Journal of Hydrology 644 (2024) 132052

Contents lists available at ScienceDirect

Journal of Hydrology

journal homepage: www.elsevier.com/locate/jhydrol

Research papers

Wetland hydrological change and recovery across three decades of shifting groundwater management

Jessica A. Balerna [a,*], Andrew M. Kramer [a], Shawn M. Landry [b], Mark C. Rains [b], David B. Lewis [a]

[a] Department of Integrative Biology, University of South Florida, Tampa, FL 33620, USA
[b] School of Geosciences, University of South Florida, Tampa, FL 33620, USA

ARTICLE INFO

Keywords:
Hydrology
Hydroperiod
Geographically isolated wetlands
Freshwater wetlands
Groundwater extraction
Groundwater conservation

ABSTRACT

Groundwater extraction compromises the function of groundwater-dependent ecosystems, such as freshwater wetlands. Identifying whether groundwater conservation restores wetland hydrology is a first step toward rehabilitating impaired wetlands. In the Tampa Bay region of Florida (U.S.), groundwater extraction rates have been declining since 1998, partly in response to desiccation of wetlands and waterbodies. This study uses monthly water-level data from 152 depressional wetlands over 28 years (1991–2018) to identify trends in wetland inundation, determine whether those trends vary among wetlands historically exposed to different rates of groundwater extraction, and describe relationships between the timing and extent of cutbacks in groundwater extraction and the timing and extent of changes in wetland inundation. Many wetlands (57 %) exhibited increased inundation in response to cutbacks in groundwater extraction, indicating that water conservation measures are inducing recovery. Further, increased inundation began in most wetlands immediately upon, or within two years of, the time extraction cutbacks occurred, although some recovering wetlands exhibited longer lags. An additional 26 % of wetlands had steady-state water levels with inundation similar to that of reference wetlands, potentially revealing a population of wetlands hydrologically unimpaired by nearby groundwater extraction. Another subset of wetlands (14 %) with steady-state water depths exhibited increasing deviations from basin-full water levels, suggesting subsidence of the wetland basin. Active intervention beyond cutbacks in groundwater extraction may be necessary to restore this subset, whereas passive restoration (reducing extraction) appears adequate for the majority of impacted wetlands. Rising water levels may amplify surface-water connections among wetlands, with ecological and biogeochemical consequences both for individual wetlands and for the whole wetlandscape. As a host of human activities continue to rely on groundwater extraction, this study demonstrates the potential for, as well as variability in, hydrological recovery across a wetland-rich, low-relief landscape following the enactment of water conservation policies.

1. Introduction

Freshwater depressional wetlands help protect downstream waters from pollution, store carbon, and foster biodiversity (Evenson et al., 2018; Marton et al., 2015; McLaughlin and Cohen, 2013). Yet depressional wetlands often lie outside of floodplains and lack permanent surface water connections to surrounding freshwater bodies (Creed et al., 2017), so are not afforded consistent protections from anthropogenic disturbance, drainage, and destruction (Adler, 2015; Mihelcic and Rains, 2020; Rains et al., 2016). The U.S. Fish and Wildlife Service estimates 53 % of the wetland area in the United States was lost between 1780 and 1980 primarily due to agricultural drainage and land development (Dahl, 1990). Population growth in urban areas has compelled cities to increasingly rely on groundwater extraction for their water supply, further putting groundwater-dependent ecosystems like depressional wetlands at risk (Bierkens and Wada, 2019; Jame and Bowling, 2020). Wetland restoration projects can help reverse these trends and better protect the many ecosystem services wetlands provide.

Wetland restoration projects can include passive or active restoration strategies (Holl and Aide, 2011). Passive restoration includes stopping or removing a disturbance, such as water drainage, and allowing an ecosystem to respond without further human intervention. A review of studies shows that hydrological recovery in wetlands may occur within a decade of initiating passive restoration approaches while biological

* Corresponding author at: Rubenstein School of Environment and Natural Resources, University of Vermont, Burlington, VT 05405, USA.
E-mail address: jessica.balerna@uvm.edu (J.A. Balerna).

https://doi.org/10.1016/j.jhydrol.2024.132052
Received 29 November 2023; Received in revised form 13 August 2024; Accepted 9 September 2024
Available online 25 September 2024

structure and biogeochemical function may take over 100 years to reach conditions similar to undisturbed wetlands (Moreno-Mateos et al., 2012). Active restoration strategies employ additional interventions such as invasive species removal, native species replanting, or back-filling sinkholes, and may help accelerate recovery but at high labor and monetary costs (Holl and Aide, 2011).

Groundwater extraction often occurs at a rate faster than an aquifer can be replenished from precipitation and groundwater infiltration, causing depletion of groundwater storage and thus lower water tables (Alley, 2002; Konikow and Kendy, 2005; Shahid and Hazarika, 2010). Lower water tables can alter the timing and duration of when depressional wetlands hold surface water either through decreased water inputs (e.g., loss of discharge from groundwater to the wetland) or increased water outputs (e.g., increase in recharge from the wetland to groundwater; McLaughlin et al., 2014; Nilsson et al., 2013). While there is limited research on passive restoration of wetlands following cutbacks in groundwater extraction, previous studies have shown that passive restoration of wetlands impaired by agricultural drainage increased wetland hydroperiods within five years (De Steven et al., 2010). A combination of passive and active restoration strategies may be needed if persistent hydrological disturbance from groundwater extraction caused geological change such as collapse of underlying strata that prevent wetlands from holding surface water (Metz, 2011) or ecological change such as the proliferation of invasive species (Beas et al., 2013).

Wetlands are often hydrologically linked across regional wetlandscapes (Bertassello et al., 2020; Cohen et al., 2016), and lower water tables resulting from groundwater extraction could reduce surface-water connections among them by limiting spillover (Leibowitz et al., 2016). Yet, wetlands across a wetlandscape may also vary in their sensitivity to groundwater extraction owing to differences in basin topography and underlying hydrogeology. In places with no confining unit between wetland basins and an underlying aquifer, for instance, wetland inundation may respond rapidly to changes in groundwater extraction or be a direct manifestation of the aquifer (Nowicki et al., 2022, 2021). Conversely, in areas with confining units, wetland inundation may be unaffected by extraction and other drivers of variation in the aquifer (Rains et al., 2006). Persistent overextraction of groundwater can additionally create new fissures in underlying strata, which, following passive restoration, can alter wetland surface water quantity and quality (Castaño et al., 2018). The structure and connectivity of a wetlandscape may accordingly change over time owing to the spatially variable response of the water table and wetland inundation to groundwater extraction.

The Tampa Bay region of Florida (U.S.) contains over 800 km^2 of wetland area interacting with heterogeneous topography and stratigraphy (Rains et al., 2013), as well as shifting groundwater management policies, that included region-wide groundwater conservation measures enacted between 1998 and 2010 (Asefa et al., 2014). Following these shifts in groundwater management, researchers began assessing environmental changes in the Tampa Bay region, including in depressional wetlands. Previous reports by the U.S. Geological Survey investigated small samples (<15) of wetlands in the region and found variable responses in wetland inundation depending on which groundwater extraction wells wetlands were closest to (Haag and Pfeiffer, 2012; Metz, 2011). Larger studies (>1,000 wetlands) led by, or prepared for, Tampa Bay Water (TBW), the region's largest wholesale supplier of public water, assert that reductions in groundwater extraction are enhancing hydrological recovery in most depressional wetlands across the region (Hogg et al., 2020; Lee and Fouad, 2018). Balerna et al. (2023) considered how wetland inundation responded to interactive effects of precipitation, land development, basin geomorphology, and vegetation in addition to groundwater extraction for a shorter time window (2005–2018).

We build on this work by more closely assessing the trajectory, magnitude, and timing of changes in the inundation of n = 152 wetlands across the Tampa Bay region from 1991 to 2018, encompassing both before and after regional groundwater conservation measures were enacted. We seek to better understand variation in both individual wetlands as well as across wetlands within a wetlandscape by asking a series of questions: (1) How do trends in water level (increasing vs. not) during a period of groundwater conservation compare among wetlands historically exposed (pre-conservation period) to different magnitudes of groundwater extraction? (2) Are wetlands that now exhibit steady-state water levels (i.e., those exhibiting no water-level increases over this time period) failing to recover, or have they not been impacted by historic groundwater extraction? (3) For wetlands that have exhibited rising water levels during the period of groundwater conservation, does the onset of recovery vary among wetlands historically exposed to different magnitudes of groundwater extraction? (4) Does the onset of recovery coincide with the timing of cutbacks in groundwater extraction? (5) Finally, does the magnitude of change in wetland inundation correspond with the magnitude of change in groundwater extraction rates?

2. Materials and Methods

2.1. Study context

2.1.1. Climate and hydrogeology

The Tampa Bay region, which encompasses Hillsborough, Pasco, and Pinellas Counties, is a low-elevation area in the west-central portion of peninsular Florida, U.S. This region is characterized by a sub-tropical climate with a wet season from June through September. Over the past three decades, mean annual rainfall in the study area was 1,344 mm (SWFWMD, 2022).

The region is underlain by a surficial aquifer, an intermediate confining unit, and the Floridan aquifer system (Fig. 1A; S1). The surficial aquifer is, on average, 9 m thick and comprised of unconsolidated Pleistocene to Holocene deposits of sand with some clays (Arthur et al., 2008; Miller, 1997). The Upper Floridan aquifer is in the Avon Park Formation and consists of carbonate deposits. The Upper Floridan aquifer is the primary source of public water in the Tampa Bay region (Tampa Bay Water, 2022). Between the surficial aquifer and the Upper Floridan aquifer is an intermediate confining unit comprised of Hawthorn Group sediments that are primarily clay. The intermediate confining unit varies across the Tampa Bay region in both thickness and presence, influencing connectivity between depressional wetlands and the Upper Floridan aquifer (Fig. 1A). There is no study that characterizes the exact location and thickness of the intermediate confining layer beneath the wetlands used in this study. Instead, in some places, wetland water levels have been shown to be a direct manifestation of the Upper Floridan aquifer as there is no intermediate confining unit (Nowicki et al., 2022, 2021). In other places, the presence of an intermediate confining unit can slow or prevent interactions between depressional wetlands and the Upper Floridan aquifer. Depending on this connectivity, depressional wetlands will vary in how sensitive they are to changes (induced, for instance, by groundwater extraction) in the Upper Floridan aquifer (Metz, 2011).

2.1.2. History of groundwater management

TBW operates 13 wellfields across approximately 1,700 km^2 in the northern Tampa Bay region and uses extracted groundwater to help supply drinking-quality water to over 2.5 million residents. The wellfields are primarily located in publicly accessible parks that contain numerous wells for extracting groundwater from the Upper Floridan Aquifer, as well as depressional and riparian wetlands surrounded primarily by pine flatwoods upland habitat.

Prior to the creation of TBW in 1998, the West Coast Regional Water Supply Authority provided water for the region from 100 % groundwater sources (Asefa et al., 2014). Overextraction of groundwater at this time resulted in visible drainage of local lakes and wetlands. Residents responded to this environmental change with a class-action lawsuit

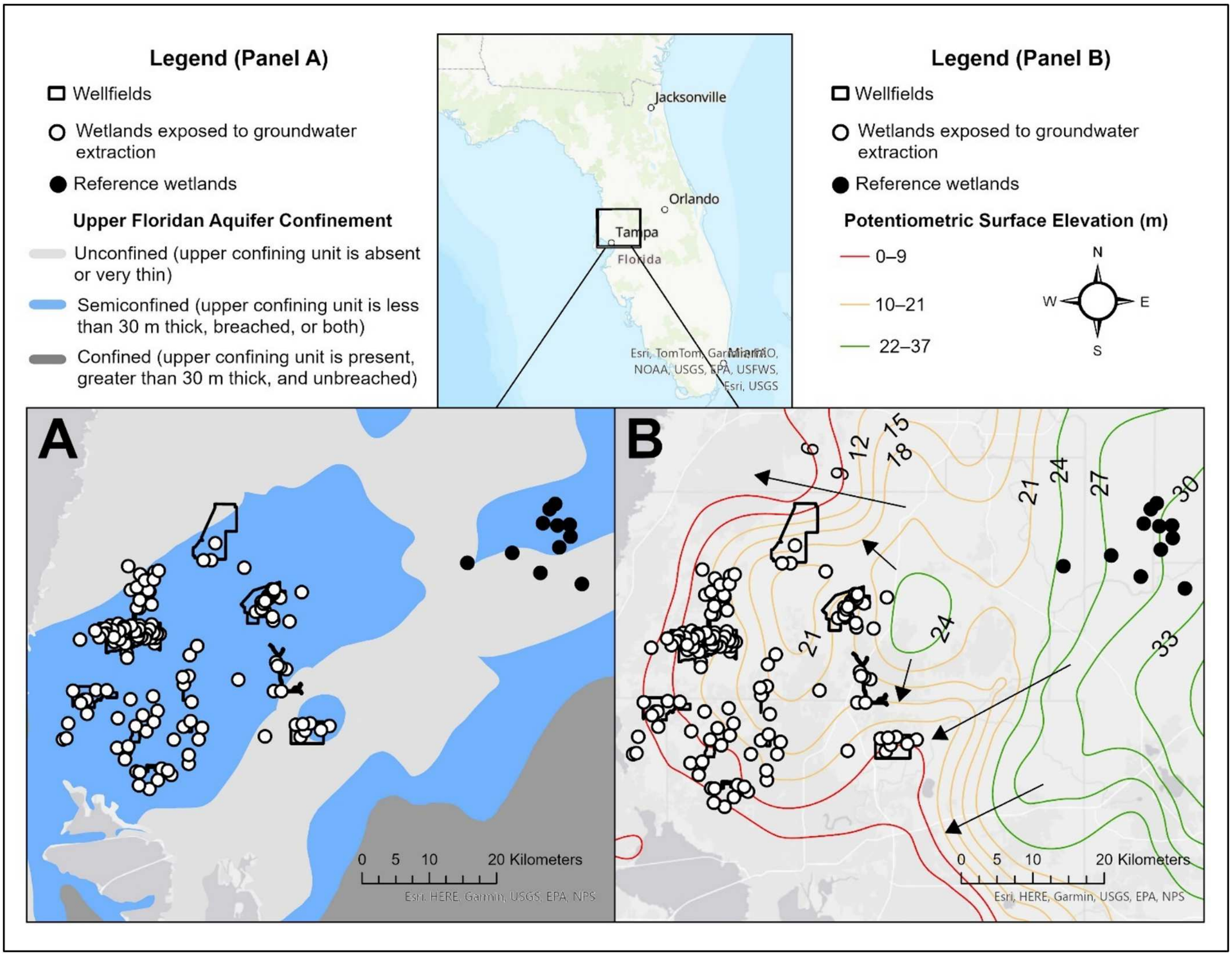

Fig. 1. Map of the exposed (open circles) and reference wetlands (closed circles) used in this study and the underlying hydrogeology. Panel A displays the regional extent of the intermediate confining layer, where both the exposed and reference wetlands are underlain by either an unconfined unit or semiconfined unit (Bellino, 2011). Panel B displays the potentiometric surface which is highest beneath the reference wetlands and lowers directionally towards the exposed wetlands and the Gulf of Mexico (SWFWMD, 2016).

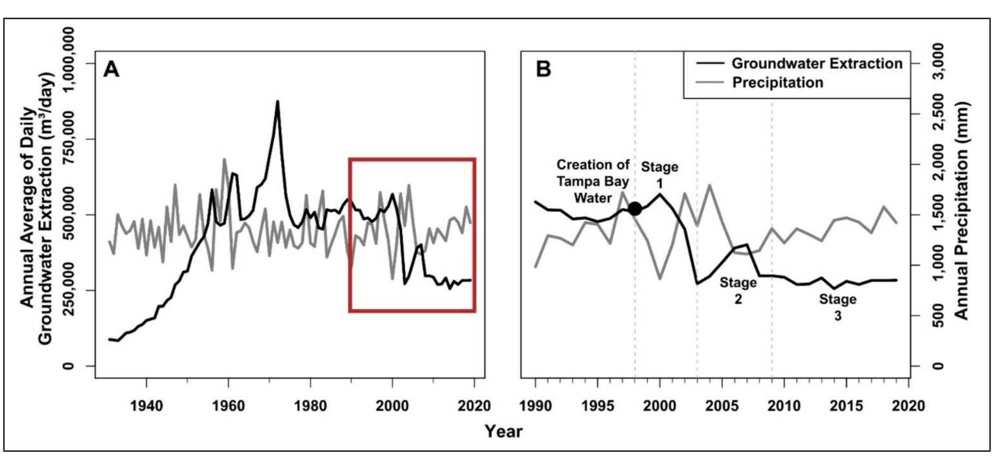

Fig. 2. A) Total groundwater extraction (m^3 day^{-1}) averaged across the wellfield complex (annual average of daily extraction volume) and annual precipitation from 1930 to 2018. B) Groundwater extraction and precipitation from 1990 to 2018, illustrating staged cutbacks in extraction that began after 1998 with the creation of Tampa Bay Water.

citing property value losses and damages from mismanagement of groundwater (Fernandez, 2013; Glennon, 2002; Rand, 2003). In response, TBW was created in 1998 to replace the West Coast Regional Water Supply Authority and was legally mandated by the courts to develop alternative water supply sources, including surface-water sources and a desalination plant, to ultimately lower their total annual extraction of groundwater (Asefa et al., 2014; Interlocal Agreement, 1998). Groundwater extraction from the TBW wellfield complex averaged approximately 720,000 m^3 day^{-1} in the 1990s before being lowered in stages to an average of approximately 600,000 m^3 day^{-1} from 1998 to 2002 (16.7 % cutback), followed by an average of approximately 400,000 m^3 day^{-1} from 2003 to 2008 (36.1 % cutback), and finally to an average of approximately 340,000 m^3 day^{-1} starting in 2009 (52.7 % cutback; Interlocal Agreement 1998; Fig. 2).

2.2. Wetland selection

We selected 141 wetlands in and around wellfields operated by TBW that span a gradient of historic exposure to groundwater extraction prior to the contemporary period of groundwater conservation (hereafter, exposed wetlands; Fig. 3). We additionally selected 11 wetlands in the Green Swamp Wilderness Preserve as reference sites (hereafter, reference wetlands).

These reference wetlands are ecologically, geomorphologically, and hydrogeologically similar to the exposed wetlands but located much further away from any groundwater extraction wells. The reference wetlands have a median (IQR) geographical distance of 40 km (3.79 km) to the closest groundwater extraction well (Balerna et al., 2023). Exposed wetlands, by comparison, have a median (IQR) distance of 0.69 km (1.37 km) to the closest well. Ecologically, both exposed and reference wetlands consist of either marshes—dominated by various grasses and sedges including maidencane (*Panicum hemitomon*) and pickerelweed (*Pontederia cordata*)—or swamps—dominated by cypress (*Taxodium* spp.) or hardwood species such as water tupelo (*Nyssa aquatica*) or sweet gum (*Liquidambar styraciflua*; Haag and Lee, 2010). Both the exposed and reference wetlands are also surrounded by pine flatwood upland habitat, which consists of several *Pinus* species and an understory of saw palmetto (*Serenoa repens*) shrubs. Geomorphologically, the reference sites have a similar ($p > 0.05$) area but are significantly ($p < 0.01$) shallower in depth than the exposed wetlands (Fig. S2). Hydrogeologically, exposed and reference wetlands are located in regions where the intermediate clay confining unit (see section 2.1.1) is either absent or less than 30 m thick, allowing some flow between the surficial and Upper Floridan aquifers (Fig. 1A). The reference wetlands are atop a high point in the regional potentiometric surface, meaning that water in the surficial aquifer beneath the reference wetlands moves towards the exposed wetlands and the Gulf of Mexico (Fig. 1B).

Other regional studies have used wetlands in the Green Swamp Wilderness Preserve as reference sites claiming that they are unaffected by pumping (GPI Southeast, Inc, 2009; Haag and Pfeiffer, 2012; Powell et al., 2019). When the Southwest Florida Water Management District did a modeling study to determine areas of impact to the surficial aquifer

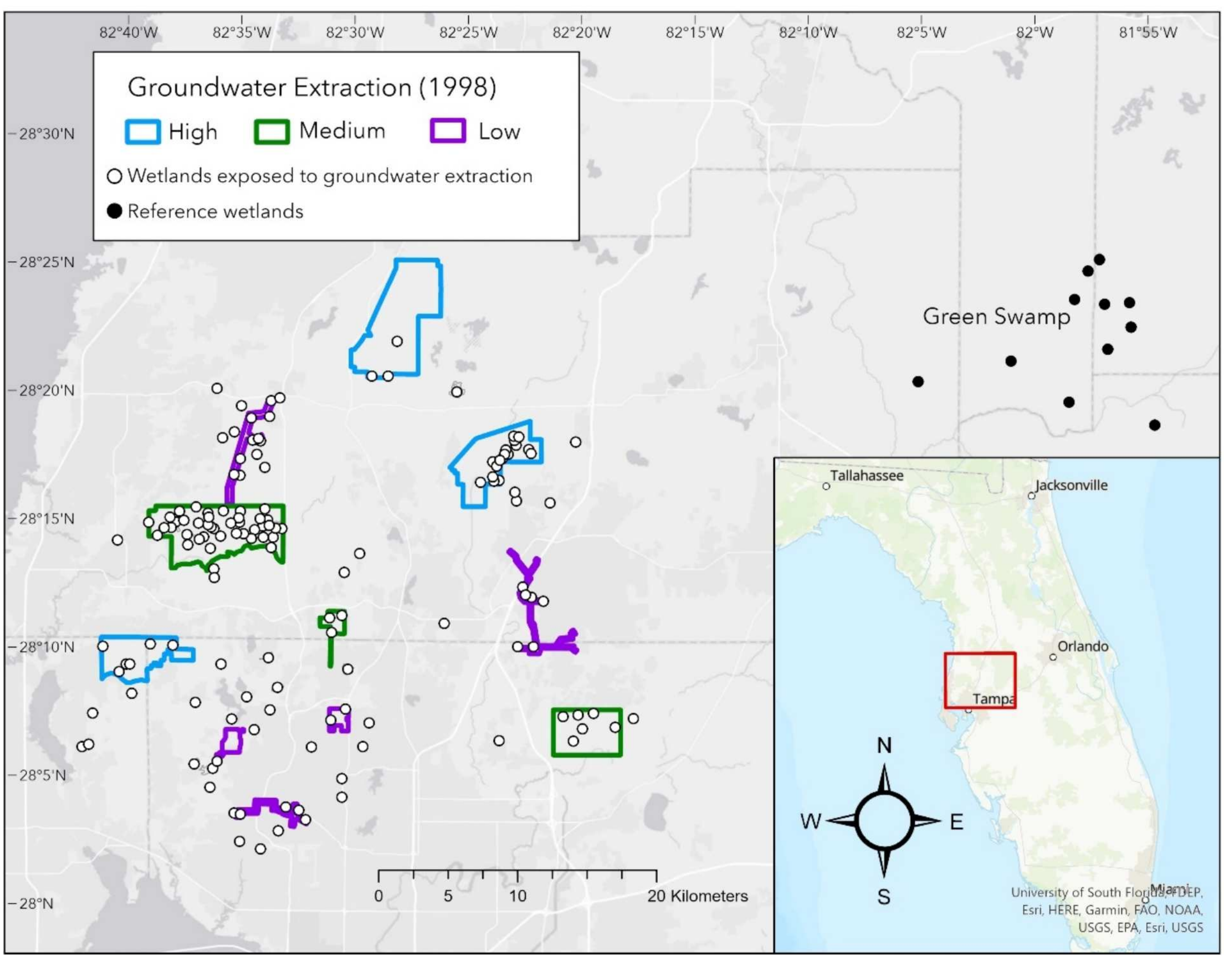

Fig. 3. Site map of 152 study wetlands in the northern Tampa Bay area separated into two groups: wetlands exposed to groundwater extraction (open circles) and reference wetlands (black circles). Wellfields are separated into three groups: high (blue), medium (green), and low (purple) based on historic (1998) groundwater extraction rates. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

preceding cutbacks in groundwater extraction, the areas of impact did not include the Green Swamp Wilderness Preserve but did include areas in and around the wellfields (Fig. S3). In sum, the reference sites exhibit many ecological and hydrogeological similarities to the exposed wetlands and fall outside the zone where groundwater extraction induces water-table drawdown; while they are not perfect control systems in a strict experimental sense, they nevertheless provide insight into wetland ecohydrological condition and variability arising from climate drivers without disturbance from groundwater extraction.

Both wetlands exposed to groundwater extraction and reference wetlands were selected based on the following criteria: 1) a score of 4 or 5 in hydrological data appropriateness as determined in GPI Southeast, Inc (2009), and 2) recorded historic normal pool and wetland bottom elevations. Hydrological data appropriateness was rated on a score from 1 to 5, where a score of 4 or 5 indicated a hydrograph with infrequent or no errors in sampling measurements (GPI Southeast, Inc, 2009). We additionally assessed wetland hydrographs to ensure no errors since the 2009 report (e.g., long periods of repeated values, values dropping far below the wetland bottom or above the historic normal pool elevations). Historic normal pool elevation is the historic high water-level elevation determined from various plant and soil indicators including the height of cypress tree buttresses, adventitious roots, the lowest limit of attachment of lichen or moss, and the location of the saw palmetto fringe and hydric soils; it is generally taken as the upper, outer edge of the wetland basin (Carr et al., 2006; Hull et al., 1989) with a 4 % exceedance frequency (Nilsson et al., 2013). All historic normal pool elevation measurements were determined through ground surveys, provided by the Southwest Florida Water Management District (SWFWMD), and last updated during the summer of 2019. Wetland bottom elevations were determined by topographic surveys and provided by GPI Southeast, Inc. (2009).

2.3. Compilation of dataset

Multiple indicators of wetland inundation were calculated using wetland water-level elevation (WWLE) data collected between 1991 and 2018, thereby encompassing times both before (1991–1997) and after (1998–2018) the shift from complete reliance on groundwater extraction to a focus on groundwater conservation. WWLE data were provided by TBW for the 152 study wetlands from composite time series of both staff gauges and monitoring wells. Staff gauges are graduated rulers installed at or near the deepest part of a wetland and provide WWLE data when surface water is present. Monitoring wells extend belowground and provide WWLE data when the water level is beneath the ground surface. Water-level observations are relative to the NAVD88 vertical datum and recorded at varying frequencies. During the 28-year study period (1991–2018), water-level observations were most commonly recorded 24 times per year (e.g., twice monthly) for each wetland, though it ranged from 12 to 365 sampling events per year.

From the WWLE data, three hydrological indicators were calculated: normal pool offset (NPO), wetland bottom offset (WBO), and monthly hydroperiod. Normal pool offset is the WWLE minus the historic normal pool elevation and reflects the proximity of the water surface to the top of the wetland basin (Fig. S4). Wetland bottom offset is the water-level elevation relative to the wetland bottom elevation. Mean monthly NPO and WBO were computed from the WWLE measurements for each wetland for each month from 1991 to 2018 and used in subsequent analyses. Monthly hydroperiod was calculated for each wetland as the number of times within each month in which a WWLE measurement was greater than the wetland bottom elevation (i.e., in which the wetland basin contained surface water) divided by the total measurements in each month. When there was only one water-level observation in a month, monthly hydroperiod was constrained to values of 0 or 1. Similarly, when there were two water-level observations in a month, monthly hydroperiod was constrained to values of 0, 0.5, or 1.

Monthly averages of daily groundwater extraction rates from 1931 to 2018 were provided by TBW. Wellfields were binned according to historic (1998) groundwater extraction rates. Historic extraction groups were determined from 1998 groundwater extraction rates because 1998 was the year TBW was created and is representative of the final high-volume pumping era prior to cutbacks (Fig. 2). The Jenks Natural Breaks algorithm was used to separate the wellfields into three bins (high, medium, and low historic extraction) as the algorithm maximizes the differences between each bin, while minimizing differences within each bin (Jenks, 1977). The low historic extraction bin includes five wellfields with an average groundwater extraction rate of approximately 29,900 m^3 day^{-1} $wellfield^{-1}$ in 1998. The medium historic extraction bin includes three wellfields with an average groundwater extraction rate of approximately 49,500 m^3 day^{-1} $wellfield^{-1}$ in 1998. Finally, the high historic extraction bin includes three wellfields with an average groundwater extraction rate of approximately 87,400 m^3 day^{-1} $wellfield^{-1}$ in 1998. Each of the $n = 141$ exposed wetlands were then classified as having been historically exposed to low, medium, or high rates of groundwater extraction based on the wellfield they were associated with as designated by the Southwest Florida Water Management District. Forty-nine wetlands were included in the low exposure group, 58 wetlands in the medium exposure group, and 34 wetlands in the high exposure group.

2.4. Data analysis

2.4.1. Directional trends in wetland inundation

All statistical analysis was completed in RStudio 4.1.3 using the R scientific programming language (R Core Team, 2023). Question 1 asked, how do trends in water level (increasing vs. not) during the period of groundwater conservation compare among wetlands historically exposed to different magnitudes of groundwater extraction? To answer this question, we calculated median and interquartile range values for the inundation metrics (NPO, WBO, and hydroperiod) among all wetlands within each historic exposure group (i.e., high, medium, and low) and in the reference group for four time periods (i.e., 1991–1997, 1998–2002, 2003–2008, and 2009–2018). The time periods were selected to correspond with sequential groundwater extraction cutbacks by TBW (Fig. 2).

Question 1 was further addressed by identifying 28-year (1991–2018) water-level trends in each individual wetland. The monthly WWLE data of each wetland were subjected to the non-parametric Mann-Kendall (MK) test (Kendall, 1975; Mann, 1945), which indicates whether the time series has an increasing or decreasing trend versus a null hypothesis of no directional change. Serial autocorrelation was assessed using the Durbin-Watson test (Durbin and Watson, 1950) and, when present in a time series, a modified MK test was then used on the autocorrelated time series. The modified MK test applies the MK test to residuals of a lag-one autoregression (AR(1)) model for each time series (Bayazit and Önöz, 2007; Hamed and Ramachandra Rao, 1998; Yue and Wang, 2004). If there was a significant directional trend in the time series (i.e., if the original or modified MK test was significant), the non-parametric Sen's slope estimator (Sen, 1968) was calculated, which indicates whether the trend in the time series is linear and if so, estimates the slope of that trend. We then noted the historical exposure group (historically exposed to high, medium, or low rates of groundwater extraction or to no extraction, i.e., reference group) that each wetland belonged to, so we could assess whether water level trends (increasing, steady-state, or declining) during the contemporary period of groundwater conservation corresponded with the magnitude of historical exposure to extraction.

2.4.2. Hydrological condition of wetlands with steady-state water levels

The trend analysis just described identified 57 exposed wetlands with no 28-year trend in water-level elevation (see section 3.1.2). Question 2 asked, are wetlands that exhibit steady-state water levels (i.e., those exhibiting no water-level increases over this time period) failing to recover, or have they not been impacted by historic groundwater extraction? To answer this question, we used paired Wilcoxon signed rank tests (Lam and Longnecker, 1983) to compare monthly mean NPO, WBO, or hydroperiod values for each steady-state exposed wetland with the lower 95 % confidence limit of the monthly mean NPO, WBO, or hydroperiod for the reference wetlands. Since multiple tests were employed in this analysis, a Bonferroni-corrected (Rice, 1989) alpha value of 0.00088 was used, as our original alpha value was 0.05 and there were 57 exposed wetlands exhibiting steady-state water levels, each separately subjected to these tests ($0.05/57 = 0.00088$). We decided to use a Bonferroni correction to determine our significance level to reduce the likelihood of a type I error (Cabin and Mitchell, 2000) given that we repeated the same statistical test multiple times using the same data (e.g., monthly mean NPO, WBO, or hydroperiod for reference wetlands).

A non-significant result is taken as evidence that exposed wetlands had NPO, WBO, or hydroperiod values similar to reference wetlands. We consider those wetlands to be unaffected or minimally affected by groundwater extraction. A significant result is taken as evidence that exposed steady-state wetlands had NPO, WBO, or hydroperiods lower than the lower confidence limit for reference wetlands (one-tailed test). We consider those wetlands to have been affected by historical pumping and not recently showing signs of recovery. We also noted the historical exposure group (historically exposed to high, medium, or low rates of groundwater extraction) that each of these 57 steady-state wetlands belonged to, so we could assess whether their hydrological condition (not affected or not recovering) corresponded with historical exposure to extraction.

2.4.3. Onset of increased inundation

The trend analysis described above (section 2.4.1) identified 81 exposed wetlands with increasing water levels (see section 3.1.2). Question 3 asked, does the onset of recovery vary among wetlands historically exposed (pre-conservation period) to different magnitudes of groundwater extraction? To answer this question, we fit generalized linear models with correlated error structures. Each hydrological indicator (NPO, WBO, or hydroperiod) for each historic exposure group (high, medium, or low) was modeled separately (i.e., nine analyses).

In each analysis, we first fit a linear model in which the response variable was monthly mean NPO, WBO, or hydroperiod, averaged across all wetlands in the exposure group under consideration. The predictor variables were (1) monthly mean NPO, WBO, or hydroperiod values among reference wetlands, (2) time period, a categorical variable (1991–1997, 1998–2002, 2003–2008, and 2009–2018), and (3) the interaction between 1 and 2. The grouped time periods were selected to correspond with the staged cutbacks in groundwater extraction rates (Fig. 2).

The correlated error structure was then determined by creating autocorrelation function (ACF) and partial autocorrelation function (PACF) plots of the residuals of each linear model. From these plots, we determined that autoregressive (AR) and moving average (MA) components of order 1 were indicated. Three generalized linear models were fit for each hydrological response variable using AR(1), MA(1), and ARMA(1,1) as error terms. The best-fit model was selected as whichever of those three had the lowest AIC value.

2.4.4. Synchrony of wetland water-level increases with groundwater extraction cutbacks

Question 4 asked, does the onset of recovery coincide with the timing of cutbacks in groundwater extraction? To answer this question, the Pettit change point test (Pettitt, 1979) was applied to the time series of groundwater extraction rates for each wellfield, and to the time series of water-level elevation for each wetland. This test detects and reports the first time point at which there is a significant change in the central tendency of the time series. Accordingly, we identified whether change points in wetland water-level elevations were synchronous with, or temporally lagged behind, change points in extraction rates of their corresponding wellfields.

2.4.5. Magnitude of wetland inundation changes with groundwater extraction cutbacks

Finally, question 5 asked, does the magnitude of change in wetland inundation correspond with the magnitude of change in groundwater extraction rates? To answer this question, Spearman correlation tests (Spearman, 1904) were used. To calculate the changes in each hydrological indicator and in groundwater extraction, the year 1998 was used as the baseline. The change in groundwater extraction rate for each wellfield j and year i ($i = 1991$–2018, excluding 1998) was calculated as the difference between annual mean daily extraction (m^3/day averaged over 365 days) from wellfield j in year i and annual mean daily extraction from wellfield j in 1998. Similarly, the change in a given hydrological indicator (e.g., WBO) for each wetland k and year i was calculated as the difference between the mean WBO of wetland k in year i and the mean WBO of wetland k in 1998, where mean WBO refers to the mean of all WBO observations in a year i for wetland k. To match the wellfield scale at which groundwater extraction data are available, the change (between year i and 1998) in each hydrological indicator was averaged among all wetlands in each wellfield, yielding $n = 27$ years x 11 wellfields = 297 values of change (from 1998) in extraction rate and in hydrological indicator.

3. Results

3.1. Directional trends in wetland inundation

3.1.1. Landscape-scale directional trends

Groundwater extraction rates decreased the most in the wellfields that historically had high extraction rates (Table 1; Fig. 4A). Median extraction rates in wellfields that historically had medium and low extraction rates increased briefly between 1998 and 2002 before declining, starting after 2003, likely to meet water demand following reductions in extraction in the historically high-extraction wellfields. After 1998, there were multiple drought periods including longer droughts from 2000 to 2001, and 2006–2008 as well as shorter droughts in 2009, 2011, 2012, 2013, and 2017 (National Oceanic and Atmospheric Administration, 2024). In 2000 and 2006, groundwater extraction rates increased in all wellfields.

Median wetland water-level indicators (i.e., NPO and WBO) and hydroperiod increased over the study period (1991–2018) across all three groups of historic exposure to groundwater extraction (Table 1; Fig. 4B–D; S5–7). Wetlands historically exposed to high and medium rates of groundwater extraction exhibited greater increases in NPO, WBO, and hydroperiod than did wetlands historically exposed to low rates of groundwater extraction. While median WBO and hydroperiod values in the exposed wetlands reached similar values as reference wetlands by 2018, median NPO values in the exposed wetlands remained lower than NPO values in reference wetlands, even after two decades of extraction cutbacks (Table 1; Fig. 4B–D). Wetlands in all three historic exposure groups had large enough increases in WBO to shift from primarily belowground water levels to aboveground (Table 1; Fig. 4C; S6).

3.1.2. Wetland-scale directional trends

Wetland water levels increased in 57 % (81/141) of exposed wetlands during the 1991–2018 study period (Table 2, S1; Figs. S8–9). Another 40 % (57/141) of exposed wetlands did not exhibit significant directional trends in water levels during the study period. Wetland water levels decreased in 2 % (3/141) of exposed wetlands during the study period. Eighty-two percent (9/11) of reference wetlands exhibited no directional trend in water level, while 18 % (2/11) exhibited increasing water levels over the study period. No reference wetlands had decreasing water levels during the study period. Exposed wetlands with

Table 1
Median (IQR) NPO, WBO, and hydroperiod in wetlands, and groundwater extraction rates in wellfields. Wetlands and wellfields are grouped according to historic (1998) exposure to groundwater extraction (columns), with values specified for each stage of groundwater extraction cutbacks (rows). The period 1991–1997 is prior to cutbacks.

Hydrological Variable	Cutbacks Stage	High historic exposure (n = 34 wetlands; n = 3 wellfields)	Medium historic exposure (n = 49 wetlands; n = 3 wellfields)	Low historic exposure (n = 58 wetlands; n = 5 wellfields)	Reference (n = 11 wetlands)
NPO* (m)	1991–1997	-1.37 (1.73)	-0.72 (1.27)	-0.67 (0.82)	-0.33 (0.59)
	1998–2002	-1.14 (1.65)	-1.03 (1.39)	-0.94 (1.05)	-0.49 (0.69)
	2003–2008	-0.56 (1.13)	-0.64 (1.17)	-0.67 (0.79)	-0.33 (0.45)
	2009–2018	-0.52 (0.87)	-0.38 (0.72)	-0.53 (0.68)	-0.26 (0.33)
WBO* (m)	1991–1997	-0.69 (1.63)	0.02 (0.82)	0.05 (0.66)	0.23 (0.56)
	1998–2002	-0.35 (1.68)	–0.18 (1.16)	-0.16 (0.93)	0.05 (0.65)
	2003–2008	0.26 (1.13)	0.11 (1.01)	0.12 (0.68)	0.16 (0.42)
	2009–2018	0.32 (0.96)	0.34 (0.65)	0.24 (0.56)	0.24 (0.33)
Hydroperiod (%)	1991–1997	17.0 (75.0)	50.0 (50.0)	58.0 (58.0)	67.0 (39.8)
	1998–2002	33.0 (75.0)	42.0 (59.0)	33.0 (41.0)	62.5 (48.0)
	2003–2008	83.0 (75.0)	58.0 (83.0)	67.0 (67.0)	75.0 (50.0)
	2009–2018	83.0 (58.0)	83.0 (50.0)	75.0 (50.0)	83.0 (42.0)
GWE* (m^3 day^{-1})	1991–1997	100,746 (16,703)	45,567 (9,577)	30,334 (23,636)	0
	1998–2002	87,771 (12,246)	47,046 (11,853)	35,826 (10,779)	0
	2003–2008	55,734 (18,224)	28,113 (21,160)	23,216 (17,539)	0
	2009–2018	50,450 (9,266)	17,274 (9,145)	11,954 (23,296)	0

*NPO=normal pool offset, WBO=wetland bottom offset, GWE=groundwater extraction

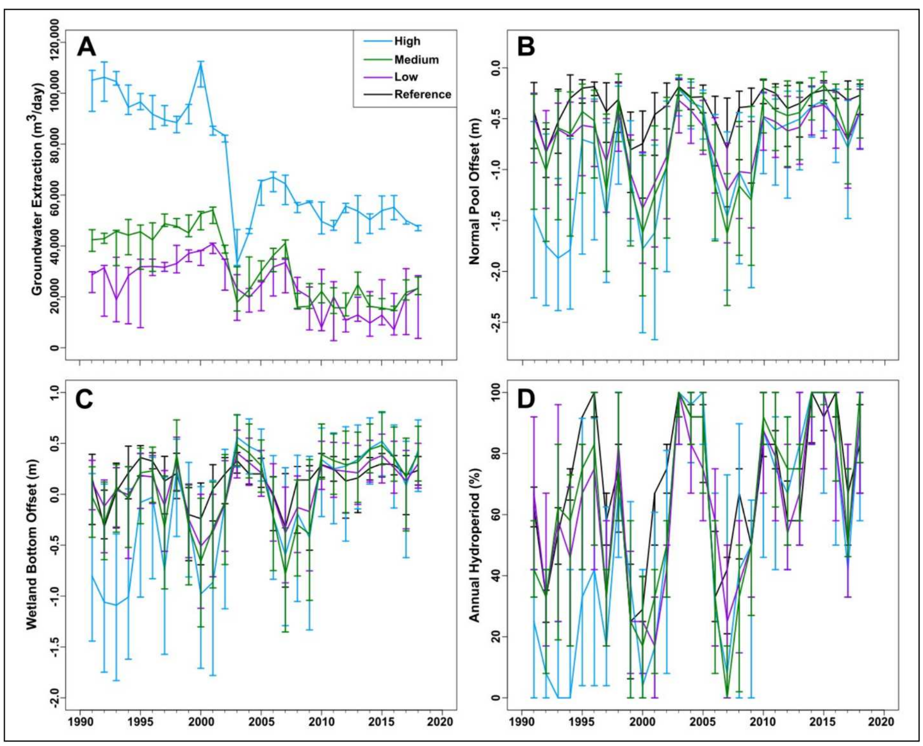

Fig. 4. Time series of median (±IQR) groundwater extraction rates on wellfields as well as normal pool offset, wetland bottom offset, and hydroperiod of wetlands, separated into groups of historic (1998) groundwater extraction.

Table 2
Count of wetlands by historic (1998) exposure to groundwater extraction and direction of trend in water-level elevation.

Water level trend	High historic exposure	Medium historic exposure	Low historic exposure	Reference	Total
Increasing	25	33	23	2	83
Steady State	9	24	24	9	66
Decreasing	0	1	2	0	3
Total	34	58	49	11	152

significantly increasing water levels had trend rates more than five times the rates exhibited by the two reference wetlands with increasing water levels. The median trend rate for the n = 81 exposed wetlands with increasing water levels was + 0.65 cm $year^{-1}$, while the trend rates for the two reference wetlands with increasing water levels were both + 0.01 cm $year^{-1}$. Wetland water levels were more likely to be increasing in wetlands historically exposed to high groundwater extraction rates (Table 2; Fig. S8).

3.2. Hydrological condition of wetlands with steady-state water levels

Here, we focused on the n = 57 exposed wetlands with steady-state water levels (those with no directional trend in water level elevation, from the analysis above), and examined whether inundation of these exposed, steady-state wetlands was similar to or less than inundation of the reference wetlands (Fig. S10). Most (>89 %) of the exposed wetlands with steady-state water levels exhibited a duration (hydroperiod) and depth (WBO) of inundation that was similar to those of reference wetlands (Table 3, S2). Conversely, a relatively large percentage (35 %) of exposed, steady-state wetlands had NPO values significantly lower than NPO values in reference wetlands.

3.3. Onset of increased inundation

For each exposure group, we examined whether mean wetland inundation was greater during any of the staged cutbacks in groundwater extraction (1998–2002, 2003–2008, and 2009–2018; Fig. 2B) than in the preceding pre-cutback period (1991–1997). We further investigated whether mean wetland inundation in each exposure group was temporally correlated with mean wetland inundation of reference wetlands. Wetlands historically exposed to high groundwater extraction rates exhibited significantly higher NPO values immediately following groundwater extraction cutbacks in 1998 and in every time period after that (2003–2008 and 2009–2018), compared to the baseline period of 1991–1997 (Table 4). Conversely, wetlands historically exposed to medium and low rates of groundwater extraction did not have significantly different NPO values in any post-cutback time period (1998–2018) compared to the baseline time period (1991–1997).

Wetlands historically exposed to high rates of groundwater extraction exhibited significantly higher WBO values immediately following groundwater extraction cutbacks in 1998 and in every time period after that (2003–2008 and 2009–2018), compared to the baseline period of 1991–1997 (Table 4). Wetlands historically exposed to medium and low levels of groundwater extraction exhibited higher WBO values only during the 2009–2018 time period compared to the baseline time period of 1991–1997.

Hydroperiods, NPO values, and WBO values in exposed wetlands, regardless of their past exposure to groundwater extraction, were significantly correlated with their counterparts in the reference wetlands, indicating synchronous temporal variation in inundation between exposed and reference wetlands (Table 4). These correlations were less strong (albeit still significant) for wetlands historically exposed to high rates of groundwater extraction, as denoted by model coefficients. Wetland hydroperiods in the high historic exposure group correlated with hydroperiods in the reference wetlands more strongly after extraction cutbacks (1998 and beyond) than before (1991–1997), as denoted by significant interactions between reference wetlands

Table 4
Coefficients from generalized linear regression models used to investigate whether average wetland inundation was greater during any of the staged cutbacks in groundwater extraction (1998–2002, 2003–2008, and 2009–2018) than in the preceding pre-cutback period (1991–1997). The correlated error structure used in each model is also indicated.

Hydrological Response Variable	Predictor Variables	Exposure Group		
		High(n = 25)	Medium (n = 33)	Low(n = 23)
Normal Pool Offset (NPO)	Reference wetlands NPO	0.520***	0.790***	0.650***
	1998–2002	0.380**	0.180	0.080
	2003–2008	0.650***	0.170	0.140
	2009–2018	0.640***	0.160	0.090
	Ref*1998–2002	0.290*	0.180	0.310**
	Ref*2003–2008	0.210	-0.100	0.110
	Ref*2009–2018	0.070	-0.300*	–0.050
	Correlated Error Structure	AR(1)	ARMA (1,1)	AR(1)
	AIC	-136.9	-168.1	-325.2
Wetland Bottom Offset (WBO)	Reference wetlands WBO	0.530***	0.800***	0.670***
	1998–2002	0.290*	0.110	–0.020
	2003–2008	0.640***	0.260	0.170
	2009–2018	0.720***	0.360*	0.230*
	Ref*1998–2002	0.280†	0.170	0.280**
	Ref*2003–2008	0.190	–0.120	0.080
	Ref*2009–2018	0.060	–0.310	–0.070
	Correlated Error Structure	AR(1)	ARMA (1,1)	AR(1)
	AIC	-135.4	-168.6	-328.2
Hydroperiod (H)	Reference wetlands hydroperiod	0.100*	0.340***	0.260***
	1998–2002	0.030	-0.080	-0.180†
	2003–2008	0.120	0.040	-0.120
	2009–2018	0.140	0.180†	-0.040
	Ref*1998–2002	0.190**	0.120	0.190
	Ref*2003–2008	0.200**	0.070	0.270
	Ref*2009–2018	0.200***	-0.00	0.210
	Correlated Error Structure	AR(1)	ARMA (1,1)	AR(1)
	AIC	-527.7	-430.1	-367.0

† $p < 0.1$, *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$; sample size (n) indicates number of wetlands.

Table 3
Wetlands with steady-state water levels over the study period (n = 57) categorized by historic exposure to groundwater extraction and inundation status relative to reference wetlands.

Hydrological Indicator	Inundation status relative to reference wetlands	Historic Exposure Group			
		High (n = 9 wetlands)	Medium (n = 24 wetlands	Low (n = 24 wetlands)	All (n = 57 wetlands)
Normal Pool Offset (NPO)	Similar to	6 (66.7 %)	18 (75.5 %)	13 (54.2 %)	37 (64.9 %)
	Less than	3 (33.3 %)	6 (25.0 %)	11 (45.8 %)	20 (35.1 %)
Wetland Bottom Offset (WBO)	Similar to	8 (88.8 %)	20 (83.3 %)	23 (95.8 %)	51 (89.5 %)
	Less than	1 (11.2 %)	4 (16.7 %)	1 (4.2 %)	6 (10.5 %)
Hydroperiod (H)	Similar to	8 (88.8 %)	21 (87.5 %)	23 (95.8 %)	52 (91.2 %)
	Less than	1 (11.2 %)	3 (12.5 %)	1 (4.2 %)	5 (8.8 %)

Note: Values in cells are number (and percent) of exposed, steady-state wetlands in each historic exposure group exhibiting the indicated inundation status relative to reference wetlands. Inundation status is deemed less than that of reference wetlands when the mean value of the hydrological indicator in an exposed wetland is below the lower 95% confidence limit of the hydrological indicator in reference wetlands.

hydroperiod and time periods.

3.4. Synchrony of wetland water-level increases with groundwater extraction cutbacks

Focusing on the n = 81 exposed wetlands with significantly increasing water levels, we investigated whether the onset of water-level increases was synchronous with cutbacks in groundwater extraction. All wellfields exhibited a significant change point in their extraction rates after 1998, when TBW was created and cutbacks began (Fig. 5; Table S1). When evaluating the correlation between change points in wetland water levels and change points in groundwater extraction rates in wellfields, four groups emerged. Group 1 contains most of the wetlands (64.0 %, 16/25) associated with historically high-extraction wellfields. These wetlands exhibited change points in water-level elevation within one year of groundwater extraction cutbacks, indicating a rapid water-level response to extraction cutbacks. Group 2 contains most wetlands (87.9 %, 29/33) associated with historically medium-extraction wellfields. These wetlands exhibited significant change points in water-level elevation within 2 years of extraction cutbacks in wellfields. Group 3 contains a mix of wetlands associated with both historically low-extraction wellfields (26.1 %, 6/23) and historically high-extraction wellfields (32.0 %, 8/25). These wetlands experienced significant change points in water levels over 10 years after significant cutbacks in groundwater extraction. Finally, group 4 contains wetlands primarily associated with historically low-extraction wellfields (21.7 %, 5/23) and these wetlands exhibited significant change points 3–5 years before groundwater extraction cutbacks in the wellfields where they were located.

When a time series of monthly precipitation was tested during the same period (1991–2018), no significant change points were found (Fig. 2). However, multiple drought periods occurred over the study period as enumerated above. Only 2 of the 11 reference wetlands exhibited significant change points in water-level elevation (Table S1). One exhibited a change point in 2002 and the other in 2009, two years that both coincide with the end of drought periods.

3.5. Magnitude of wetland inundation changes with groundwater extraction cutbacks

Finally, we examined whether the magnitude by which wetland inundation changed (in the n = 81 wetlands with rising water levels) correlated with the magnitude by which groundwater extraction rates changed. Spearman correlation tests demonstrated significant, negative relationships between change in each hydrological indicator variable and change in groundwater extraction rates (Fig. 6). That is, wetlands exhibited the greatest increases in inundation when and where extraction was cut back the most. Wetlands also exhibited lower inundation when and where extraction was greater. Change in NPO and change in WBO had similar Spearman correlation coefficients (r = -0.479 and r = -0.448, respectively), while change in hydroperiod had the lowest Spearman correlation coefficient (r = -0.376).

4. Discussion

Decades of overextraction of groundwater have altered surface water bodies including freshwater depressional wetlands. Management policies that prioritize groundwater conservation may allow wetland hydrological regimes, and subsequently wetland ecological and biogeochemical function, to recover. We evaluated the recovery patterns of wetland inundation across a wetlandscape over 28 years of shifting water conservation policies that included staged cutbacks in groundwater extraction. We found that inundation increased in many wetlands (57 %, 81/141) that had historically been exposed to groundwater extraction (Table 2). These increases in inundation were generally the greatest a) in wetlands historically exposed to the highest extraction rates (Table 4); and b) when and where groundwater extraction cutbacks were most extreme (Fig. 6). For most recovering wetlands, increased inundation began immediately upon, or within two years of, initiating extraction cutbacks (Fig. 5, groups 1–2), although some recovering wetlands exhibited longer lags (Fig. 5, group 3). These findings suggest successful and quick passive restoration via groundwater conservation. A subset of wetlands (14 %, 20/141) exhibited increases in water depths (greater WBO) without getting closer to filling their basins (no change in NPO; Table 3), possibly indicating subsidence and the need for active

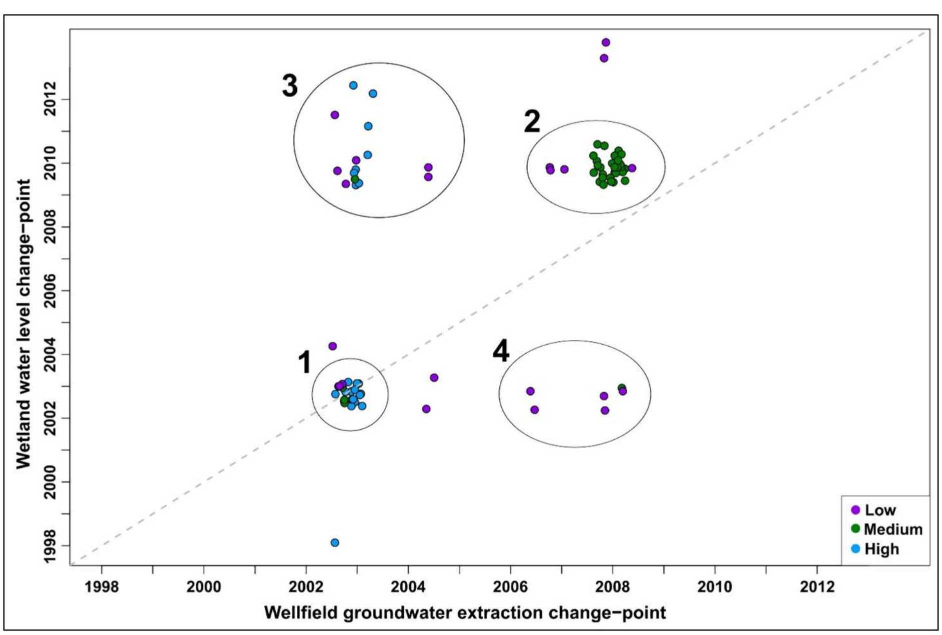

Fig. 5. Relationship between change point in wetland water level and change point in wellfield groundwater extraction rate. Points are colored according to exposure to historic (1998) rates of groundwater extraction. Groups (1–4) indicate the diversity among wetlands across the wetlandscape in how rapidly they responded to cutbacks in groundwater extraction. The grey dashed line denotes a 1-to-1 relationship, indicating an immediate increase in wetland water levels upon a decrease in groundwater extraction. Points were jittered to increase visibility.

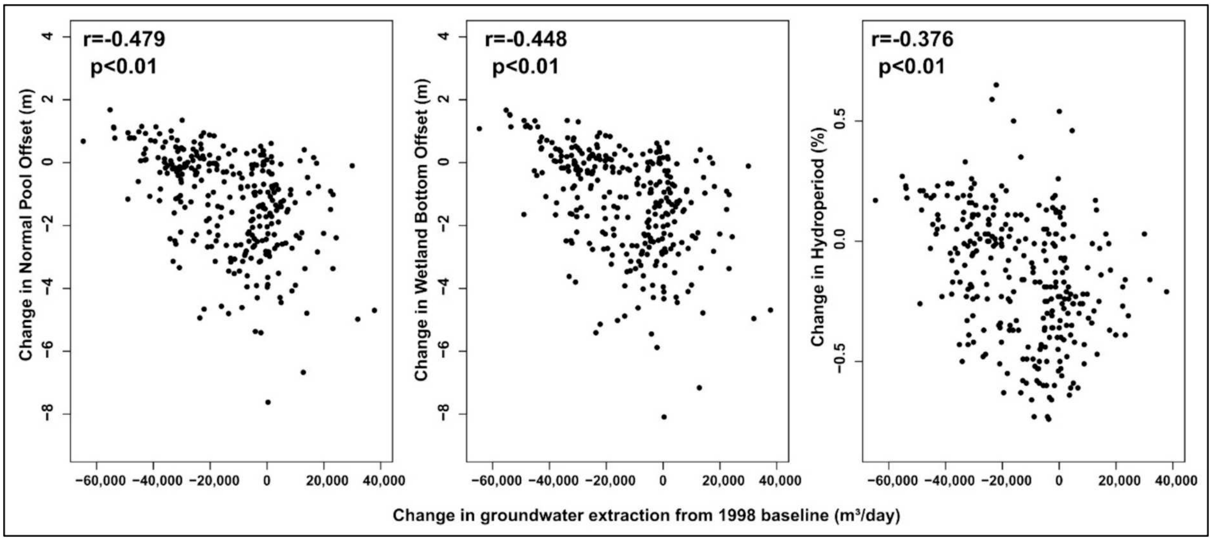

Fig. 6. Correlations between change in wetland hydrological indicator (normal pool offset, wetland bottom offset, and hydroperiod) and change in groundwater extraction rate.

restoration approaches beyond cutbacks in extraction. Another subset of wetlands (26 %, 37/141) exhibited steady-state water levels (Table 2) and were as inundated as reference wetlands (Table 3), suggesting their inundation had not been notably affected by past groundwater extraction. While some wetlands were not detectably affected by past groundwater extraction, many were, and most (albeit not all) of those now appear to be recovering. Targeted reductions in areas where historic groundwater extraction had been the greatest has particularly improved inundation of surrounding wetlands.

4.1. Groundwater management successes

Following cutbacks in groundwater extraction, wetland water levels increased in 57 % (81/141) of wetlands exposed to groundwater extraction, and 78 % (81/104) of impaired wetlands (i.e., wetlands other than those with steady-state water levels similar to the reference wetlands). All significant increases in water levels occurred after 1998 when staged groundwater extraction cutbacks began. The timing and magnitude of change in wetland inundation corresponded with the timing and magnitude of cutbacks in groundwater extraction, further supporting the hypothesized link between groundwater conservation and wetland hydrological recovery. Groundwater extraction from wellfields with the highest historic extraction rates was cut back the earliest and by the greatest magnitude. Water levels in wetlands associated with those wellfields generally responded immediately and by the greatest magnitude. Conversely, wellfields with historically low groundwater extraction rates had greater variability in pumping volumes after cutbacks began, with some wellfields increasing pumping rates to compensate for rapid declines in water supply from the historically high-extraction wellfields. This variability led to differences in the timing of when wetlands associated with those lower-extraction wellfields began to experience increases in inundation, with some wetlands experiencing change right away (Fig. 5, groups 1–2) and others experiencing hydrological change over 10 years later (Fig. 5, group 3).

These findings indicate the success of water management policies in the region including diversifying the water supply to include surface water sources and a desalination plant as well as consolidating the management of the many wellfields in the northern Tampa Bay region under one agency, TBW. Prior to consolidation, communities with high water demand relied on select wellfields, often in other political jurisdictions, impacting nearby wetlands and causing disputes among residents (Glennon, 2002; Rand, 2003). After consolidation, TBW developed an Integrated Hydrologic Model (Asefa et al., 2014; Geurink et al., 2007) to determine optimal groundwater extraction rates at these wellfields based on water demand and environmental considerations, such as nearby wetland condition. TBW can now balance high demand for water across all wellfields as well as surface water sources and the desalination plant. Without this consolidation, and the dispersed water sourcing it affords, the most heavily pumped wellfields might still be serving communities with high demand for water and the level of recovery seen in this study would not have been possible.

4.2. Groundwater management opportunities

A subset of wetlands did not have increasing water levels over the study period and instead may not be recovering hydrologically due to subsidence of the wetland basin. Among the 57 exposed wetlands with steady-state water levels, 35 % (20/57) exhibited lower NPO values than observed in reference wetlands, while only 10 % (6/57) of them exhibited lower WBO values. That is, some exposed wetlands with steady-state water levels exhibit apparently unimpaired surface water depth yet greater deviation of the water level below the top of the wetland basin. These wetlands may be experiencing soil subsidence, compaction, or possibly the development of sinkholes in the underlying karst. Soil subsidence and compaction typically occur in drained wetlands where soils are oxidized, allowing organic matter to be fully decomposed (Ewing and Vepraskas, 2006). Without the low-density organic matter, soils become more compact and soil bulk density increases. Groundwater extraction can further cause sinkholes to develop in wetlands by draining cavities in carbonate deposits that groundwater usually fills. Overlying soil or clay can then produce enough pressure for the carbonate deposits to collapse (Tihansky, 1999). Even with cutbacks in groundwater extraction, replenished groundwater may not be able to fill in sinkholes, preventing wetlands from holding surface water.

Soil subsidence is theoretically reversible as decomposition will slow and organic matter will be replenished once wetland hydroperiods increase again. More time may be needed before this occurs in these wetlands. Even 20 to 30 years after initiating restoration, wetland soil carbon stocks may still be lower than reference conditions (Moreno-Mateos et al., 2012). As the present study only reflects 20 years of reduced groundwater extraction, soil organic matter may be slowly reaccumulating and thus not require active interventions. Alternatively, if soil organic matter does not increase over time, more active recovery approaches may be needed. Recovering wetlands can develop different plant communities than reference wetlands given differences in seed banks and invasion of non-native species (Beas et al., 2013; Yepsen et al., 2014), which can affect biogeochemical processes including soil organic matter storage (Bernal and Mitsch, 2012). Active replanting of native species to complement cutbacks in groundwater extraction may help some wetlands recover. Restoring some wetlands may be neither feasible nor a desired management goal where subsidence and water-level decline are resulting from the natural hydrological variation among wetlands in this wetlandscape or from the natural process of sinkhole development that is common throughout this karst landscape (Tihansky, 1999).

4.3. Ecological considerations for individual wetlands and the wetlandscape

Changes in the timing and magnitude of wetland inundation, such as from changes in groundwater extraction, can influence wetland ecosystem function. Increased inundation is associated with decreasing soil bulk density, increasing soil organic matter content (Bartholomew et al., 2019; Lewis and Feit, 2015), increasing soil carbon storage (Lewis 2016), and increasing wetland plant (Goodwillie et al., 2020) and amphibian biodiversity (Pechmann et al., 1989). Pechman et al. (1989) found exponential growth in amphibian populations once wetlands were inundated for more than 100–150 days each year. In this study, many wetlands in the historically high-exposure group were inundated less frequently than this prior to 2003, when groundwater extraction was cut back by over 36 %, suggesting amphibian diversity and abundance would have likely been compromised. The rising water levels we observed in most wetlands also signal a fundamental shift in wetland water exchanges with the surficial aquifer. As water levels rise, wetlands in our study region shift from recharging the surficial aquifer to receiving discharge from the surficial aquifer, owing to the high sensitivity of the water table beneath uplands surrounding wetlands (Nilsson et al. 2013). Whether this shift in water exchange makes these wetlands more minerotrophic warrants further investigation.

The rising water levels observed in most wetlands we studied may also increase the incidence of spillover that connects and merges wetland basins via surface water (Bertassello et al., 2020; Leibowitz et al., 2016). As water did not rise uniformly, new patterns of inundation variability among wetlands may emerge. Such changes in the configuration and connectivity of wetland patches across the wetlandscape would affect the dynamics and evolution of metapopulations, the delivery of ecosystem services, and *meta*-ecosystem properties (Cohen et al., 2016). A more well-connected network of larger, longer-lived patches of suitable habitat may increase dispersal of wetland species such as birds (Naugle et al., 1999) and amphibians (Zamberletti et al., 2018). Such an outcome would influence metapopulation genetics (Watts et al., 2015)—potentially genetically homogenizing a metapopulation via increased gene flow (Unglaub et al., 2021), facilitating evolutionary rescue of local populations in deteriorating wetlands (Bell and Gonzalez, 2011), and/or selecting for a dispersal phenotype (Olivieri et al., 1995)—and influence metapopulation stability (Bertassello et al., 2022; Wang et al., 2015). Movement of water through a wetlandscape supports biodiversity and the retention of nutrients (Åhlén et al., 2020), with sufficiently high movement of water delivering material from wetlands to downstream areas (Cohen et al., 2016). Any increased movement of water resulting from the higher water levels we observed may accordingly mediate the provision of these and other wetlandscape services. Increased connection of surface water across wetland basins for a sufficiently long time may also promote the convergence of dissolved ion concentrations and macroinvertebrate assemblages (Leibowitz et al., 2016), thereby reducing ecological and biogeochemical heterogeneity across a wetlandscape.

4.4. Factors beyond groundwater extraction influencing wetland inundation

A small number of wetlands (6 %, 9/141) had water-level increases that preceded cutbacks in groundwater extraction (Fig. 5, group 4). Most (78 %, 7/9) of those wetlands were historically exposed to low volumes of groundwater extraction. This finding, coupled with the large number of low-exposure wetlands exhibiting steady-state water levels (Table 2), bolsters previous findings that when groundwater extraction is sufficiently low, it no longer imprints a signal on wetland water-level variability and instead climate factors are more significant predictors of variability in wetland inundation (Balerna et al. 2023). For example, during years with drought conditions, such as in 2000 and again from 2006 to 2008, groundwater extraction rates increased to meet demands (Fig. 2B, 4A). We found that inundation accordingly declined during those years (Fig. 4B–D), indicating that groundwater conservation and correspondingly favorable wetland responses remain subject to climate drivers. Climate change is expected to increase variability in precipitation, characterized by long drought periods (Mahjabin and Abdul-Aziz, 2020). Increases in groundwater extraction during those projected periods of drought will further exacerbate declines in wetland inundation. This synergy between drought and groundwater extraction highlights the importance of both diversifying water supply sources as well as encouraging water conservation approaches to reduce demands during droughts thereby better protecting freshwater resources.

Additional reasons for water-level variation among these wetlands may include differences in land development, topography, or underlying hydrogeology that likely arise from their wide dispersion across the study region (Fig. S11). For example, some wetlands may receive water inputs from connections with surrounding water bodies such as lakes and streams or from land development that reroutes surface-water runoff into wetlands thereby offsetting any losses from groundwater extraction (Balerna et al., 2023; McCauley et al., 2015). Variably present clay confining units common in this region can further disrupt interactions between wetlands and underlying aquifers even during periods of high groundwater extraction (Fig. 1A; Metz, 2011; Tihansky and Knochenmus, 2001). In places with lower connectivity between the surficial aquifer and underlying aquifers, groundwater extraction may not impact wetland inundation as directly as it does in the Tampa Bay region and fewer wetlands may respond to changes in groundwater management policies. Some of our results may also be influenced by our selection of reference wetlands, which sit in a slightly different geographic and hydrogeologic context (Fig. 1) and tend to be shallower than the exposed wetlands (Fig. S2). Studying additional reference sites in this region may provide more context for water-level variation beyond groundwater extraction. Overall, the diversity of wetland responses to changes in groundwater levels, and difficulty in predicting those responses, adds to growing evidence of the importance of preserving individual wetlands across a wetlandscape (Balerna et al., 2023; Bertassello et al., 2020; Cohen et al., 2016; Rains et al., 2016) to ultimately protect a diversity of hydrological regimes and thus biological and biogeochemical function.

5. Conclusions

Trends across the United States indicate that depressional wetlands have been declining in quantity and quality, partially due to groundwater extraction. Reversing this trend requires careful consideration of how to meet human demands for water without compromising freshwater resources. This study shows two management approaches that communities can consider to help reverse aquifer depletion and rapidly restore water levels in wetland landscapes. The first is diversifying water supply beyond groundwater to include surface water sources, desalination, or re-use. The second is taking a landscape-scale approach to dynamically shift locations of extraction across a landscape, which can prevent small, specific locations from experiencing the worst effects of intense groundwater extraction. In coming decades, it is hoped that restoration of hydrology will influence wetland biodiversity and biogeochemical function, including how well these wetlands store carbon to mitigate against climate change, enhance water-quality, and provide habitat for numerous species. Not every wetland in this study shows increasing inundation with groundwater extraction cutbacks alone, however. Active recovery approaches in those wetlands may thus be needed to better safeguard a diversity of ecological and biogeochemical wetland function across the landscape.

Funding

This material is based upon work supported by the National Science Foundation under Grant No. 1930451. JAB was supported by the Fred L. and Helen M. Tharp Scholarship and Presidential Doctoral Fellowship distributed by the University of South Florida.

CRediT authorship contribution statement

Jessica A. Balerna: Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis, Data curation, Conceptualization. **Andrew M. Kramer:** Writing – review & editing, Methodology, Conceptualization. **Shawn M. Landry:** Writing – review & editing, Methodology, Conceptualization. **Mark C. Rains:** Writing – review & editing, Methodology, Conceptualization. **David B. Lewis:** Writing – review & editing, Supervision, Methodology, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgments

We thank Chris Shea and Erin Hayes from Tampa Bay Water as well as Mike Hancock, Mark Hurst, TJ Venning, and Donna Campbell from the Southwest Florida Water Management District for providing data necessary for completion of this project. We additionally thank Dr. Rebecca Zarger and Dr. Luanna Prevost for additional help with project conception. Finally, we thank Cassandra Campbell, Carley DeFillips, and Sarita Emmanuel for their suggestions to improve the manuscript.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jhydrol.2024.132052.

References

Adler, R.W., 2015. US Environmental Protection Agency's new Waters of the United States Rule: connecting law and science. Freshwater Science 34, 1595–1600. https://doi.org/10.1086/684002.

Agreement, I., 1998. Amended and restated interlocal agreement. State of Florida legal document, West Coast Regional Water Supply Authority https://www.tampabaywater.org/wp-content/uploads/Amended-and-Restated-Interlocal-Agreement-050198.pdf (accessed 7.14.24.

Åhlén, I., Hambäck, P., Thorslund, J., Frampton, A., Destouni, G., Jarsjö, J., 2020. Wetlandscape size thresholds for ecosystem service delivery: Evidence from the Norrström drainage basin. Sweden. Science of the Total Environment 704, 135452. https://doi.org/10.1016/j.scitotenv.2019.135452.

Alley, W.M., 2002. Flow and Storage in Groundwater Systems. Science 296, 1985–1990. https://doi.org/10.1126/science.1067123.

Arthur, J.D., Fischler, C., Kromhout, C., Clayton, J.M., Kelley, G.M., Lee, R., Li, L., O'Sullivan, M., Green, R.C., Werner, C.L., 2008. Hydrogeologic Framework of the Southwest Florida Water Management District, Florida Geological Survey Bulletin 68. Florida Geological Survey and the Southwest Florida Water Management District.

Asefa, T., Adams, A., Kajtezovic-Blankenship, I., 2014. A tale of integrated regional water supply planning: Meshing socio-economic, policy, governance, and sustainability desires together. J. Hydrol. 519, 2632–2641. https://doi.org/10.1016/j.jhydrol.2014.05.047.

Balerna, J.A., Kramer, A.M., Landry, S.M., Rains, M.C., Lewis, D.B., 2023. Synergistic effects of precipitation and groundwater extraction on freshwater wetland inundation. J. Environ. Manage. 337, 117690. https://doi.org/10.1016/j.jenvman.2023.117690.

Bartholomew, M.K., Anderson, C.J., Berkowitz, J., 2019. Soil Conditions Following Hydrologic Restoration in Cypress Dome Wetlands. Wetlands 39, 185–196. https://doi.org/10.1007/s13157-018-1061-2.

Bayazit, M., Önöz, B., 2007. To prewhiten or not to prewhiten in trend analysis? Hydrol. Sci. J. 52, 611–624. https://doi.org/10.1623/hysj.52.4.611.

Beas, B.J., Smith, L.M., LaGrange, T.G., Stutheit, R., 2013. Effects of sediment removal on vegetation communities in Rainwater Basin playa wetlands. J. Environ. Manage. 128, 371–379. https://doi.org/10.1016/j.jenvman.2013.04.063.

Bell, G., Gonzalez, A., 2011. Adaptation and Evolutionary Rescue in Metapopulations Experiencing Environmental Deterioration. Science 332, 1327–1330. https://doi.org/10.1126/science.1203105.

Bellino, J.C., 2011. Digital Surfaces and Hydrogeologic Data for the Floridan Aquifer System in Florida and in Parts of. and South Carolina, Georgia, Alabama.

Bernal, B., Mitsch, W.J., 2012. Comparing carbon sequestration in temperate freshwater wetland communities. Glob Change Biol 18, 1636–1647. https://doi.org/10.1111/j.1365-2486.2011.02619.x.

Bertassello, L.E., Rao, P.S.C., Jawitz, J.W., Aubeneau, A.F., Botter, G., 2020. Wetlandscape hydrologic dynamics driven by shallow groundwater and landscape topography. Hydrol. Process. 34, 1460–1474. https://doi.org/10.1002/hyp.13661.

Bertassello, L.E., Jawitz, J.W., Bertuzzo, E., Botter, G., Rinaldo, A., Aubeneau, A.F., Hoverman, J.T., Rao, P.S.C., 2022. Persistence of amphibian metapopulation occupancy in dynamic wetlandscapes. Landsc Ecol 37, 695–711. https://doi.org/10.1007/s10980-022-01400-4.

Bierkens, M.F.P., Wada, Y., 2019. Non-renewable groundwater use and groundwater depletion: a review. Environ. Res. Lett. 14, 063002. https://doi.org/10.1088/1748-9326/ab1a5f.

Cabin, R.J., Mitchell, R.J., 2000. To Bonferroni or Not to Bonferroni: When and How Are the Questions. Bull. Ecol. Soc. Am. 81, 246–248.

Carr, D.W., Leeper, D.A., Rochow, T.F., 2006. Comparison of six biologic indicators of hydrology and the landward extent of hydric soils in west-central Florida, USA cypress domes. Wetlands 26, 1012–1019. https://doi.org/10.1672/0277-5212(2006)26[1012:COSBIO]2.0.CO;2.

Castaño, S., de la Losa, A., Martínez-Santos, P., Mediavilla, R., Santisteban, J.I., 2018. Long-term effects of aquifer overdraft and recovery on groundwater quality in a Ramsar wetland: Las Tablas de Daimiel National Park, Spain. Hydrol. Process. 32, 2863–2873. https://doi.org/10.1002/hyp.13225.

Cohen, M.J., Creed, I.F., Alexander, L., Basu, N.B., Calhoun, A.J.K., Craft, C., D'Amico, E., DeKeyser, E., Fowler, L., Golden, H.E., Jawitz, J.W., Kalla, P., Kirkman, L.K., Lane, C.R., Lang, M., Leibowitz, S.G., Lewis, D.B., Marton, J., McLaughlin, D.L., Mushet, D.M., Raanan-Kiperwas, H., Rains, M.C., Smith, L., Walls, S.C., 2016. Do geographically isolated wetlands influence landscape functions? Proc. Natl. Acad. Sci. u.s.a. 113, 1978–1986. https://doi.org/10.1073/pnas.1512650113.

Creed, I.F., Lane, C.R., Serran, J.N., Alexander, L.C., Basu, N.B., Calhoun, A.J.K., Christensen, J.R., Cohen, M.J., Craft, C., D'Amico, E., DeKeyser, E., Fowler, L., Golden, H.E., Jawitz, J.W., Kalla, P., Kirkman, L.K., Lang, M., Leibowitz, S.G., Lewis, D.B., Marton, J., McLaughlin, D.L., Raanan-Kiperwas, H., Rains, M.C., Rains, K.C., Smith, L., 2017. Enhancing protection for vulnerable waters. Nature Geosci 10, 809–815. https://doi.org/10.1038/ngeo3041.

Dahl, T.E., 1990. Wetlands losses in the United States 1780's to 1980's. Fish & Wildlife Service, U.S.

De Steven, D., Sharitz, R.R., Barton, C.D., 2010. Ecological Outcomes and Evaluation of Success in Passively Restored Southeastern Depressional Wetlands. Wetlands 30, 1129–1140. https://doi.org/10.1007/s13157-010-0100-4.

Durbin, J., Watson, G.S., 1950. Testing for serial correlation in least squares regression I. Biometrika 37, 409–428.

Evenson, G.R., Golden, H.E., Lane, C.R., McLaughlin, D.L., D'Amico, E., 2018. Depressional wetlands affect watershed hydrological, biogeochemical, and ecological functions. Ecol Appl 28, 953–966. https://doi.org/10.1002/eap.1701.

Ewing, J.M., Vepraskas, M.J., 2006. Estimating primary and secondary subsidence in an organic soil 15, 20, and 30 years after drainage. Wetlands 26, 119–130. https://doi.org/10.1672/0277-5212(2006)26[119:EPASSI]2.0.CO;2.

Fernandez, D.P., 2013. From Litigation To Arbitration: A Case Study In Water Resources Conflict. JBCS 9, 235–242. https://doi.org/10.19030/jbcs.v9i3.7801.

Geurink, J.S., Adams, A., Ross, M.A., 2007. Water Management Advantages of Comprehensive Representation of Wetlands in an Integrated HSPF-Modflow Hydrologic Model, in: World Environmental and Water Resources Congress 2007. Presented at the World Environmental and Water Resources Congress 2007, American Society of Civil Engineers, Tampa, Florida, United States, pp. 1–22. https://doi.org/10.1061/40927(243)305.

Glennon, R., 2002. Tampa Bay's Avarice: Cypress Groves, Wetlands, Springs, and Lakes in Florida. In: Water Follies: Groundwater Pumping and the Fate of America's Fresh Waters. Island Press, Washington, D. C., pp. 71–86

Goodwillie, C., McCoy, M.W., Peralta, A.L., 2020. Long-term nutrient enrichment, mowing, and ditch drainage interact in the dynamics of a wetland plant community. Ecosphere 11. https://doi.org/10.1002/ecs2.3252.

GPI Southeast Inc, 2009. Consolidated Water Use Permit Annual Interpretive Report (Annual Interpretive Report). GPI Southeast Inc, Tampa, FL.

Haag, K., Pfeiffer, W., 2012. Flooded area and plant zonation in isolated wetlands in well fields in the Northern Tampa Bay Region, Florida, following reductions in groundwater-withdrawal rates (Scientific Investigations Report No. 2012–5039), Scientific Investigations Report. US Geological Survey.

Haag, K.H., Lee, T.M., 2010. Hydrology and ecology of freshwater wetlands in central Florida: a primer. U.S. Geological Survey circular. U.S, Geological Survey, Reston, VA.

Hamed, K.H., Ramachandra Rao, A., 1998. A modified Mann-Kendall trend test for autocorrelated data. J. Hydrol. 204, 182–196. https://doi.org/10.1016/S0022-1694(97)00125-X.

Hogg, W., Hayes, E., Keesecker, D., Kiehn, W., Shea, C., 2020. Tampa Bay Water Recovery Assessment: Final Report of Findings (Final Report). Tampa Bay Water, Clearwater, FL.

Holl, K.D., Aide, T.M., 2011. When and where to actively restore ecosystems? For. Ecol. Manage. 261, 1558–1563. https://doi.org/10.1016/j.foreco.2010.07.004.

Hull, H., Post, J., Lopez, M., Perry, R., 1989. Analysis of water level indicators in wetlands: Implications for the design of surface water management systems. In: Fisk, D.W. (Ed.), Wetlands: Concerns and Successes, Symposium Proceedings of the American Water Resources Association, pp. 195–204.

Jame, S.A., Bowling, L.C., 2020. Groundwater Doctrine and Water Withdrawals in the United States. Water Resour Manage 34, 4037–4052. https://doi.org/10.1007/s11269-020-02642-0.

Jenks, G., 1977. Optimal data classification for choropleth maps. University of Kansas, Department of Geography, Lawrence, Kansas, Occasional paper.

Kendall, M., 1975. Rank correlation measures, 4th ed. Charles Griffin, London.

Konikow, L.F., Kendy, E., 2005. Groundwater depletion: A global problem. Hydrogeol J 13, 317–320. https://doi.org/10.1007/s10040-004-0411-8.

Lam, F.C., Longnecker, M.T., 1983. A Modified Wilcoxon Rank Sum Test for Paired Data. Biometrika 70, 510–513. https://doi.org/10.1093/biomet/70.2.510.

Lee, T.M., Fouad, G.G., 2018. Changes in Wetland Groundwater Conditions in the Northern Tampa Bay Area from 1990 to 2015. Data products and technical report prepared for Tampa Bay Water, HSW Engineering Inc, Tampa, FL.

Leibowitz, S.G., Mushet, D.M., Newton, W.E., 2016. Intermittent Surface Water Connectivity: Fill and Spill Vs. Fill and Merge Dynamics. Wetlands 36, 323–342. https://doi.org/10.1007/s13157-016-0830-z.

Lewis, D.B., Feit, S.J., 2015. Connecting carbon and nitrogen storage in rural wetland soil to groundwater abstraction for urban water supply. Glob Change Biol 21, 1704–1714. https://doi.org/10.1111/gcb.12782.

Mahjabin, T., Abdul-Aziz, O.I., 2020. Trends in the Magnitude and Frequency of Extreme Rainfall Regimes in Florida. Water 12, 2582. https://doi.org/10.3390/w12092582.

Mann, H.B., 1945. Nonparametric Tests Against Trend. Econometrica 13, 245. https://doi.org/10.2307/1907187.

Marton, J.M., Creed, I.F., Lewis, D.B., Lane, C.R., Basu, N.B., Cohen, M.J., Craft, C.B., 2015. Geographically Isolated Wetlands are Important Biogeochemical Reactors on the Landscape. Bioscience 65, 408–418. https://doi.org/10.1093/biosci/biv009.

McCauley, L.A., Anteau, M.J., van der Burg, M.P., Wiltermuth, M.T., 2015. Land use and wetland drainage affect water levels and dynamics of remaining wetlands. Ecosphere 6, 22. https://doi.org/10.1890/ES14-00494.1.

McLaughlin, D.L., Cohen, M.J., 2013. Realizing ecosystem services: wetland hydrologic function along a gradient of ecosystem condition. Ecol. Appl. 23, 1619–1631. https://doi.org/10.1890/12-1489.1.

McLaughlin, D.L., Kaplan, D.A., Cohen, M.J., 2014. A significant nexus: Geographically isolated wetlands influence landscape hydrology. Water Resour. Res. 50, 7153–7166. https://doi.org/10.1002/2013WR015002.

Metz, P., 2011. Factors that Influence the Hydrologic Recovery of Wetlands in the Northern Tampa Bay Area, Florida (Scientific Investigations Report No. 5127), Scientific Investigations Report. U.S. Geological Survey.

Mihelcic, J.R., Rains, M., 2020. Where's the Science? Recent Changes to Clean Water Act Threaten Wetlands and Thousands of Miles of Our Nation's Rivers and Streams. Environ. Eng. Sci. 37, 173–177. https://doi.org/10.1089/ees.2020.0058.

Miller, J., 1997. The hydrogeology of Florida. In: Jones, D., Randazzo, R. (Eds.), The Geology of Florida. University Press of Florida, Gainesville, FL, pp. 69–88.

Moreno-Mateos, D., Power, M.E., Comín, F.A., Yockteng, R., 2012. Structural and Functional Loss in Restored Wetland Ecosystems. PLoS Biol 10, e1001247.

Naugle, D.E., Higgins, K.F., Nusser, S.M., Johnson, W.C., 1999. Scale-dependent habitat use in three species of prairie wetland birds. Landsc. Ecol. 14, 267–276.

Nilsson, K.A., Rains, M.C., Lewis, D.B., Trout, K.E., 2013. Hydrologic characterization of 56 geographically isolated wetlands in west-central Florida using a probabilistic method. Wetlands Ecol Manage 21, 1–14. https://doi.org/10.1007/s11273-012-9275-1.

Nowicki, R.S., Rains, M.C., LaRoche, J.J., Pasek, M.A., 2021. The Peculiar Hydrology of West-Central Florida's Sandhill Wetlands, Ponds, and Lakes—Part 1: Physical and Chemical Evidence of Connectivity to a Regional Water-Supply Aquifer. Wetlands 41, 113. https://doi.org/10.1007/s13157-021-01493-8.

Nowicki, R.S., Rains, M.C., LaRoche, J.J., Downs, C., Kruse, S.E., 2022. The Peculiar Hydrology of West-Central Florida's Sandhill Wetlands, Ponds, and Lakes –Part 2: Hydrogeologic Controls. Wetlands 42, 43. https://doi.org/10.1007/s13157-022-01560-8.

The National Oceanic and Atmospheric Administration (NOAA), National Integrated Drought Information System, 2024. Drought Conditions for Hillsborough County [WWW Document]. URL https://www.drought.gov/states/florida/county/hillsborough (accessed 4.5.24).

Olivieri, I., Michalakis, Y., Gouyon, P.-H., 1995. Metapopulation Genetics and the Evolution of Dispersal. Am. Nat. 146, 202–228. https://doi.org/10.1086/285795.

Pechmann, J.H.K., Scott, D.E., Whitfield Gibbons, J., Semlitsch, R.D., 1989. Influence of wetland hydroperiod on diversity and abundance of metamorphosing juvenile amphibians. Wetlands Ecol Manage 1. https://doi.org/10.1007/BF00177885.

Pettitt, A.N., 1979. A Non-Parametric Approach to the Change-Point Problem. Appl. Stat. 28, 126. https://doi.org/10.2307/2346729.

Powell, K.M., Wynn, J.G., Rains, M.C., Stewart, M.T., Emery, S., 2019. Soil indicators of hydrologic health and resilience in cypress domes of West-Central Florida. Ecol. Ind. 97, 269–279. https://doi.org/10.1016/j.ecolind.2018.10.008.

R Core Team, 2023. R: A language and environment for statistical computing. [WWW Document]. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rains, M.C., Fogg, G.E., Harter, T., Dahlgren, R.A., Williamson, R.J., 2006. The role of perched aquifers in hydrological connectivity and biogeochemical processes in vernal pool landscapes, Central Valley. California. Hydrol. Process. 20, 1157–1175. https://doi.org/10.1002/hyp.5937.

Rains, M.C., Landry, S., Rains, K.C., Seidel, V., Crisman, T.L., 2013. Using Net Wetland Loss, Current Wetland Condition, and Planned Future Watershed Condition for Wetland Conservation Planning and Prioritization, Tampa Bay Watershed, Florida. Wetlands 33, 949–963. https://doi.org/10.1007/s13157-013-0455-4.

Rains, M.C., Leibowitz, S.G., Cohen, M.J., Creed, I.F., Golden, H.E., Jawitz, J.W., Kalla, P., Lane, C.R., Lang, M.W., McLaughlin, D.L., 2016. Geographically isolated wetlands are part of the hydrological landscape. Hydrol. Process. 30, 153–160. https://doi.org/10.1002/hyp.10610.

Rand, H., 2003. Water Wars: A Story of People. Politics & Power, Xlibris Corp, Bloomington, Indiana.

Rice, W.R., 1989. ANALYZING TABLES OF STATISTICAL TESTS. Evolution 43, 223–225. https://doi.org/10.1111/j.1558-5646.1989.tb04220.x.

Sen, P.K., 1968. Estimates of the Regression Coefficient Based on Kendall's Tau. J. Am. Stat. Assoc. 63, 12.

Shahid, S., Hazarika, M.K., 2010. Groundwater Drought in the Northwestern Districts of Bangladesh. Water Resour Manage 24, 18. https://doi.org/10.1007/s11269-009-9534-y.

Spearman, C., 1904. The Proof and Measurement of Association between Two Things. AJP 15, 72–101.

Swfwmd, 2016. Potentiometric Surface Data 2011 [WWW Document]. accessed 4.12.24 Geospatial Open Data Portal. https://data-swfwmd.opendata.arcgis.com/datasets/84fd0c4d47c0456898d66d5d559caccf_34/explore?location=28.198144%2C-81.955300%2C7.47.

SWFWMD, 2022. Rainfall Summary Data by Region [WWW Document]. Southwest Florida Water Management District Data & Maps. URL https://www.swfwmd.state.fl.us/resources/data-maps/rainfall-summary-data-region (accessed 6.16.20).

Tihansky, A.B., 1999. Sinkholes, west-central Florida. In: Galloway, D.L., Jones, D.R., Ingebritsen, S.E. (Eds.), Land Subsidence in the United States. U.S, Geological Survey Circular, pp. 121–140.

Tihansky, A.B., Knochenmus, L.A., 2001. Karst Features and Hydrogeology in West-central Florida—A Field Perspective, in: Kuniansky, E.L. (Ed.), U.S. Geological Survey Karst Interest Group Proceedings, Water-Resources Investigations Report 01-4011. U.S. Geological Survey, Tampa, FL, pp. 198–211.

Unglaub, B., Cayuela, H., Schmidt, B.R., Preißler, K., Glos, J., Steinfartz, S., 2021. Context-dependent dispersal determines relatedness and genetic structure in a patchy amphibian population. Mol. Ecol. 30, 5009–5028. https://doi.org/10.1111/mec.16114.

Wang, S., Haegeman, B., Loreau, M., 2015. Dispersal and Metapopulation Stability. Peerj 3, e1295.

Tampa Bay Water, 2022. Current Drinking Water Sources [WWW Document]. Supply. URL https://www.tampabaywater.org/current-drinking-water-sources (accessed 6.20.22).

Watts, A.G., Schlichting, P.E., Billerman, S.M., Jesmer, B.R., Micheletti, S., Fortin, M.-J., Funk, W.C., Hapeman, P., Muths, E., Murphy, M.A., 2015. How spatio-temporal habitat connectivity affects amphibian genetic structure. Front. Genet. 6. https://doi.org/10.3389/fgene.2015.00275.

Yepsen, M., Baldwin, A.H., Whigham, D.F., McFarland, E., LaForgia, M., Lang, M., 2014. Agricultural wetland restorations on the USA Atlantic Coastal Plain achieve diverse native wetland plant communities but differ from natural wetlands. Agr Ecosyst Environ 197, 11–20. https://doi.org/10.1016/j.agee.2014.07.007.

Yue, S., Wang, C., 2004. The Mann-Kendall Test Modified by Effective Sample Size to Detect Trend in Serially Correlated Hydrological Series. Water Resour.Management 18, 201–218. https://doi.org/10.1023/B:WARM.0000043140.61082.60.

Zamberletti, P., Zaffaroni, M., Accatino, F., Creed, I.F., De Michele, C., 2018. Connectivity among wetlands matters for vulnerable amphibian populations in wetlandscapes. Ecol. Model. 384, 119–127. https://doi.org/10.1016/j.ecolmodel.2018.05.008.