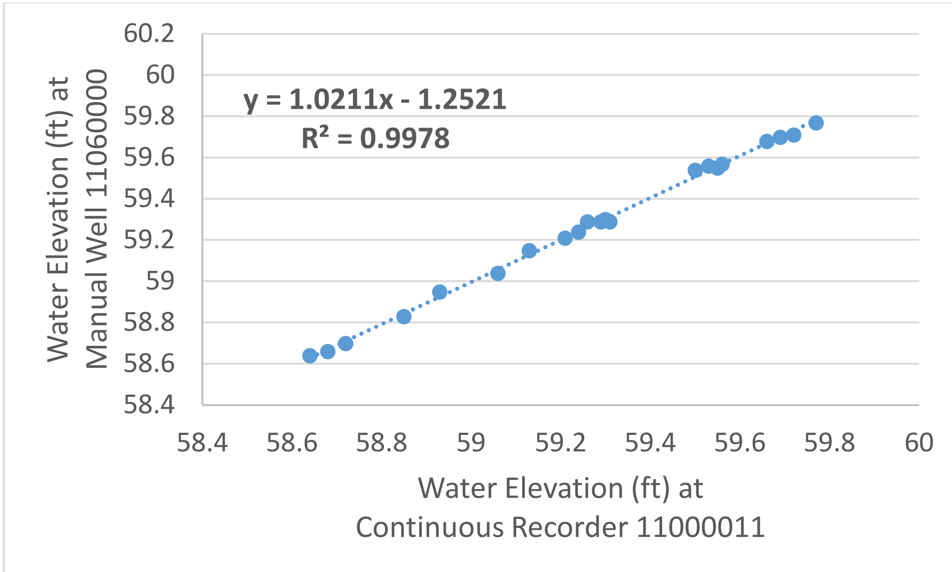


Figure B- 55. Water elevation regression of manual well 11060000 vs. continuous recorder 11000011.

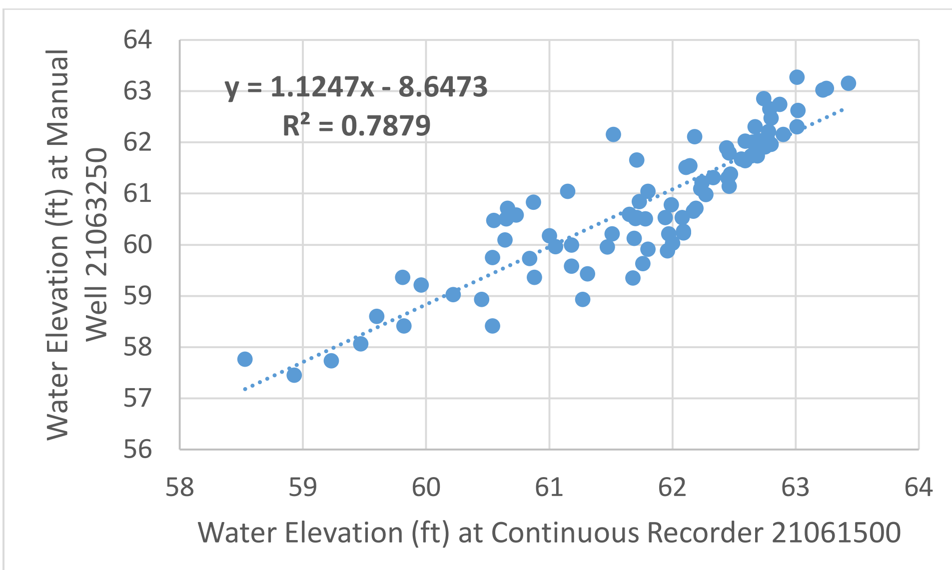


Figure B- 566. Water elevation regression of manual well 21063250 vs. continuous recorder 21061500.

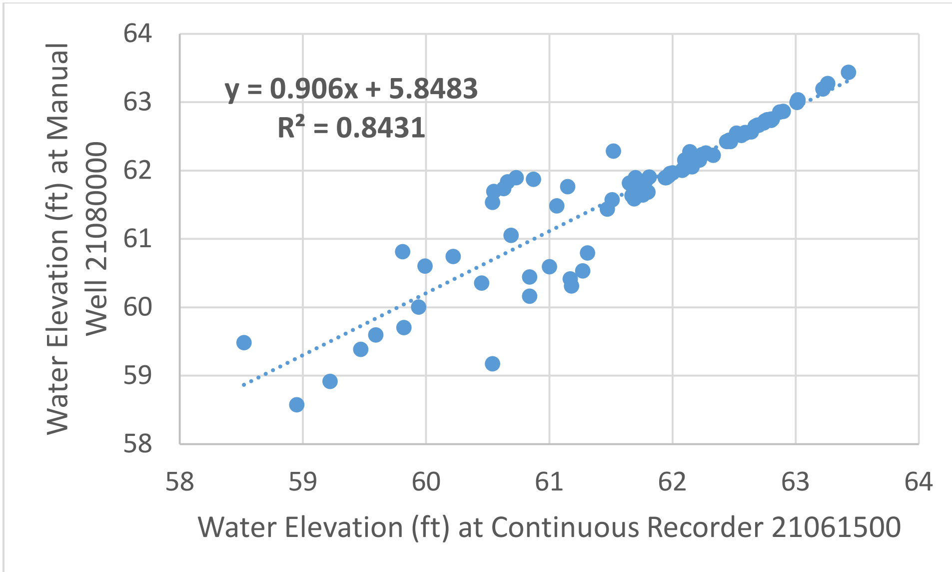


Figure B- 577. Water elevation regression of manual well 21080000 vs. continuous recorder 21061500.

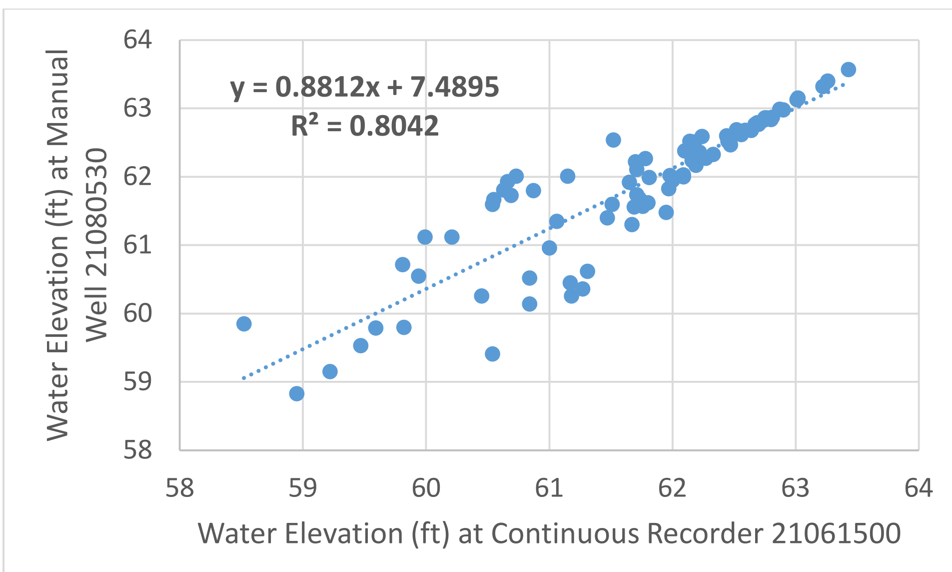


Figure B- 588. Water elevation regression of manual well 21080530 vs. continuous recorder 21061500.

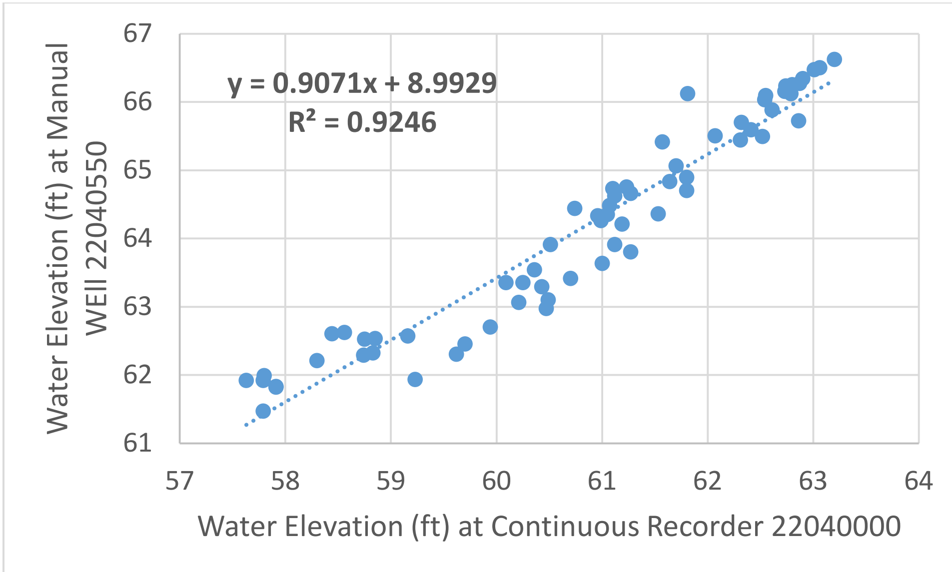


Figure B- 599. Water elevation regression of manual well 22040550 vs. continuous recorder 22040000.

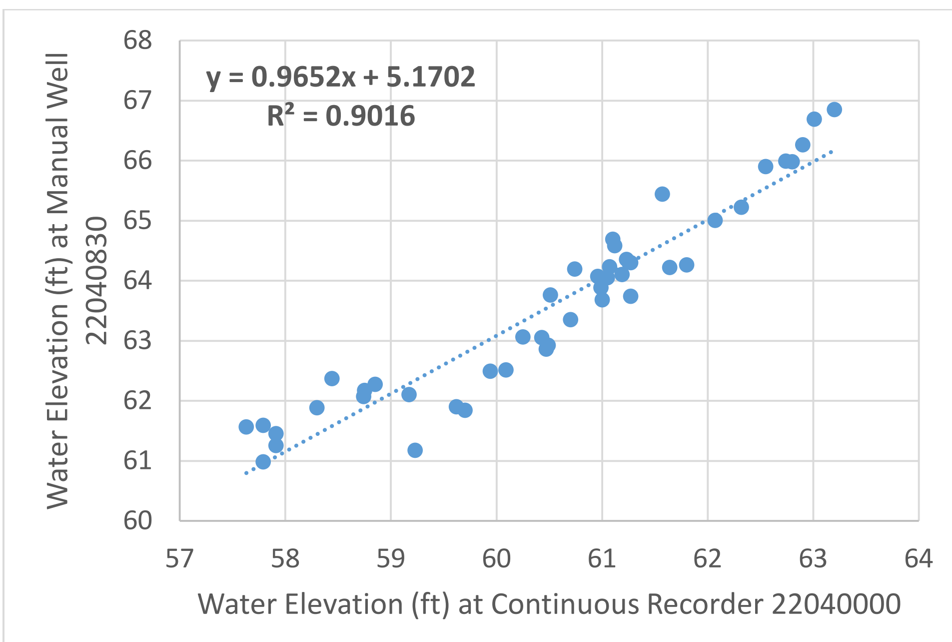


Figure B- 60. Water elevation regression of manual well 22040830 vs. continuous recorder 22040000.

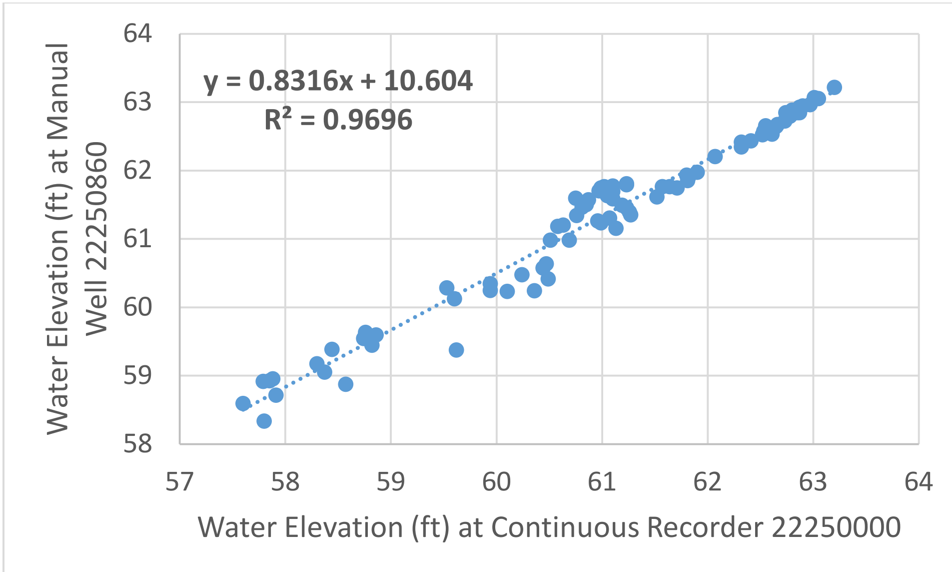


Figure B- 61. Water elevation regression of manual well 22250860 vs. continuous recorder 22250000.

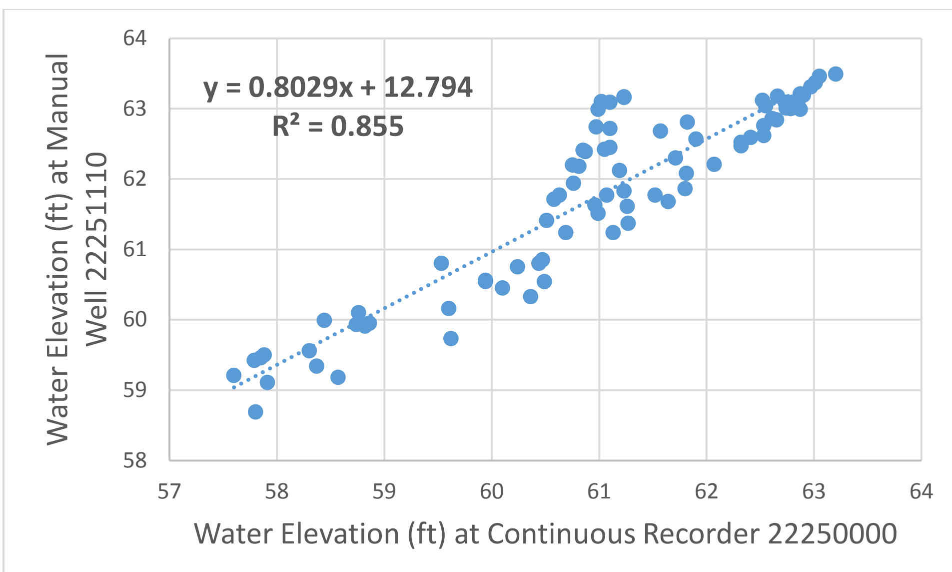


Figure B- 62. Water elevation regression of manual well 22251110 vs. continuous recorder 22250000.

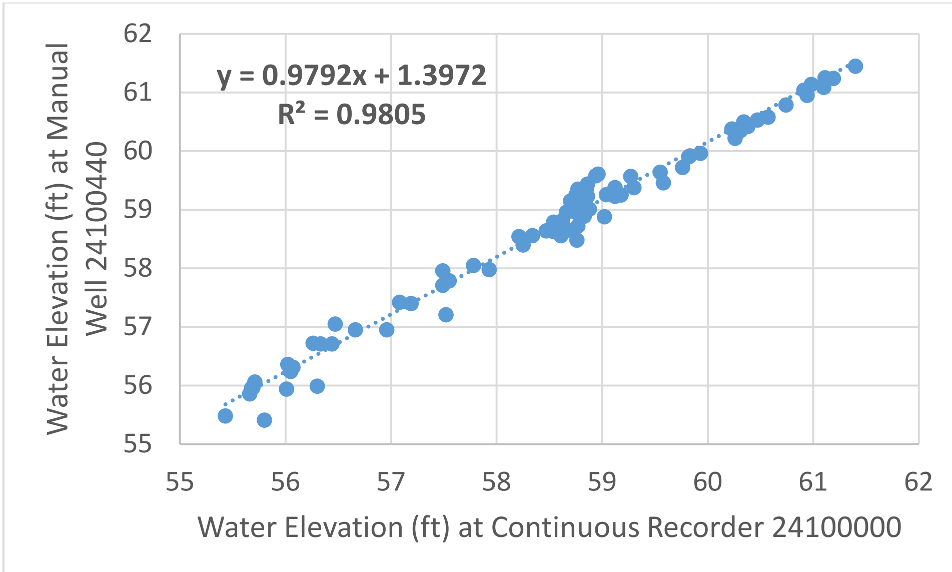


Figure B- 63. Water elevation regression of manual well 24100440 vs. continuous recorder 24100000.

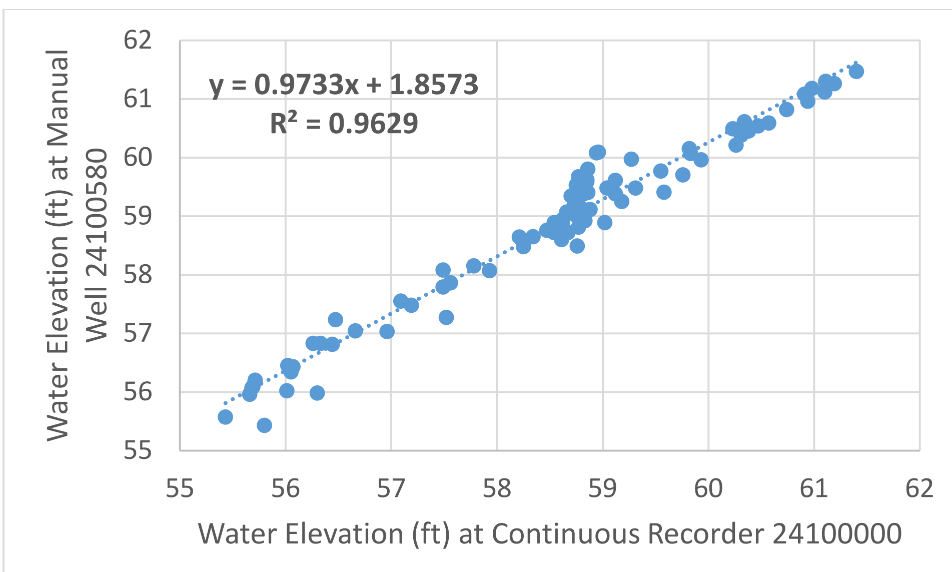


Figure B- 64. Water elevation regression of manual well 24100580 vs. continuous recorder 24100000.

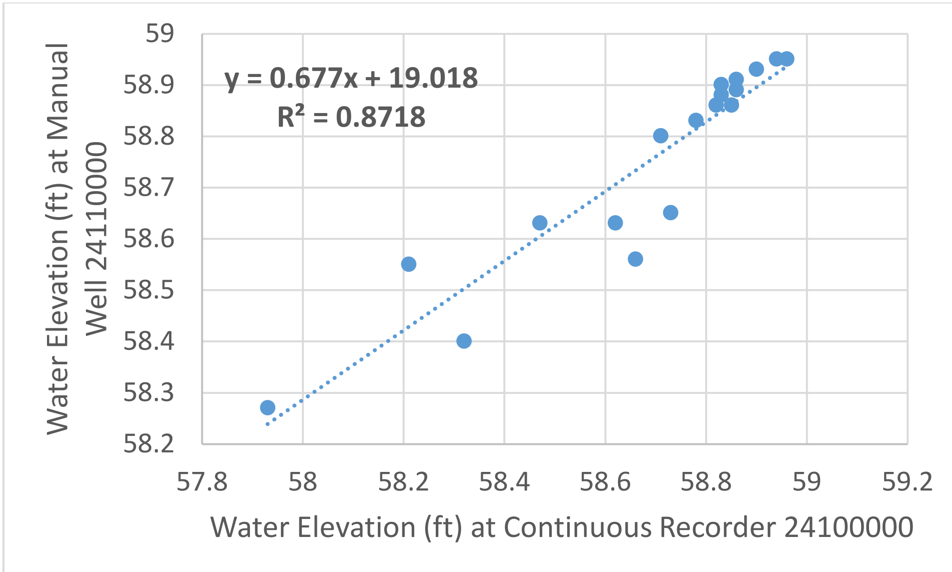


Figure B- 65. Water elevation regression of manual well 24110000 vs. continuous recorder 24100000.

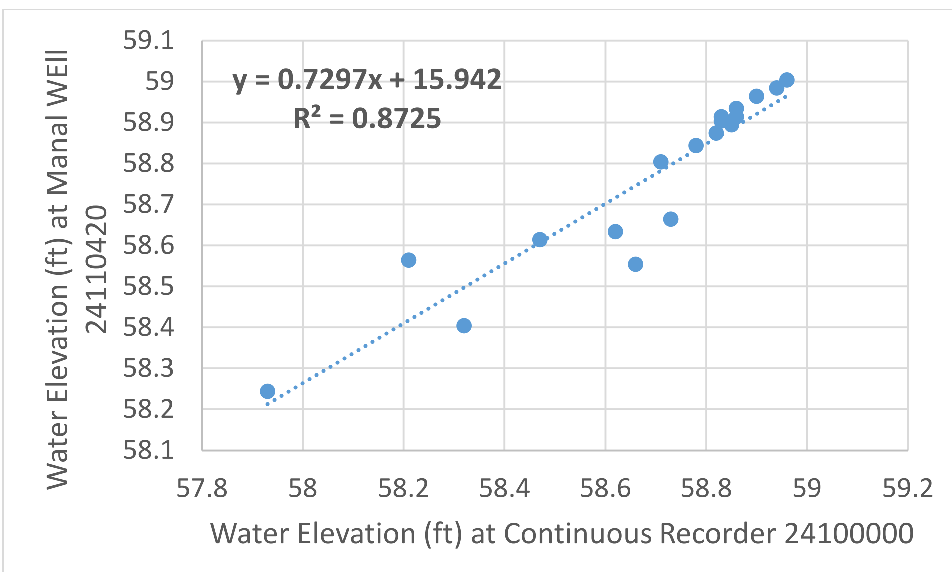


Figure B- 66. Water elevation regression of manual well 24110420 vs. continuous recorder 24100000.

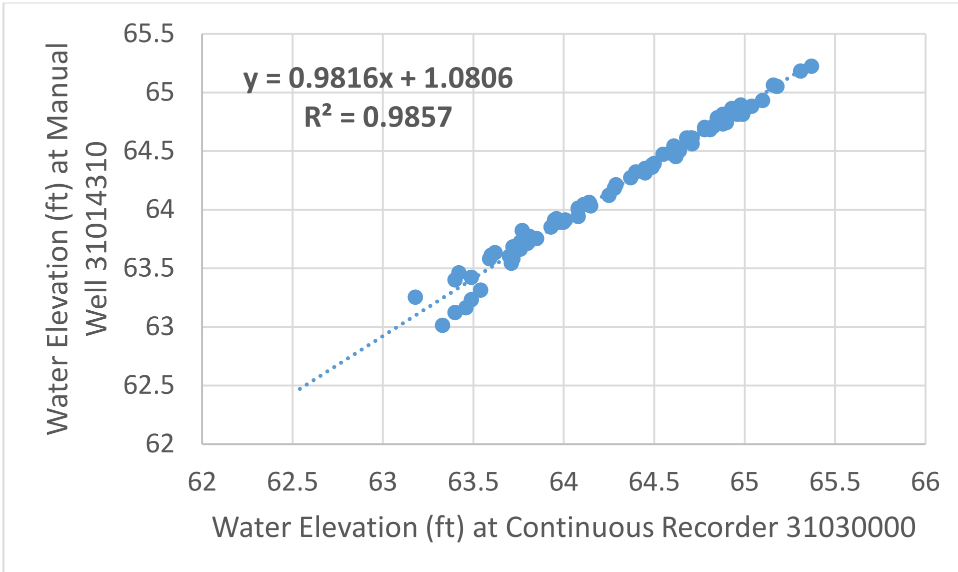


Figure B- 67. Water elevation regression of manual well 31014310 vs. continuous recorder 31030000.

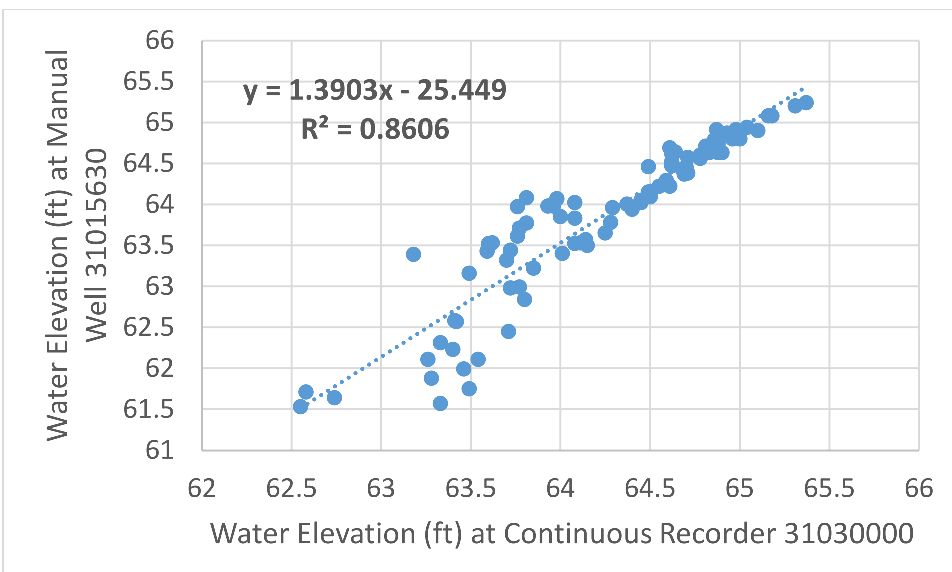


Figure B- 68. Water elevation regression of manual well 31015630 vs. continuous recorder 31030000.

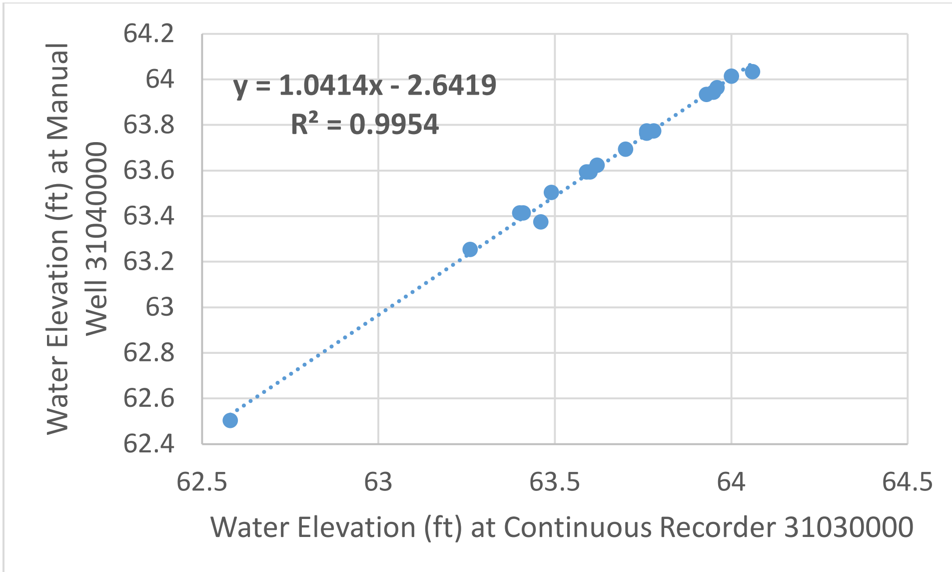


Figure B- 69. Water elevation regression of manual well 31040000 vs. continuous recorder 31030000.

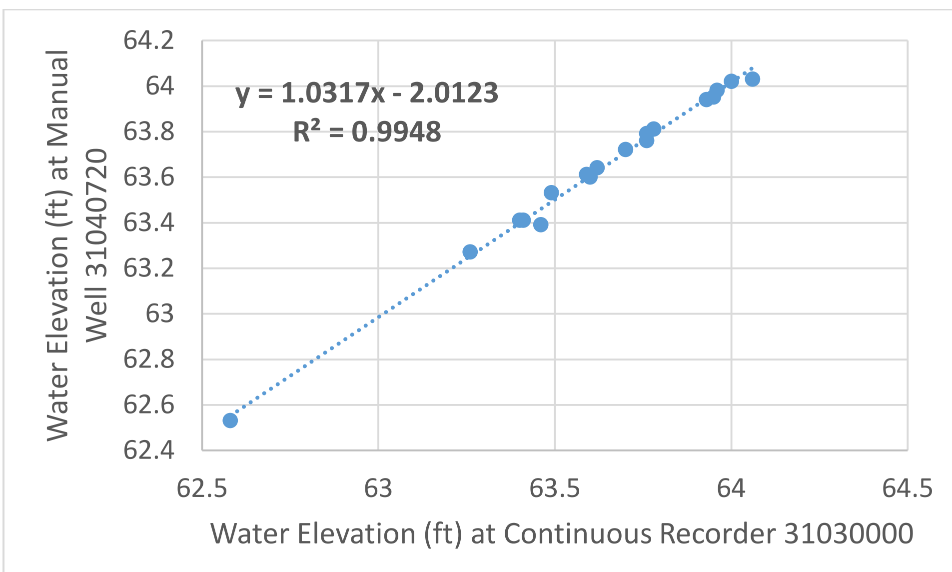


Figure B- 70. Water elevation regression of manual well 31040720 vs. continuous recorder 31030000.

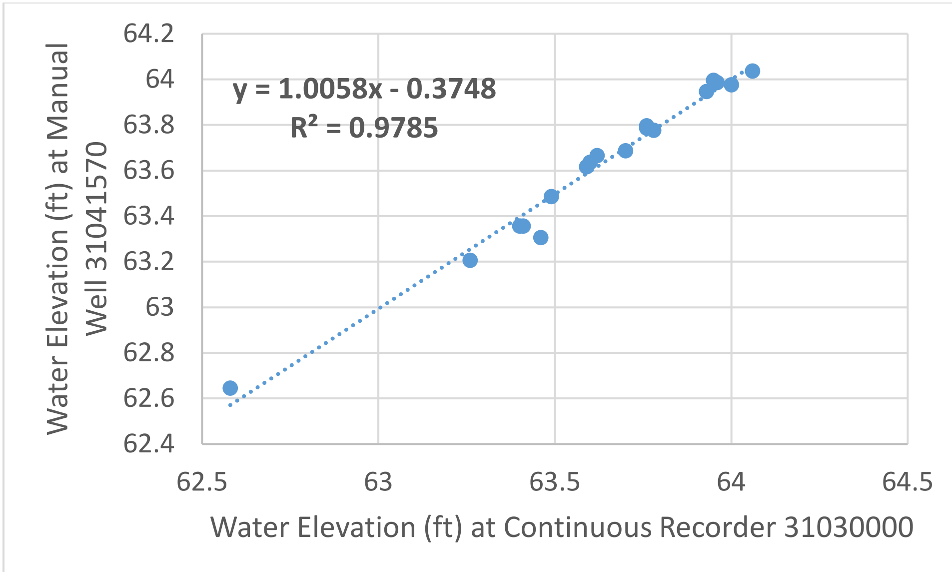


Figure B- 71. Water elevation regression of manual well 31041570 vs. continuous recorder 31030000.

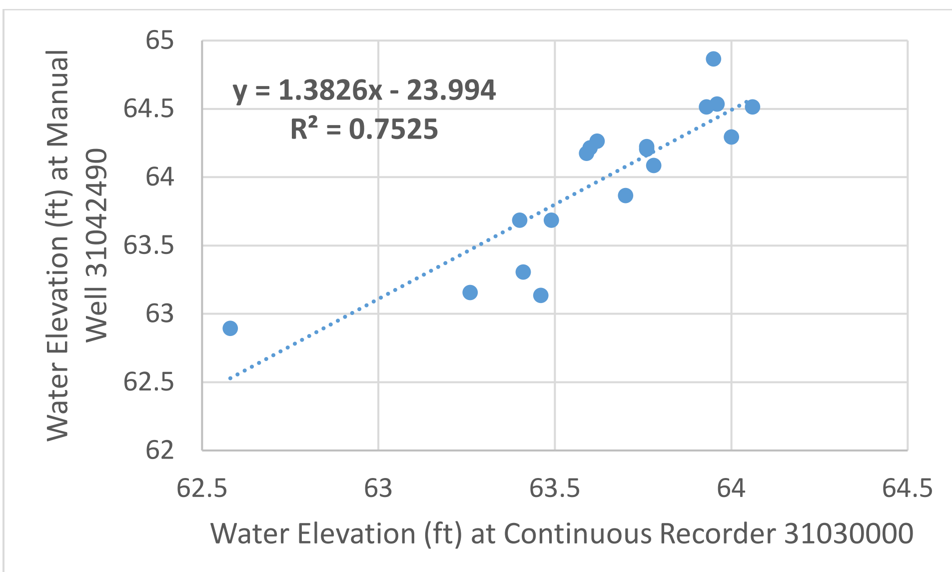


Figure B- 72. Water elevation regression of manual well 31042490 vs. continuous recorder 31030000.

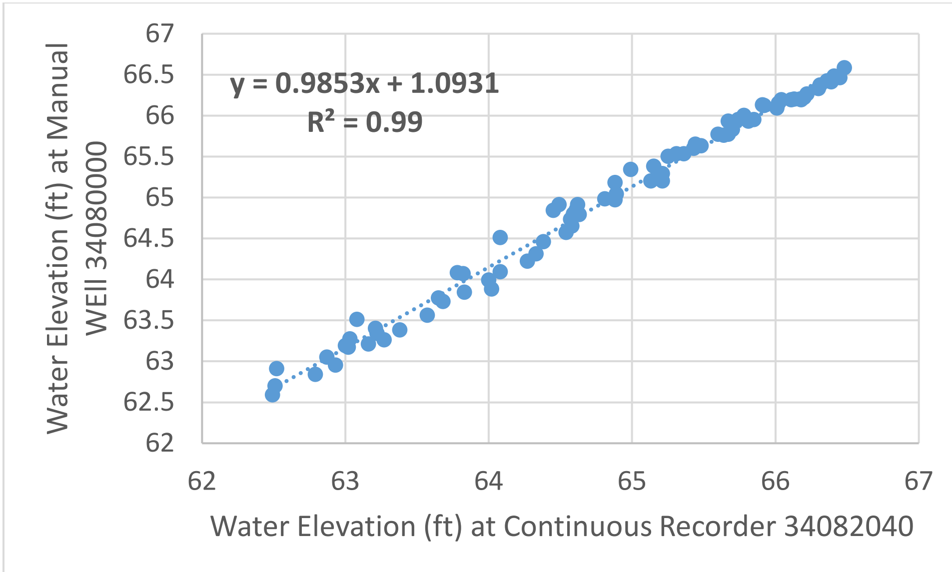


Figure B- 73. Water elevation regression of manual well 34080000 vs. continuous recorder 34082040.

APPENDIX C:
DISTRIBUTION PLOTS OF MODEL RANDOM EFFECTS

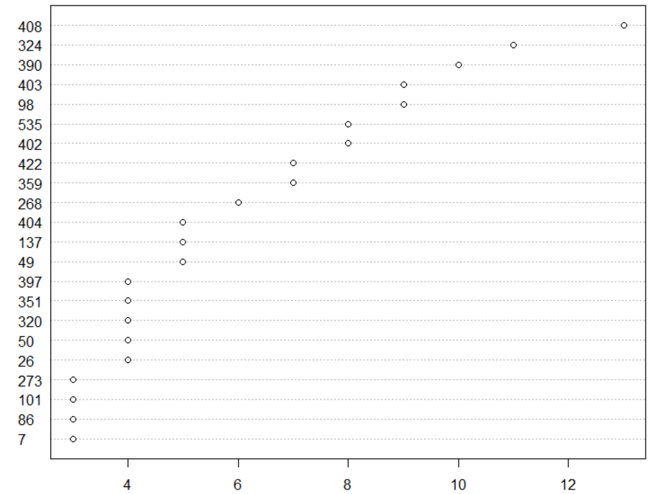


Figure C- 1. Distribution of soil sampling points across individual wetlands.

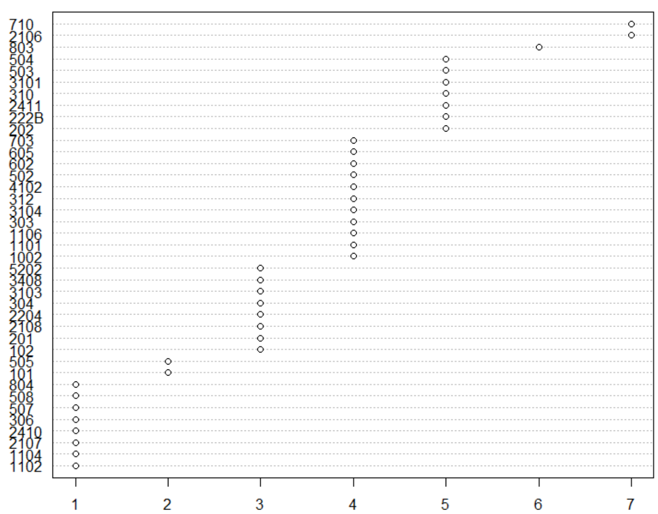


Figure C- 2. Distribution of soil sampling across individual transects.

APPENDIX D:
HYDROPERIOD ~ COMMUNITY MODEL OUPTUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = HP_mean2 ~ Pt_Com + (1 | fWL_ID2/HU_T), data = Data.hyd
ro.whole, family = "Gamma")

AIC: 1713.9

Coefficients:
              Estimate Std. Error z value Pr(>|z|)
(Intercept)      3.417      0.296   11.53  < 2e-16 ***
Pt_Combayhead    1.702      0.311    5.47  4.5e-08 ***
Pt_Comcypress    1.573      0.327    4.81  1.5e-06 ***
Pt_Commarsh      2.983      0.494    6.04  1.6e-09 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=137, fWL_ID2=22, fWL_ID2:HU_T=39
Random effect variance(s):
Group=fWL_ID2
            Variance StdDev
(Intercept)   0.2595 0.5094
Group=fWL_ID2:HU_T
            Variance StdDev
(Intercept)    1.193  1.092

Gamma shape parameter: 1.6295 (std. err.: 0.23272)

Log-likelihood: -849.947
```

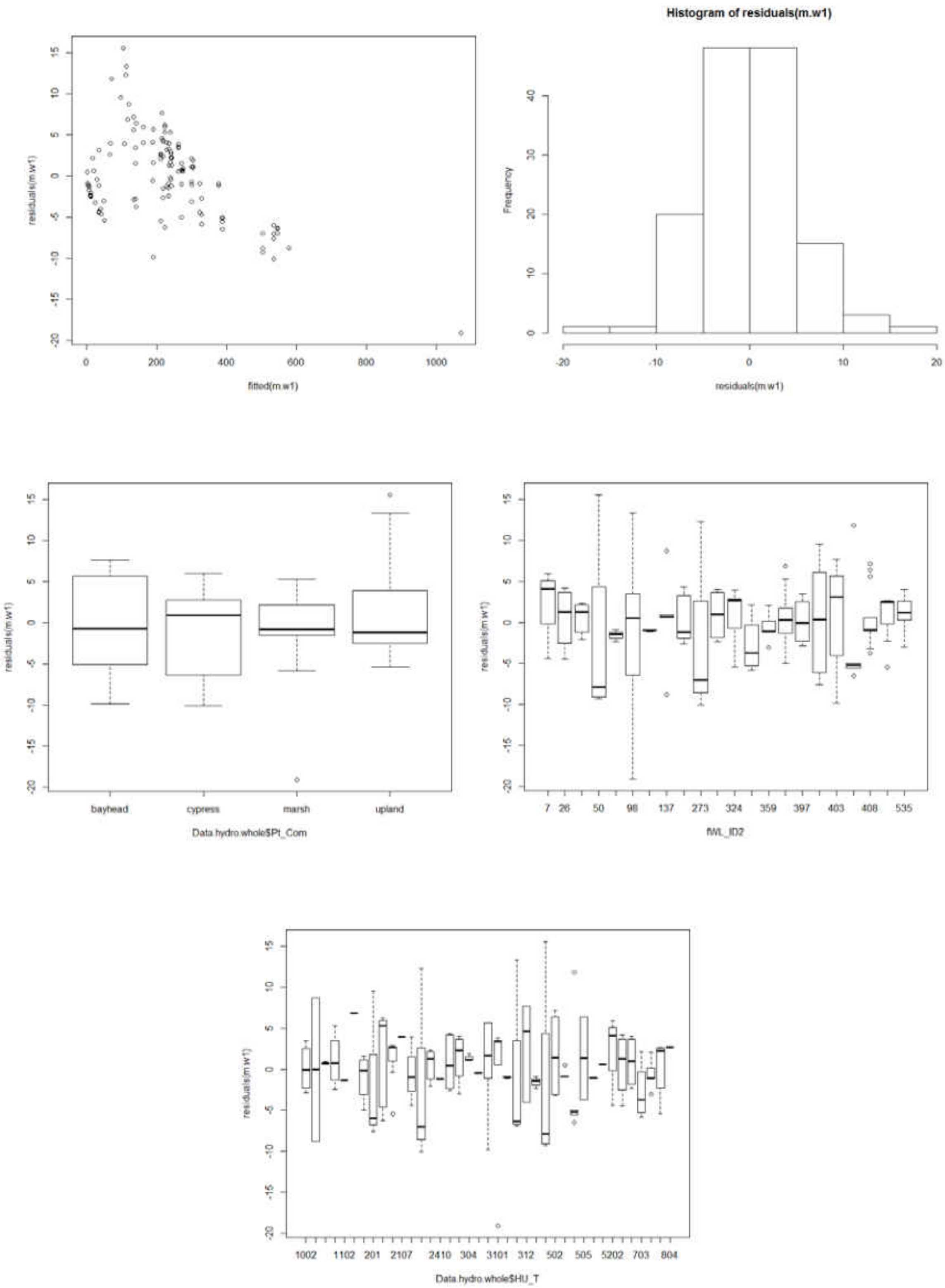


Figure D- 1. Residual plots of mean hydroperiod ~ vegetative community model.

APPENDIX E:
WATER DEPTH ~ COMMUNITY MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = WD_avg2 ~ Pt_Com + (1 | fWL_ID2/HU_T), data = Data.hydr
o.whole, family = "Gamma")

AIC: 122.3

Coefficients:
              Estimate Std. Error z value Pr(>|z|)
(Intercept)     0.8960     0.0330   27.13  < 2e-16 ***
Pt_Commarsh     0.3164     0.0331    9.56  < 2e-16 ***
Pt_Combayhead   0.1761     0.0316    5.57  2.6e-08 ***
Pt_Comcypress   0.2020     0.0331    6.11  1.0e-09 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=137, fWL_ID2=22, fWL_ID2:HU_T=39
Random effect variance(s):
Group=fWL_ID2
            Variance  StdDev
(Intercept)  0.00466 0.06826
Group=fWL_ID2:HU_T
            Variance StdDev
(Intercept)  0.02106 0.1451

Gamma shape parameter: 133.43 (std. err.: 21.099)

Log-likelihood: -54.1457
```

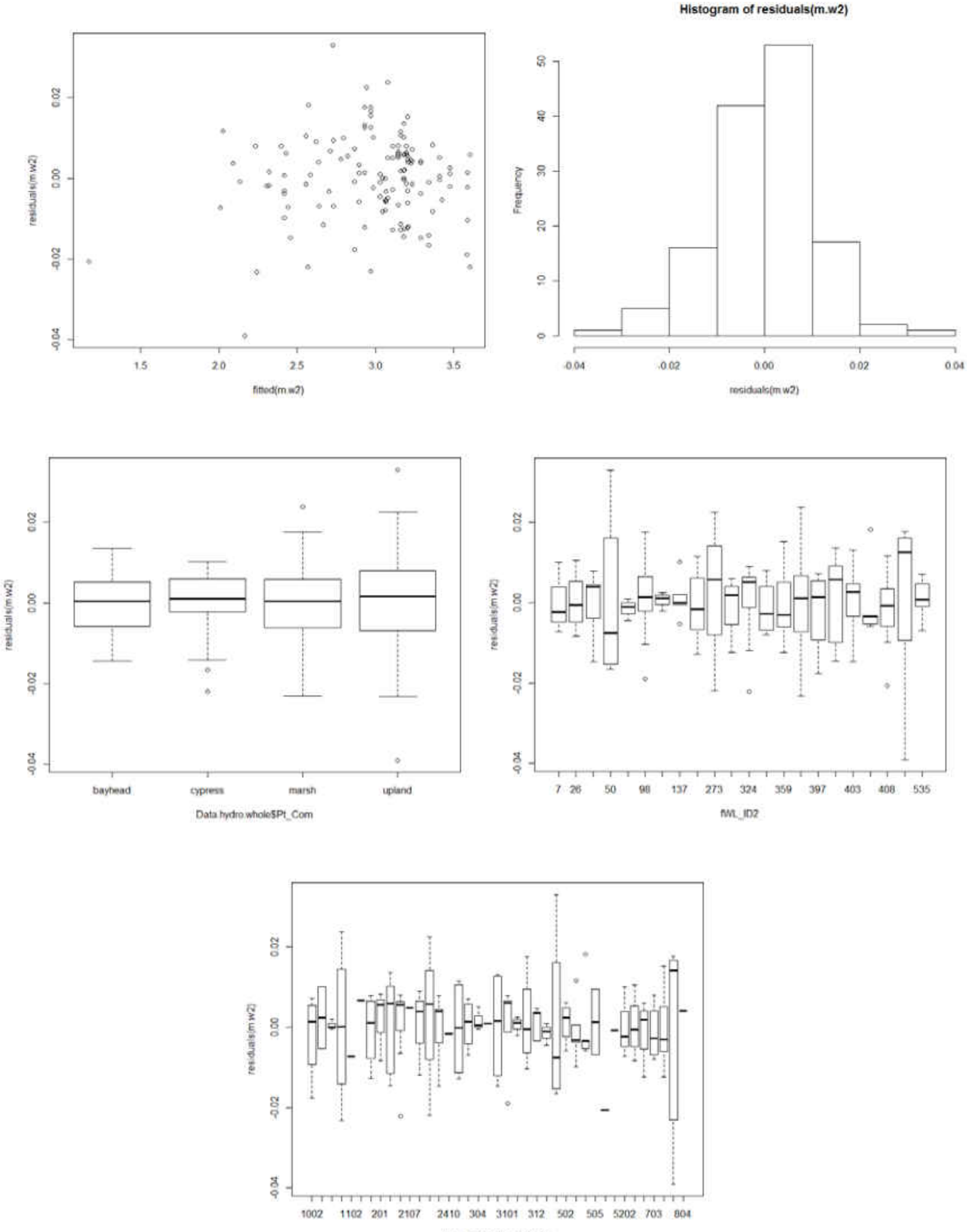


Figure E- 1. Residual plots for mean water depth ~ vegetative community model.

APPENDIX F:
SOIL CARBON CONCENTRATION ~ COMMUNITY MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = C_Conc3 ~ Pt_Com + (1 | fWL_ID3/HU_T), data = Data.carb
on.whole, family = "Gamma")

AIC: 1643.6

Coefficients:
               Estimate Std. Error z value Pr(>|z|)
(Intercept)       2.923      0.218   13.40  < 2e-16 ***
Pt_Combayhead     2.450      0.236   10.36  < 2e-16 ***
Pt_Comcypress     1.749      0.269    6.51  7.6e-11 ***
Pt_Commarsh       0.915      0.258    3.54    4e-04 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=149, fWL_ID3=22, fWL_ID3:HU_T=39
Random effect variance(s):
Group=fWL_ID3
            Variance StdDev
(Intercept)   0.3145 0.5608
Group=fWL_ID3:HU_T
            Variance StdDev
(Intercept)   0.3464 0.5885

Gamma shape parameter: 1.8124 (std. err.: 0.2192)

Log-likelihood: -814.822
```

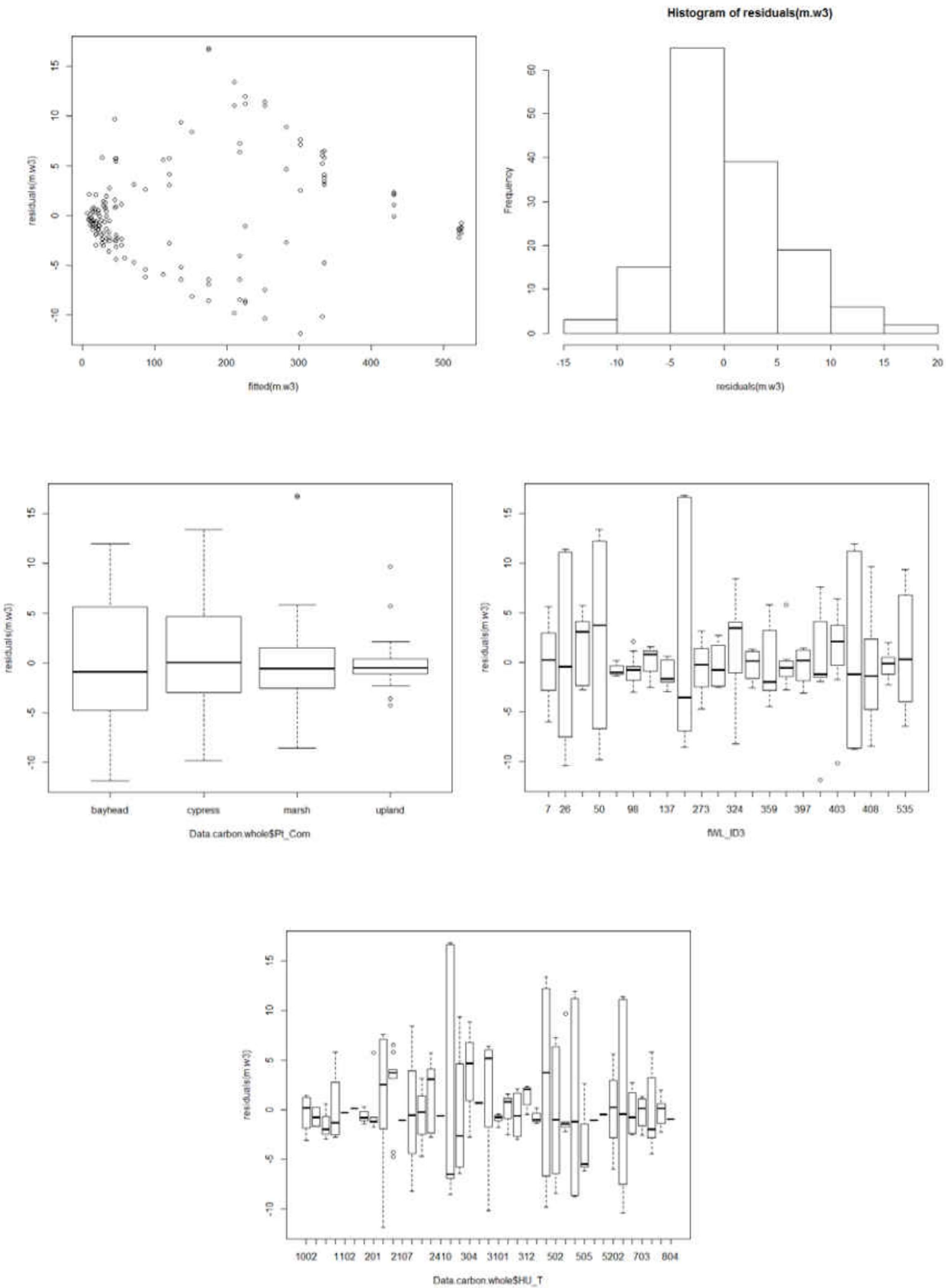
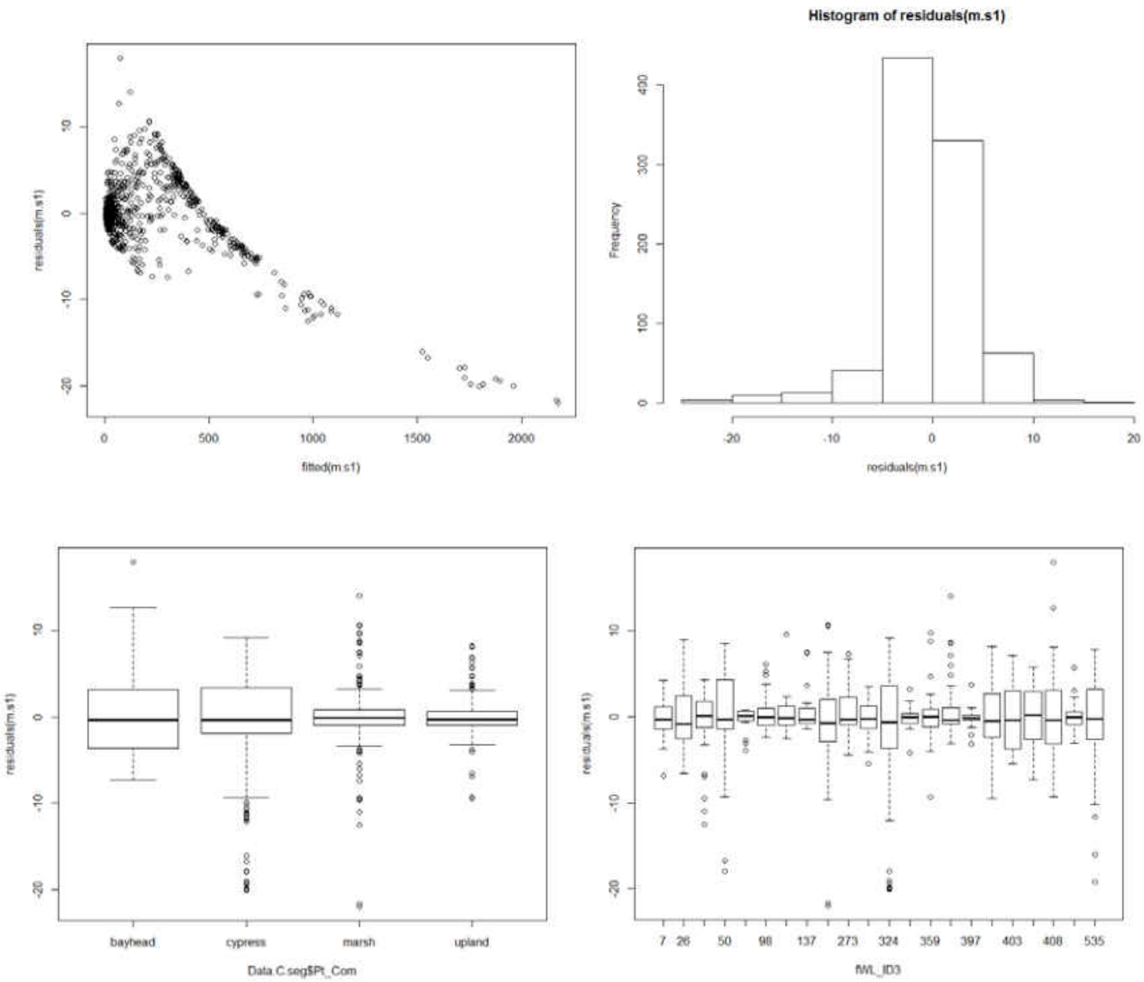


Figure F- 1. Residual plots for soil carbon concentration ~ vegetative community model.

APPENDIX G:
SOIL CARBON CONCENTRATION ~ COMMUNITY AND SOIL DEPTH MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = C_Conc ~ Pt_Com * Seg_Code + (1 | fWL_ID3/HU_T/fPt_ID3,
data = Data.C.seg, family = "Gamma")

AIC: 9579.9

Coefficients:
                        Estimate Std. Error z value Pr(>|z|)
(Intercept)              4.48633    0.24104   18.61  < 2e-16 ***
Pt_Combayhead            1.16614    0.29472    3.96  7.6e-05 ***
Pt_Comcypress            1.38311    0.31034    4.46  8.3e-06 ***
Pt_Commarsh              0.62721    0.30046    2.09  0.03684 *
Seg_CodeB               -0.86568    0.15166   -5.71  1.1e-08 ***
Seg_CodeC               -1.76555    0.15382  -11.48  < 2e-16 ***
Seg_CodeD               -2.35990    0.15544  -15.18  < 2e-16 ***
Seg_CodeE               -2.42863    0.15773  -15.40  < 2e-16 ***
Seg_CodeF               -2.31062    0.15895  -14.54  < 2e-16 ***
Pt_Combayhead:Seg_CodeB  0.76463    0.20465    3.74  0.00019 ***
Pt_Comcypress:Seg_CodeB  0.27574    0.21199    1.30  0.19336
Pt_Commarsh:Seg_CodeB    0.07531    0.19976    0.38  0.70617
Pt_Combayhead:Seg_CodeC  1.51453    0.20764    7.29  3.0e-13 ***
Pt_Comcypress:Seg_CodeC  0.53644    0.21570    2.49  0.01288 *
Pt_Commarsh:Seg_CodeC    0.41506    0.20301    2.04  0.04090 *
Pt_Combayhead:Seg_CodeD  1.85917    0.20965    8.87  < 2e-16 ***
Pt_Comcypress:Seg_CodeD  0.72061    0.21967    3.28  0.00104 **
Pt_Commarsh:Seg_CodeD    0.53592    0.20496    2.61  0.00893 **
Pt_Combayhead:Seg_CodeE  1.80715    0.21222    8.52  < 2e-16 ***
Pt_Comcypress:Seg_CodeE  0.57630    0.22281    2.59  0.00970 **
Pt_Commarsh:Seg_CodeE    0.33839    0.20697    1.63  0.10205
Pt_Combayhead:Seg_CodeF  1.59215    0.21380    7.45  9.6e-14 ***
Pt_Comcypress:Seg_CodeF  0.33930    0.22491    1.51  0.13141
Pt_Commarsh:Seg_CodeF   -0.00619    0.20876   -0.03  0.97636
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=898, fWL_ID3=22, fWL_ID3:HU_T=39, fWL_ID3:HU
_T:fPt_ID3=150
Random effect variance(s):
Group=fWL_ID3
            Variance StdDev
(Intercept)   0.2872 0.5359
Group=fWL_ID3:HU_T
            Variance StdDev
(Intercept)   0.3212 0.5668
Group=fWL_ID3:HU_T:fPt_ID3
            Variance StdDev
(Intercept)   0.6397 0.7998

Gamma shape parameter: 2.8025 (std. err.: 0.13606)

Log-likelihood: -4761.94
```


Histogram of residuals(m.s1)
residuals(m.s1)
fitted(m.s1)
Frequency
residuals(m.s1)
residuals(m.s1)
bayhead
cypress
marsh
upland
Data.C.seg$Pt_Com
residuals(m.s1)
fWL_ID3

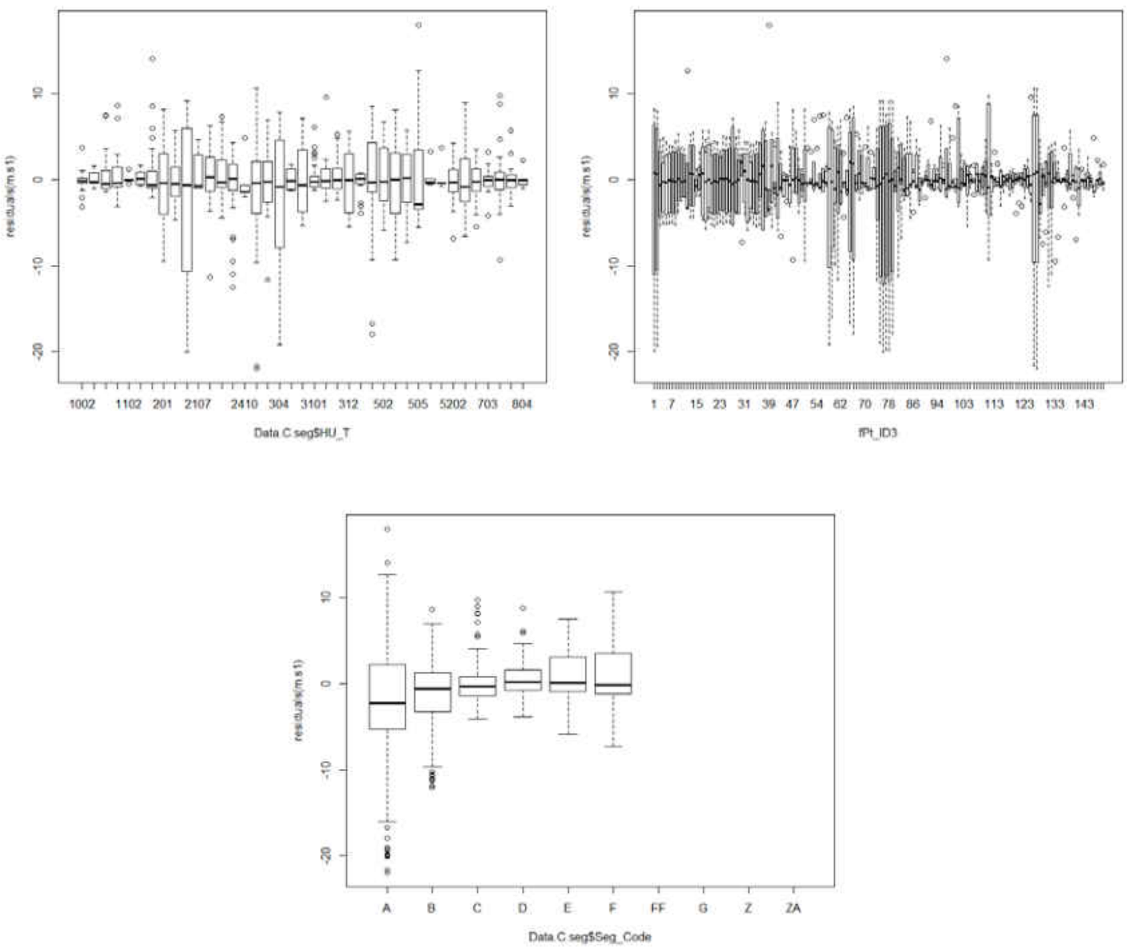
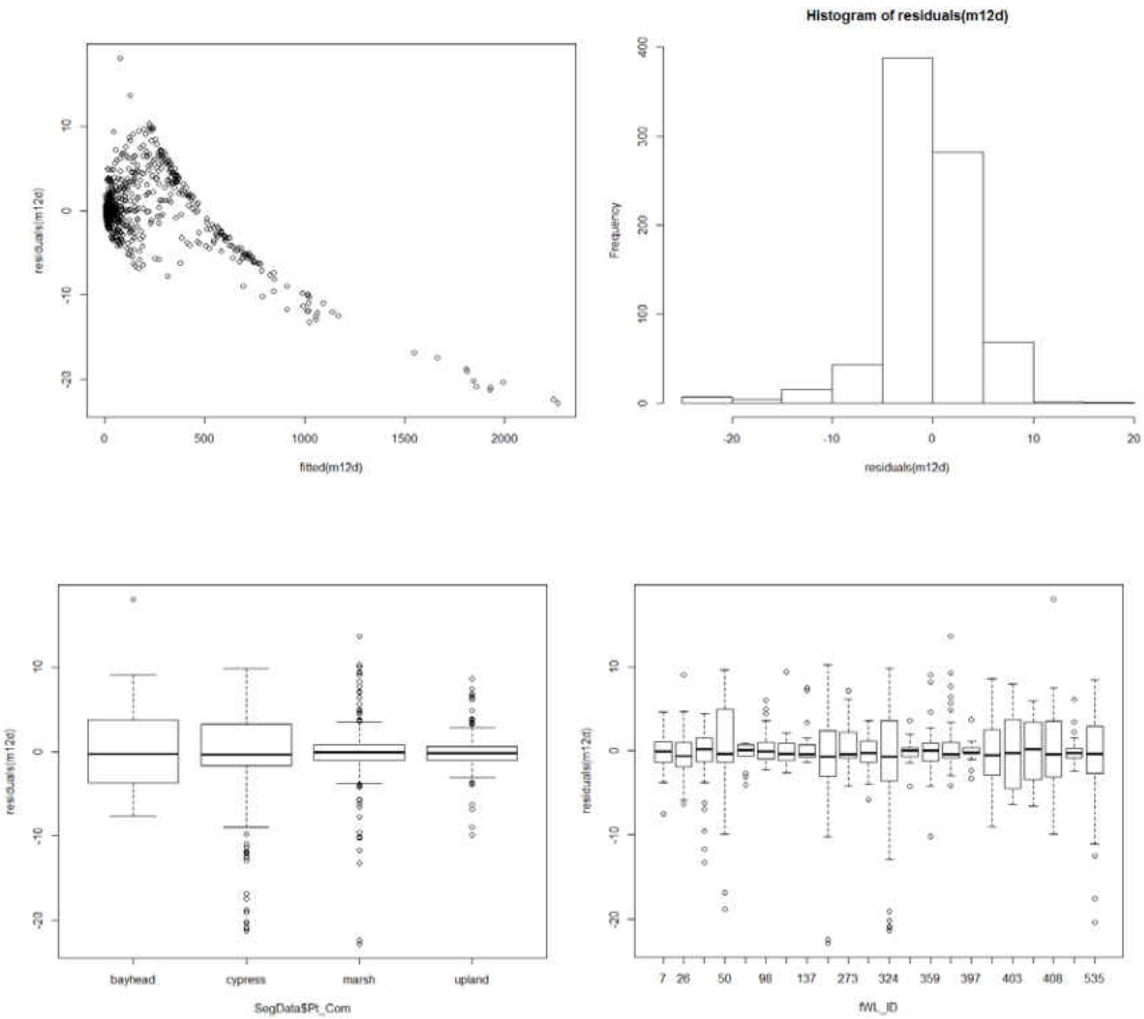


Figure G- 1. Residual plots for soil carbon concentration ~ vegetative community*soil depth model.

APPENDIX H:
SOIL CARBON CONCENTRATION ~ WATER DEPTH MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = C_Conc ~ WD_avg * Pt_Com + Seg_Code * Pt_Com +
    (1 | fWL_ID/HU_T/fPt_ID), data = SegData, family = "Gamma")

AIC: 8329.1

Coefficients:
                        Estimate Std. Error z value Pr(>|z|)
(Intercept)               4.6341     0.2878   16.10  < 2e-16 ***
WD_avg                    0.4095     0.3280    1.25  0.21179
Pt_Combayhead             0.8263     0.3467    2.38  0.01714 *
Pt_Comcypress             0.9525     0.3910    2.44  0.01486 *
Pt_Commarsh               0.0290     0.3542    0.08  0.93472
Seg_CodeB                -0.8892     0.1547   -5.75  9.0e-09 ***
Seg_CodeC                -1.7462     0.1569  -11.13  < 2e-16 ***
Seg_CodeD                -2.3340     0.1585  -14.73  < 2e-16 ***
Seg_CodeE                -2.4126     0.1593  -15.15  < 2e-16 ***
Seg_CodeF                -2.2910     0.1604  -14.28  < 2e-16 ***
WD_avg:Pt_Combayhead      1.3409     0.6871    1.95  0.05098 .
WD_avg:Pt_Comcypress      1.0321     0.9144    1.13  0.25905
WD_avg:Pt_Commarsh        2.7393     0.6187    4.43  9.5e-06 ***
Pt_Combayhead:Seg_CodeB   0.8366     0.2263    3.70  0.00022 ***
Pt_Comcypress:Seg_CodeB   0.2865     0.2176    1.32  0.18793
Pt_Commarsh:Seg_CodeB     0.0947     0.2024    0.47  0.63981
Pt_Combayhead:Seg_CodeC   1.5310     0.2290    6.68  2.3e-11 ***
Pt_Comcypress:Seg_CodeC   0.4805     0.2213    2.17  0.02993 *
Pt_Commarsh:Seg_CodeC     0.3874     0.2057    1.88  0.05963 .
Pt_Combayhead:Seg_CodeD   1.7626     0.2313    7.62  2.5e-14 ***
Pt_Comcypress:Seg_CodeD   0.6280     0.2252    2.79  0.00529 **
Pt_Commarsh:Seg_CodeD     0.5025     0.2076    2.42  0.01546 *
Pt_Combayhead:Seg_CodeE   1.6689     0.2335    7.15  8.8e-13 ***
Pt_Comcypress:Seg_CodeE   0.4725     0.2272    2.08  0.03758 *
Pt_Commarsh:Seg_CodeE     0.3200     0.2084    1.53  0.12479
Pt_Combayhead:Seg_CodeF   1.4313     0.2355    6.08  1.2e-09 ***
Pt_Comcypress:Seg_CodeF   0.2160     0.2293    0.94  0.34602
Pt_Commarsh:Seg_CodeF    -0.0229     0.2102   -0.11  0.91305
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=810, fWL_ID=22, fWL_ID:HU_T=39, fWL_ID:HU_T:
fPt_ID=135
Random effect variance(s):
Group=fWL_ID
            Variance StdDev
(Intercept)    0.312 0.5585
Group=fWL_ID:HU_T
            Variance StdDev
(Intercept)   0.3674 0.6062
Group=fWL_ID:HU_T:fPt_ID
            Variance StdDev
(Intercept)   0.4513 0.6718

Gamma shape parameter: 2.7811 (std. err.: 0.14217)

Log-likelihood: -4132.55
```



Histogram of residuals(m12d)
residuals(m12d)
fitted(m12d)
Frequency
residuals(m12d)
residuals(m12d)
bayhead
cypress
marsh
upland
SegData$Pt_Com
residuals(m12d)
7 26 50 98 137 273 324 359 397 403 408 535
fWL_ID

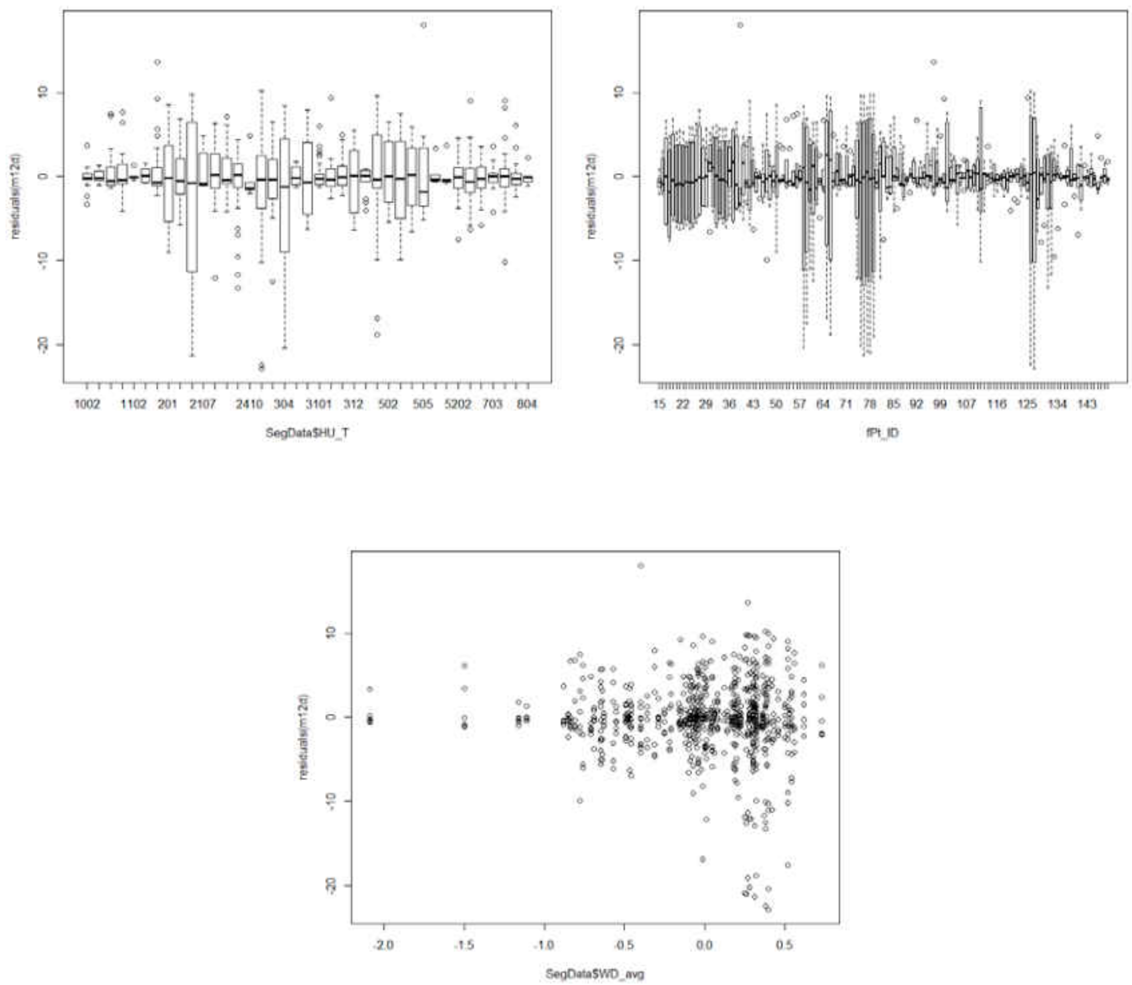


Figure H- 1. Residual plots for soil carbon concentration ~ mean water depth, soil depth, and vegetative community model.

APPENDIX I:
SOIL BULK DENSITY ~ COMMUNITY MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = Density ~ Pt_Com + (1 | fWL_ID4/HU_T), data = Data.dens
ity.whole, family = "Gamma")

AIC: 182.7

Coefficients:
              Estimate Std. Error z value Pr(>|z|)
(Intercept)      0.386      0.121    3.20   0.0014 **
Pt_Combayhead   -1.301      0.162   -8.01  1.2e-15 ***
Pt_Comcypress   -0.783      0.163   -4.80  1.6e-06 ***
Pt_Commarsh     -0.255      0.165   -1.54   0.1227
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=149, fWL_ID4=22, fWL_ID4:HU_T=39
Random effect variance(s):
Group=fWL_ID4
             Variance    StdDev
(Intercept) 8.661e-07 0.0009306
Group=fWL_ID4:HU_T
            Variance StdDev
(Intercept)   0.1739  0.417

Gamma shape parameter: 3.4662 (std. err.: 0.43212)

Log-likelihood: -84.3353
```

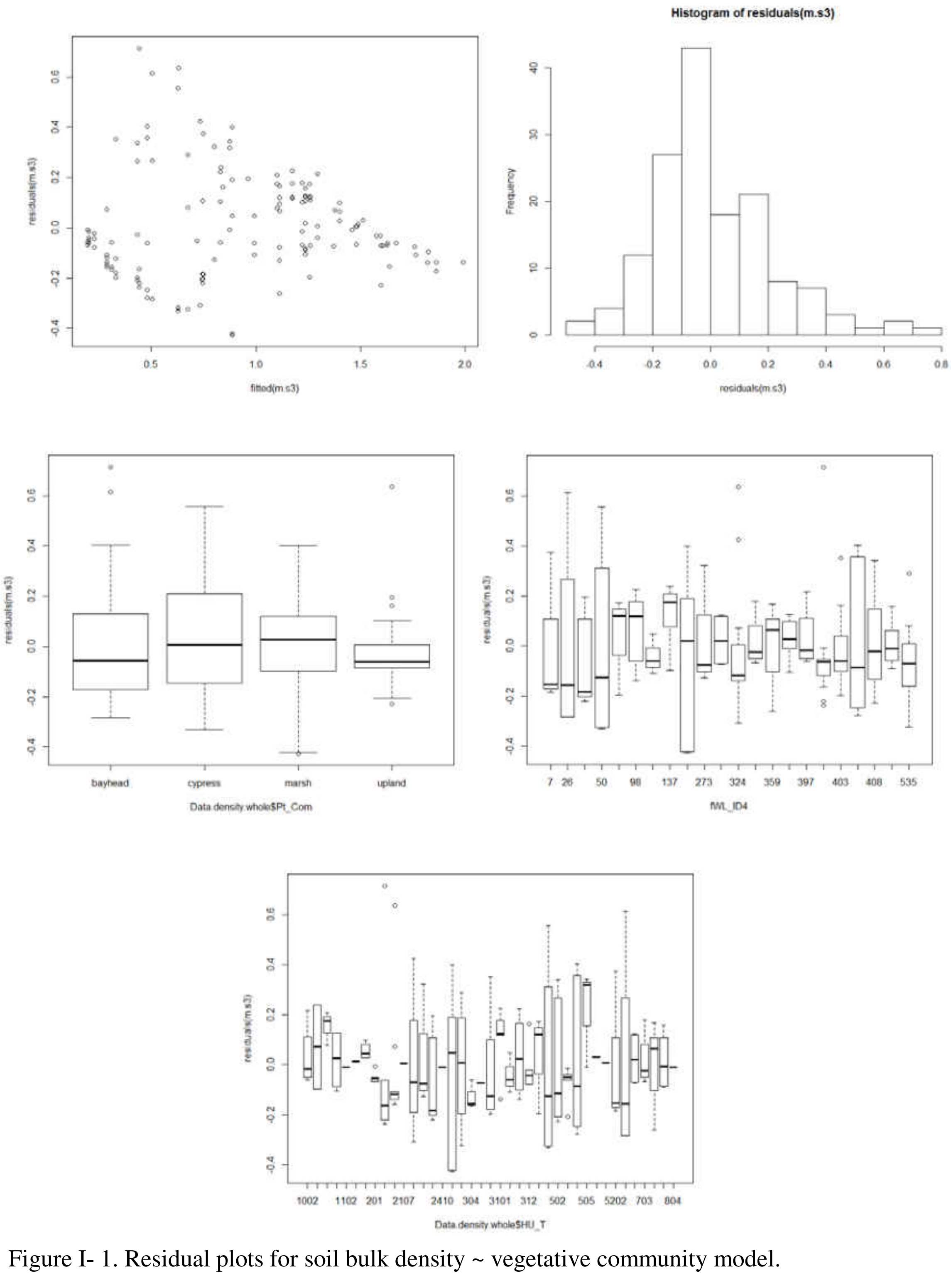
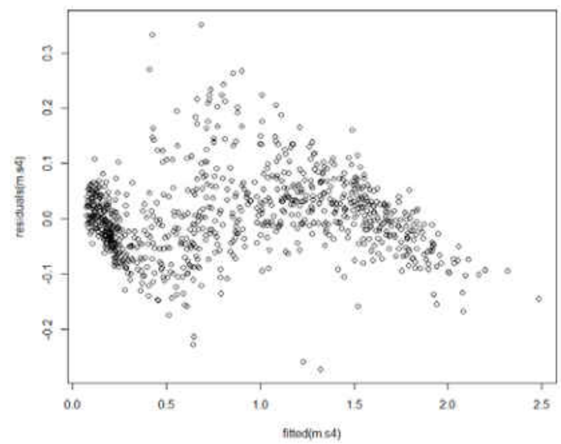
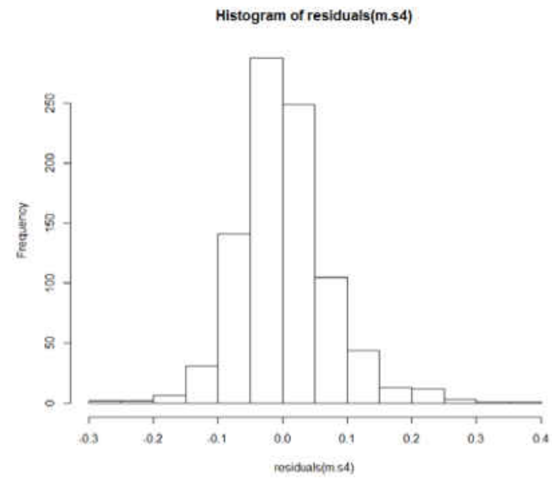


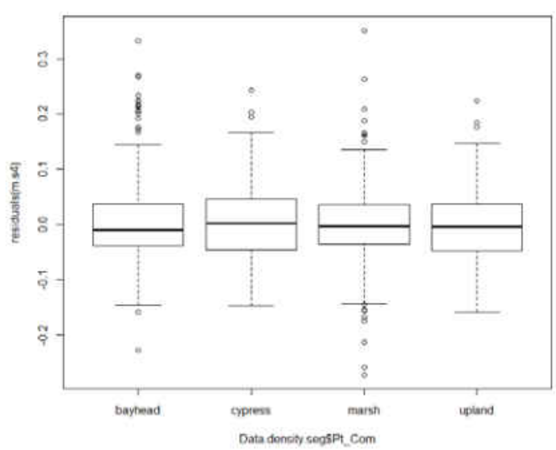
Figure I- 1. Residual plots for soil bulk density ~ vegetative community model.

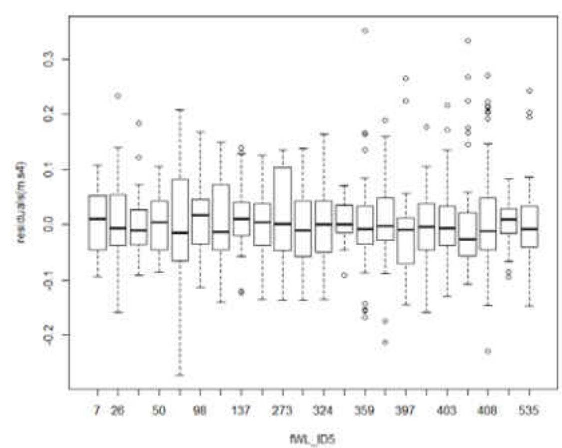
APPENDIX J:
SOIL BULK DENSITY ~ SOIL DEPTH MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = Density ~ Pt_Com * Seg_Code + (1 | fWL_ID5/HU_T/fPt_ID5
), data = Data.density.seg, family = "Gamma")

AIC: 37.6

Coefficients:
                          Estimate Std. Error z value Pr(>|z|)
(Intercept)               -0.43515    0.12861   -3.38  0.00072 ***
Pt_Combayhead             -1.35666    0.16819   -8.07  7.3e-16 ***
Pt_Comcypress             -0.85012    0.17927   -4.74  2.1e-06 ***
Pt_Commarsh               -0.38108    0.17466   -2.18  0.02912 *
Seg_CodeB                  0.52858    0.08189    6.45  1.1e-10 ***
Seg_CodeC                  0.77161    0.08213    9.39  < 2e-16 ***
Seg_CodeD                  0.86245    0.08210   10.50  < 2e-16 ***
Seg_CodeE                  0.92465    0.08296   11.15  < 2e-16 ***
Seg_CodeF                  0.93820    0.08293   11.31  < 2e-16 ***
Pt_Combayhead:Seg_CodeB -0.25723    0.10980   -2.34  0.01914 *
Pt_Comcypress:Seg_CodeB -0.17387    0.11416   -1.52  0.12776
Pt_Commarsh:Seg_CodeB   -0.00178    0.10768   -0.02  0.98681
Pt_Combayhead:Seg_CodeC -0.24758    0.11020   -2.25  0.02466 *
Pt_Comcypress:Seg_CodeC -0.16494    0.11459   -1.44  0.15004
Pt_Commarsh:Seg_CodeC    0.03742    0.10789    0.35  0.72872
Pt_Combayhead:Seg_CodeD -0.17428    0.11049   -1.58  0.11471
Pt_Comcypress:Seg_CodeD -0.11306    0.11488   -0.98  0.32502
Pt_Commarsh:Seg_CodeD    0.06888    0.10794    0.64  0.52336
Pt_Combayhead:Seg_CodeE -0.06790    0.11152   -0.61  0.54264
Pt_Comcypress:Seg_CodeE -0.04304    0.11570   -0.37  0.70990
Pt_Commarsh:Seg_CodeE    0.07453    0.10878    0.69  0.49327
Pt_Combayhead:Seg_CodeF -0.08060    0.11180   -0.72  0.47096
Pt_Comcypress:Seg_CodeF -0.01975    0.11585   -0.17  0.86462
Pt_Commarsh:Seg_CodeF    0.13013    0.10888    1.20  0.23202
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=898, fWL_ID5=22, fWL_ID5:HU_T=39, fWL_ID5:HU
_T:fPt_ID5=150
Random effect variance(s):
Group=fWL_ID5
            Variance  StdDev
(Intercept) 0.009345 0.09667
Group=fWL_ID5:HU_T
            Variance StdDev
(Intercept)   0.1468 0.3831
Group=fWL_ID5:HU_T:fPt_ID5
            Variance StdDev
(Intercept)   0.2602 0.5101

Gamma shape parameter: 9.401 (std. err.: 0.4763)

Log-likelihood: 9.20738
```


residuals(m.s4)
fitted(m.s4)


Histogram of residuals(m.s4)
Frequency
residuals(m.s4)


residuals(m.s4)
bayhead
cypress
marsh
upland
Data density seg$Pt_Com


residuals(m.s4)
7 26 50 98 137 273 324 359 397 403 408 535
fWL_ID5

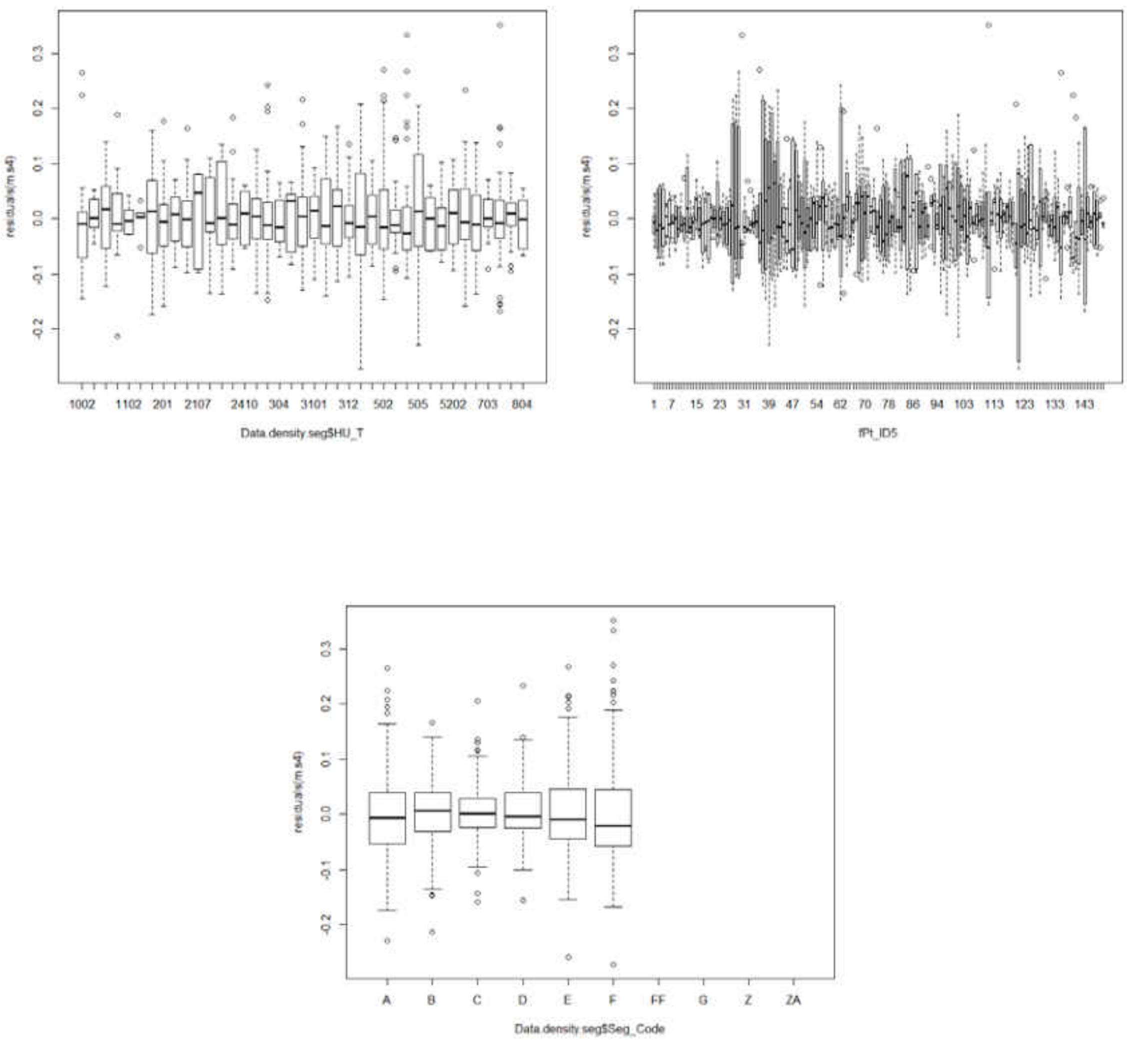


Figure J- 1. Residual plots for soil bulk density ~ vegetative community*soil depth model.

APPENDIX K:
SOIL CARBON STOCK ~ COMMUNITY MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = C_Stock ~ Pt_Com + (1 | fWL_ID3/HU_T), data = Data.carb
on.whole, family = "Gamma")

AIC: 1148

Coefficients:
              Estimate Std. Error z value Pr(>|z|)
(Intercept)      2.421      0.122   19.78  < 2e-16 ***
Pt_Combayhead    1.004      0.153    6.57  5.1e-11 ***
Pt_Comcypress    0.617      0.164    3.77  0.00016 ***
Pt_Commarsh      0.522      0.157    3.33  0.00086 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=149, fWL_ID3=22, fWL_ID3:HU_T=39
Random effect variance(s):
Group=fWL_ID3
            Variance StdDev
(Intercept)  0.06477 0.2545
Group=fWL_ID3:HU_T
            Variance StdDev
(Intercept)  0.02883 0.1698

Gamma shape parameter: 3.3757 (std. err.: 0.42936)

Log-likelihood: -567.01
```

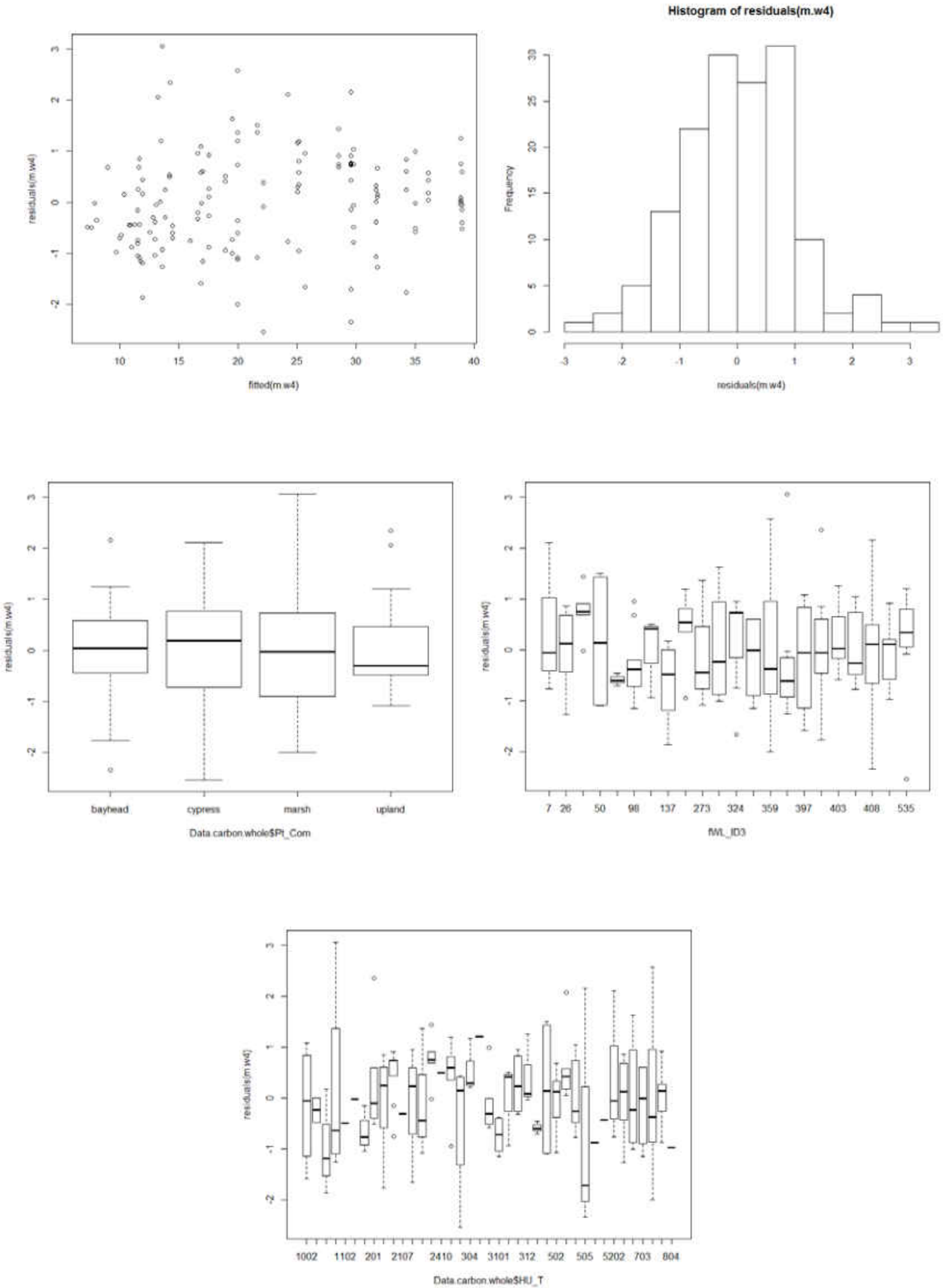
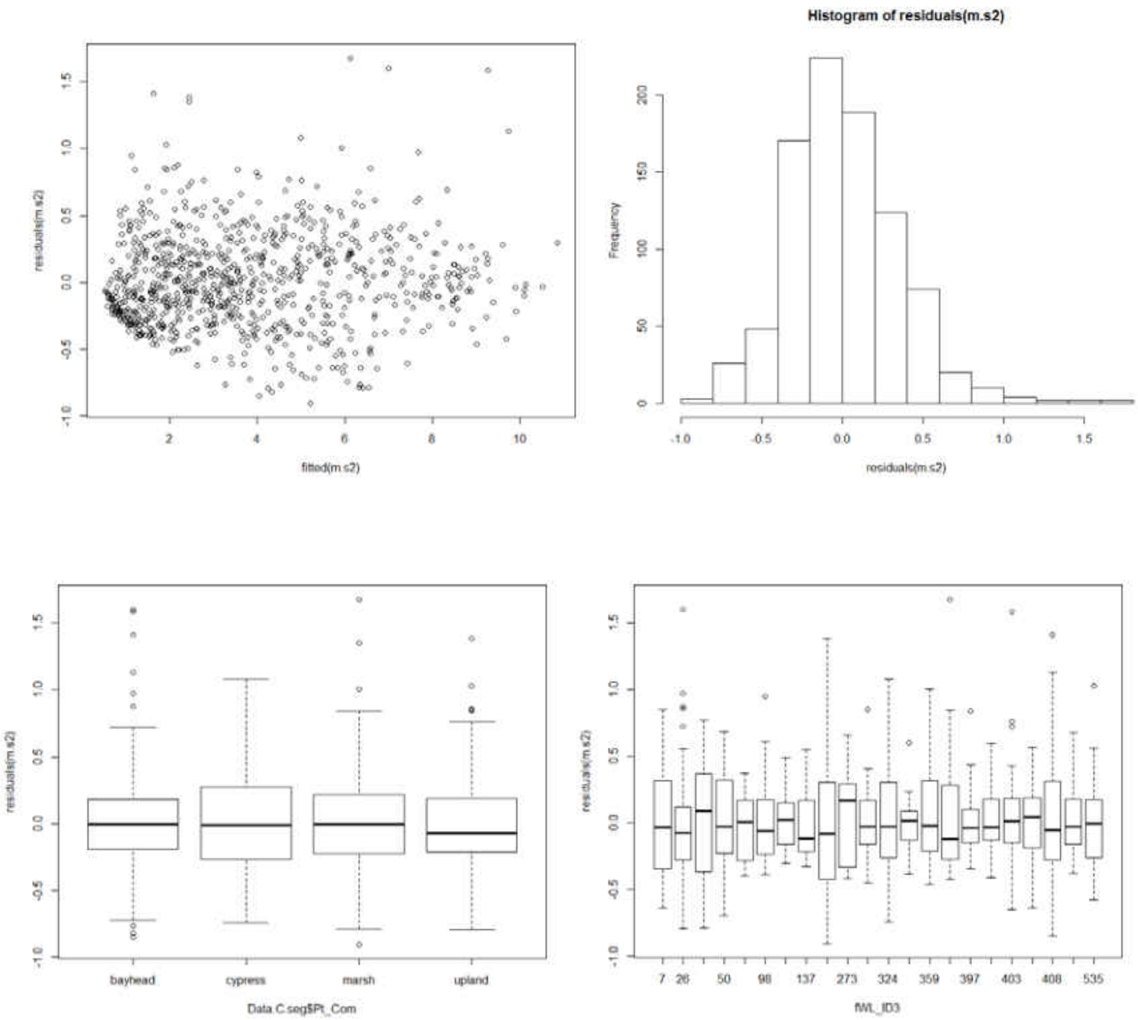


Figure K- 1. Residual plots for soil carbon stock ~ vegetative community model.

APPENDIX L:
SOIL CARBON STOCK ~ SOIL DEPTH MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = C_Stock ~ Pt_Com * Seg_Code + (1 | fWL_ID3/HU_T/fPt_ID3
), data = Data.C.seg, family = "Gamma")

AIC: 3418.8

Coefficients:
                        Estimate Std. Error z value Pr(>|z|)
(Intercept)                0.742      0.165    4.50  6.7e-06 ***
Pt_Comcypress              0.518      0.217    2.39   0.0167 *
Pt_Commarsh                0.357      0.204    1.75   0.0800 .
Pt_Comupland               0.177      0.172    1.02   0.3056
Seg_CodeB                  0.254      0.108    2.36   0.0184 *
Seg_CodeC                  1.074      0.109    9.88  < 2e-16 ***
Seg_CodeD                  0.983      0.109    9.00  < 2e-16 ***
Seg_CodeE                  0.917      0.109    8.39  < 2e-16 ***
Seg_CodeF                  0.702      0.109    6.43  1.3e-10 ***
Pt_Comcypress:Seg_CodeB   -0.333      0.159   -2.09   0.0369 *
Pt_Commarsh:Seg_CodeB     -0.377      0.149   -2.52   0.0117 *
Pt_Comupland:Seg_CodeB    -0.523      0.162   -3.23   0.0013 **
Pt_Comcypress:Seg_CodeC   -0.732      0.161   -4.54  5.6e-06 ***
Pt_Commarsh:Seg_CodeC     -0.767      0.151   -5.08  3.9e-07 ***
Pt_Comupland:Seg_CodeC    -1.323      0.163   -8.10  5.4e-16 ***
Pt_Comcypress:Seg_CodeD   -0.949      0.162   -5.84  5.1e-09 ***
Pt_Commarsh:Seg_CodeD     -1.019      0.152   -6.70  2.0e-11 ***
Pt_Comupland:Seg_CodeD    -1.697      0.164  -10.33  < 2e-16 ***
Pt_Comcypress:Seg_CodeE   -0.994      0.163   -6.09  1.1e-09 ***
Pt_Commarsh:Seg_CodeE     -1.144      0.152   -7.52  5.7e-14 ***
Pt_Comupland:Seg_CodeE    -1.625      0.166   -9.80  < 2e-16 ***
Pt_Comcypress:Seg_CodeF   -0.959      0.163   -5.88  4.2e-09 ***
Pt_Commarsh:Seg_CodeF     -1.156      0.152   -7.58  3.4e-14 ***
Pt_Comupland:Seg_CodeF    -1.295      0.166   -7.78  7.1e-15 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=898, fWL_ID3=22, fWL_ID3:HU_T=39, fWL_ID3:HU
_T:fPt_ID3=150
Random effect variance(s):
Group=fWL_ID3
            Variance StdDev
(Intercept)  0.09518 0.3085
Group=fWL_ID3:HU_T
            Variance StdDev
(Intercept)  0.06454  0.254
Group=fWL_ID3:HU_T:fPt_ID3
            Variance StdDev
(Intercept)   0.1465 0.3828

Gamma shape parameter: 4.3645 (std. err.: 0.21705)

Log-likelihood: -1681.42
```



Histogram of residuals(m.s2)
residuals(m.s2)
fitted(m.s2)
Frequency
bayhead
cypress
marsh
upland
Data.C.seg$Pt_Com
fWL_ID3

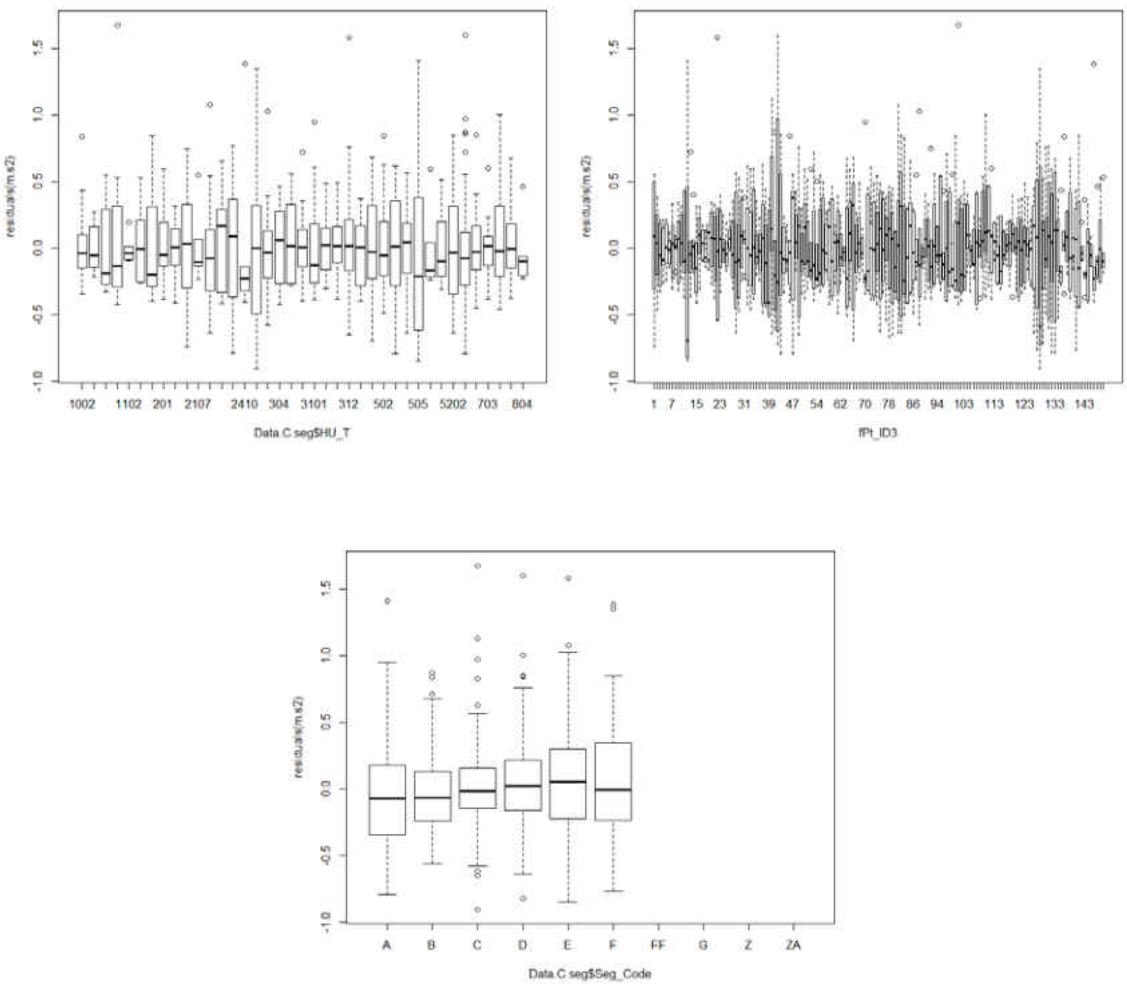
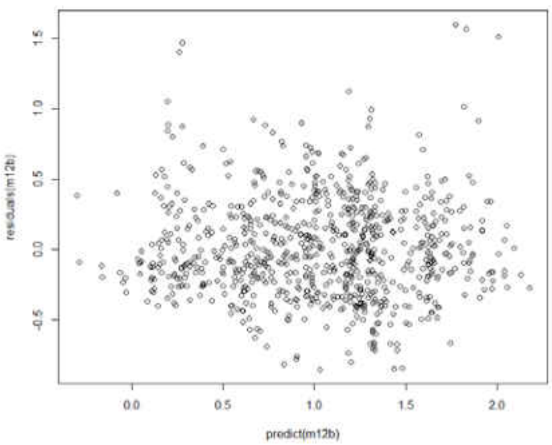
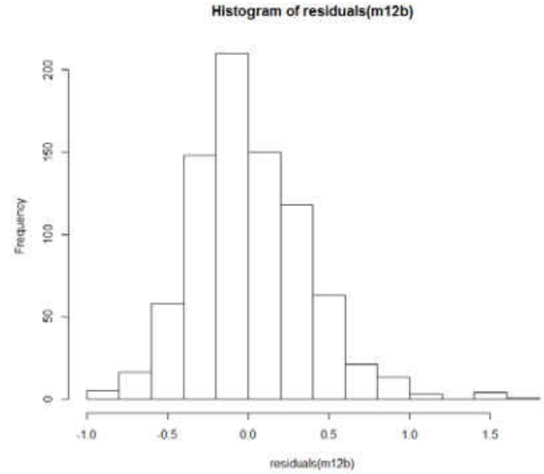


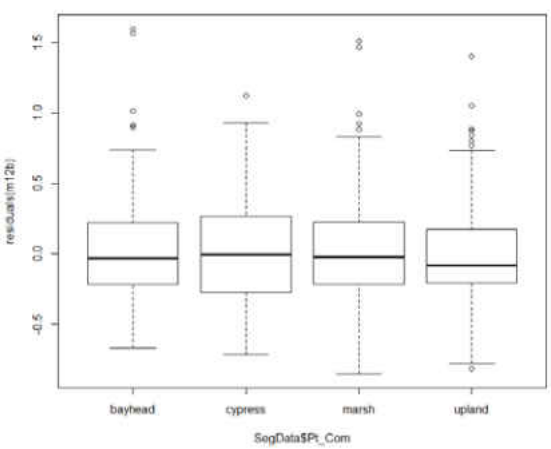
Figure L- 1. Residual plots for soil carbon stock ~ vegetative community*soil depth.

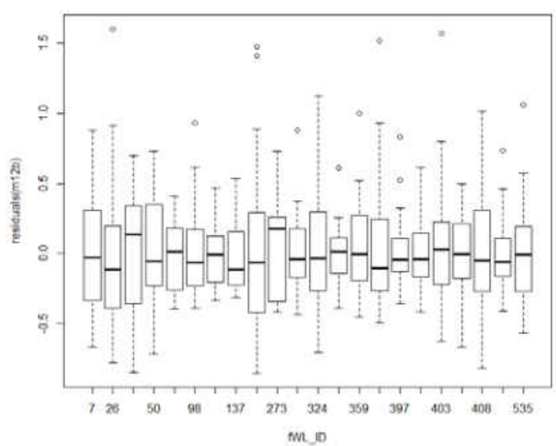
APPENDIX M: SOIL CARBON STOCK ~ WATER DEPTH MODEL OUTPUT AND RESIDUAL PLOTS

```
Call:
glmmadmb(formula = C_Stock ~ WD_avg * Pt_Com + Seg_Code * Pt_Com +
    (1 | fWL_ID/HU_T/fPt_ID), data = SegData, family = "Gamma")

AIC: 2997.1

Coefficients:
                         Estimate Std. Error z value Pr(>|z|)
(Intercept)                1.0080     0.1664    6.06  1.4e-09 ***
WD_avg                     0.2233     0.1776    1.26  0.20862
Pt_Combayhead             -0.3360     0.2095   -1.60  0.10876
Pt_Comcypress              0.2665     0.2343    1.14  0.25541
Pt_Commarsh               -0.1193     0.2056   -0.58  0.56174
Seg_CodeB                 -0.2843     0.1238   -2.30  0.02172 *
Seg_CodeC                 -0.2536     0.1248   -2.03  0.04207 *
Seg_CodeD                 -0.7084     0.1258   -5.63  1.8e-08 ***
Seg_CodeE                 -0.7050     0.1266   -5.57  2.6e-08 ***
Seg_CodeF                 -0.5876     0.1274   -4.61  4.0e-06 ***
WD_avg:Pt_Combayhead       0.3501     0.3705    0.94  0.34470
WD_avg:Pt_Comcypress      -0.0452     0.5111   -0.09  0.92959
WD_avg:Pt_Commarsh         1.2591     0.3439    3.66  0.00025 ***
Pt_Combayhead:Seg_CodeB    0.6114     0.1795    3.41  0.00066 ***
Pt_Comcypress:Seg_CodeB    0.1869     0.1736    1.08  0.28169
Pt_Commarsh:Seg_CodeB      0.1536     0.1616    0.95  0.34201
Pt_Combayhead:Seg_CodeC    1.4507     0.1802    8.05  8.2e-16 ***
Pt_Comcypress:Seg_CodeC    0.5663     0.1753    3.23  0.00124 **
Pt_Commarsh:Seg_CodeC      0.5433     0.1633    3.33  0.00088 ***
Pt_Combayhead:Seg_CodeD    1.7833     0.1812    9.84  < 2e-16 ***
Pt_Comcypress:Seg_CodeD    0.6953     0.1770    3.93  8.5e-05 ***
Pt_Commarsh:Seg_CodeD      0.6554     0.1645    3.98  6.8e-05 ***
Pt_Combayhead:Seg_CodeE    1.6783     0.1818    9.23  < 2e-16 ***
Pt_Comcypress:Seg_CodeE    0.5698     0.1782    3.20  0.00139 **
Pt_Commarsh:Seg_CodeE      0.4625     0.1653    2.80  0.00515 **
Pt_Combayhead:Seg_CodeF    1.3041     0.1822    7.16  8.3e-13 ***
Pt_Comcypress:Seg_CodeF    0.2686     0.1789    1.50  0.13334
Pt_Commarsh:Seg_CodeF      0.1241     0.1663    0.75  0.45540
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Number of observations: total=810, fWL_ID=22, fWL_ID:HU_T=39, fWL_ID:HU_T:
fPt_ID=135
Random effect variance(s):
Group=fWL_ID
            Variance StdDev
(Intercept)  0.09198 0.3033
Group=fWL_ID:HU_T
            Variance StdDev
(Intercept)  0.07148 0.2674
Group=fWL_ID:HU_T:fPt_ID
            Variance StdDev
(Intercept)   0.1255 0.3542

Gamma shape parameter: 4.3041 (std. err.: 0.22518)

Log-likelihood: -1466.56
```


residuals(m12b)
predict(m12b)


Histogram of residuals(m12b)
Frequency
residuals(m12b)


residuals(m12b)
bayhead
cypress
marsh
upland
SegData$Pt_Com


residuals(m12b)
fWL_ID

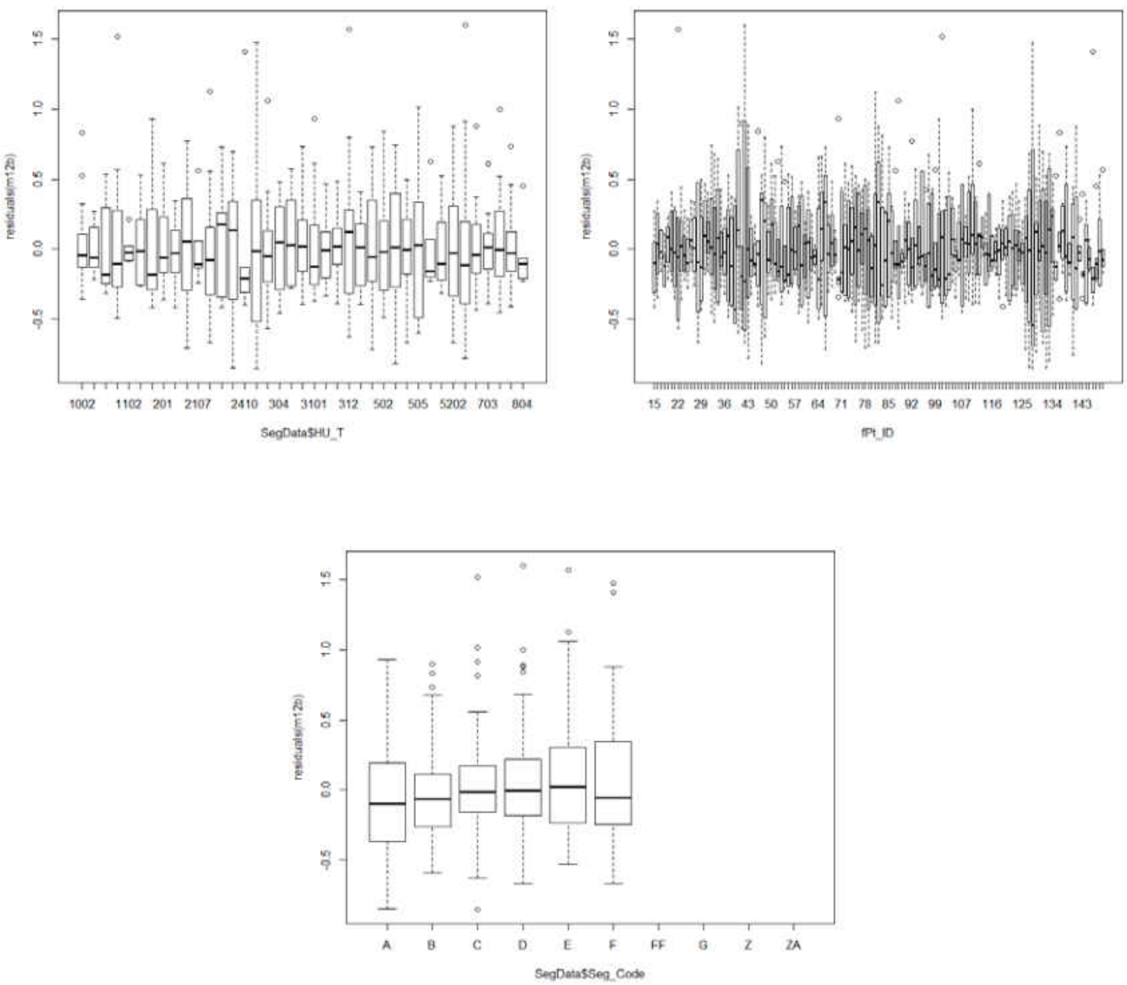


Figure M- 1. Residual plots for soil carbon stock ~ water depth, vegetative community, and soil depth model.

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