Reorganizing the Waterscape: Asymmetric Loss of Wetlands and Gain of Artificial Water Features in a Mixed-use Watershed

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WETLANDS AND GLOBAL CHANGE

Reorganizing the Waterscape: Asymmetric Loss of Wetlands and Gain of Artificial Water Features in a Mixed-use Watershed

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Abstract
Between the 1780 and 1980s, more than half of the wetlands in the conterminous US were lost. As wetlands have been lost, numerous artificial water features (AWFs), such as stormwater retention ponds, golf course water features, and reservoirs, have been constructed. We contrasted the loss of wetland area and perimeter to the gain of AWF area and perimeter and further explored how this transformation has altered the spatial characteristics of the waterscape. We conducted this analysis in the Tampa Bay Watershed, a large coastal watershed that lost 33% of its wetland area between the 1950s-2007. Trends have been towards fewer, smaller wetlands and more, smaller AWFs. The loss of wetland area far exceeds the gain in AWF area, leading to an overall loss of 23% of the combined wetland and AWF area. However, the loss of wetland perimeter almost equals the gain in AWF perimeter, leading to an overall loss of just 2% of the combined wetland and AWF perimeter. The loss of wetlands and gain of AWFs have predominantly occurred in different geographic locations, with the loss of wetlands predominantly in the headwaters and the gain in AWFs predominantly adjacent to Tampa Bay. Wetlands became further apart, though generally retained their natural distribution, while AWFs became closer to one another and now mirror the more natural wetland distribution. Overall, the physical structure of the waterscape of today is different than in the past, which likely reflects a change in functions performed and related ecological services provided at local and landscape scales.

Keywords Tampa Bay Watershed · Land use and land cover change · Spatial analysis · GIS

Introduction

During colonial settlement, the conterminous United States had approximately 89 million ha of wetlands (Dahl 1990). Between the 1780 and 1980s, more than half of these wetlands were lost at the average rate of approximately 27 ha/hr (Dahl 1990). Florida was a case-in-point, losing nearly half of its wetlands the average rate of 6 ha/hr during this same time (Dahl 1990).

Trends have reversed in recent decades as protections for wetlands increased at federal and state levels (Downing et al. 2003; Creed et al. 2017). Between 1998 and 2009, wetland area in the conterminous United States remained approximately constant (Dahl 2006, 2011). However, wetlands continue to be lost in coastal watersheds on the Atlantic and Gulf of Mexico coasts, where wetland area continued to decline by an average of approximately 26,000 ha annually between 1998 and 2004 (Stedman and Dahl 2008). Wetlands also continue to be lost throughout Florida. However, the rates of loss have been declining, with approximately 29,150 ha/yr lost between the 1950-1970s (Hefner 1986), 9,600 ha/yr lost between the 1970-1980s (Frayer and Hefner 1991), and 2,030 ha/yr lost between the 1980-1990s (Dahl 2005). Cumulatively during this time, losses were especially acute in the Tampa Bay region of west-central Florida (Stedman and Dahl 2008), with one-third of all freshwater wetlands

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lost in the Tampa Bay Watershed between the 1950s-2007 (Rains et al. 2013).

The reasons for wetland loss nationwide are myriad, with losses generally attributed to urban and rural development, and agricultural and silvicultural operations (Dahl 2006, 2011). More granular detail is available for freshwater wetland loss in the Tampa Bay Watershed between the 1950s-2007, where 27% was lost to urban and rural development, 23% was lost to agricultural operations, 19% was lost to sand and phosphate mining, and 17% was lost to drying, the latter presumably due to ditching and draining and/or groundwater extraction (Rains et al. 2013). The recently reported wetland gains that have slowed or halted the rate of net wetland loss have been largely by the creation, enhancement, or restoration of wetlands through regulatory and nonregulatory programs and the creation of artificial water features (AWFs), such as stormwater retention ponds, golf course water features, and small reservoirs (Dahl 2006, 2011).

In some locations, wetland loss may have been offset by AWF gain from the strict standpoint of the total area of aquatic features in the landscape. However, wetlands and AWFs perform different functions at different rates (Rooney et al. 2015; Beckingham et al. 2019). Furthermore, there are likely differences in the location, size, shape, and distribution between wetlands lost and AWFs gained, so a change from wetlands to a mix of wetlands and AWFs likely changes the functions that emerge at scale (Cohen et al. 2016). Therefore, wetland loss is unlikely to be offset by AWF gain from the standpoint of the total functional capacity of aquatic features in the landscape (Rooney et al. 2015; Beckingham et al. 2019; Hess et al. 2022). To our knowledge, however, no study has directly quantified how the loss of wetlands and the gain in AWFs has altered the waterscape in terms of both total area and spatial characteristics (e.g., location, size, shape, and distribution) of the aquatic features in the landscape, especially where land use-land cover (LULC) is mixed. Van Meter and Basu (2015), Serran and Creed (2016), and Serran et al. (2018) estimated change in both wetland area and spatial characteristics, comparing current conditions from direct measurements to historical conditions from indirect analyses. McIntyre et al. (2018) directly measured change in wetland area and spatial characteristics, though they did so in a predominantly agricultural setting. Rains et al. (2013) directly measured change in wetland area in a mixed-use setting but did not specifically address spatial characteristics. And none of these or any other authors, to our knowledge, explicitly addressed AWF as a separate feature class.

Rains et al. (2013) previously conducted a wetland change analysis between the 1950s-2007 in the Tampa Bay Watershed. In the current study, we extend the results of Rains et al. (2013) by explicitly adding AWF as a separate feature class and by subsequently analyzing the changes in total area and the spatial characteristics of individual wetlands, individual AWFs, and combined wetlands and AWFs. Overall, we quantify and analyze how the net loss of wetlands and net gain of AWFs have reorganized the waterscape. We are motivated by the following four hypotheses: (1) The loss of wetland area and perimeter have not been offset by the gain of AWF area and perimeter; (2) The loss of wetland area and the gain of AWF area have occurred in different subregions of the watershed; (3) Mean areas and perimeters of both wetlands and AWFs have decreased; and (4) Wetlands are further apart and AWFs are closer together, altering the characteristic network structure.

Study Area

The Tampa Bay Watershed encompasses 5,908 km^2 in west-central Florida (USA) and drains to Tampa Bay on the Gulf of Mexico (Fig. 1). It includes numerous rivers and constructed drainageways, with the Hillsborough River, Tampa Bypass Canal, Alafia River, Little Manatee River, and Manatee River among the most prominent. LULC is mixed, with the most common LULC cover classes being urban (including mining) and agriculture, which comprise 43% and 22% of the watershed, respectively (Southwest Florida Water Management District 2008).

The climate is subtropical and humid (TAMPA WSCMO ARPT, FLORIDA 088788, 1981–2010). Mean annual temperature is 22.6 °C, ranging from a minimum monthly mean of 15.9 °C (January) to a maximum monthly mean of 28.1 °C (August). Mean annual precipitation is 1,203 mm, approximately 60% of which occurs during a 4-month wet season (June-September). The geology is typified by a thin cover of unconsolidated sediments underlain by a thick sequence of carbonate rocks. The unconsolidated sediments are comprised of interbedded fine and coarse clastic sediments (Sinclair 1974), often but not always underlain by a confining unit comprised of undifferentiated clay-rich sediments (Knochenmus 2006). The thick sequence of carbonate rocks comprises multiple layers of limestone and dolomite and forms the Upper Floridan aquifer (Miller 1997), the primary source of drinking water in the Tampa Bay Watershed (Tampa Bay Water 2022). Karst subsidence is a characteristic feature of the land surface, with the differential dissolution of the limestone surface creating hummocks and hollows, with numerous wetlands and waterbodies filling the hollows (Tihansky and Knochenmus 2001). Water tables are shallow, and groundwater in the surficial sediments and surface water in wetlands and waterbodies are commonly contiguous (Nowicki et al. 2021, 2022).

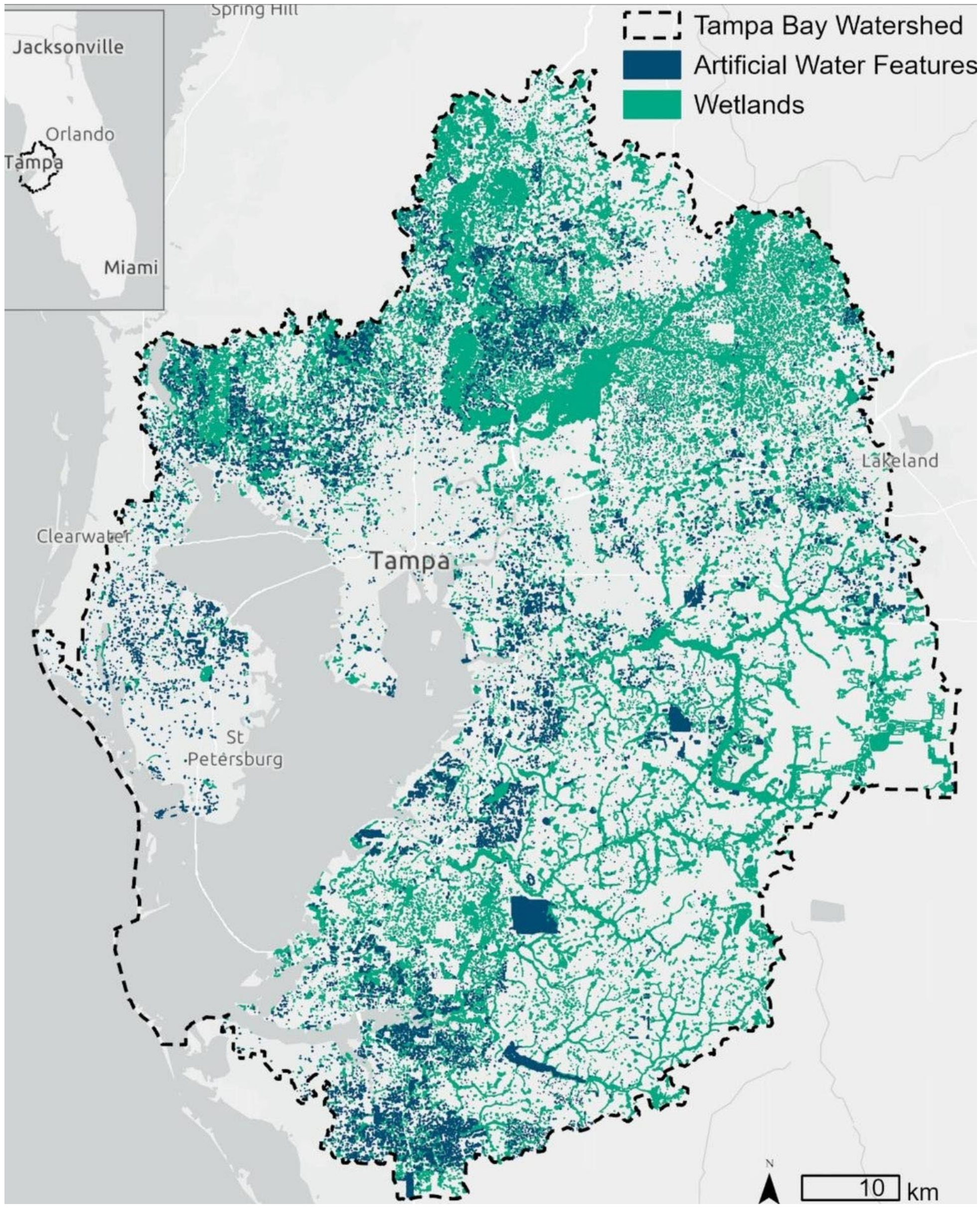

Fig. 1 Study area and distribution of wetlands and artificial water features in 2007

Florida has more wetlands in terms of total area and percentage of total land area than any of the other conterminous United States (Hefner and Brown 1984; Fretwell et al. 1996). This is typified by the Tampa Bay Watershed, where freshwater wetlands comprise 14% of the land surface, with riverine, lacustrine, flat, and depressional wetlands all being common (*sensu* Brinson 1993; Rains et al. 2013). Florida also has an abundance of AWFs. Again, this is typified by the Tampa Bay Watershed, where AWFs are common and include stormwater retention ponds, golf course water features, and reservoirs (Fig. 2).

Methods

Mapping and Classification

For this study, we set historical conditions as the 1950s (i.e., 1948–1958) and current conditions as 2007. This facilitated direct comparisons to a prior detailed wetland change analyses in the Tampa Bay Watershed, which was conducted over the same time interval (i.e., Rains et al. 2013). The 1950s was used as the historical condition because this both predated much of the development outside of the major metropolitan areas and was the period used until recently as the benchmark for setting targets for the restoration of estuarine habitats in Tampa Bay (Cicchetti and Greening 2011).

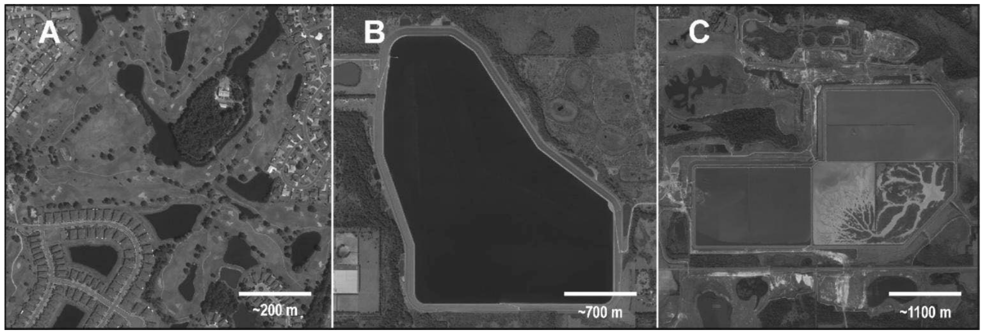

Fig. 2 Typical types of AWFs in the study area. **A** – stormwater retention ponds and golf course water features; **B** – reservoirs; and **C** – permanent mine holding ponds

The development of the 1950s and 2007 wetland geospatial datasets utilized in this study is described in Rains et al. (2013) and is consistent with the approach described below for the 1950s and 2007 AWF geospatial datasets. We utilized QGIS v2.18.20 (QGIS.org, Zürich, Switzerland) to create the 1950s and 2007 AWF geospatial datasets. To create the 2007 AWF geospatial dataset, we combined features from two other geospatial datasets. We first extracted all human-made water features from a publicly available LULC dataset depicting conditions in 2007 (Southwest Florida Water Management District 2008). These human-made water features included stormwater retention ponds, golf course water hazards, and reservoirs (Fig. 2). In this LULC dataset, the minimum mapping unit for these features is 0.8 ha except for those occurring in areas designated as "extractive" (mines) or "utilities" where they are commonly aggregated with adjacent features (Southwest Florida Water Management District 2009). Extractive lands are common, especially in the Central Florida Phosphate District located in the east-central portion of the Tampa Bay Watershed. The Mosaic Company—the most prominent phosphate mining company in the Tampa Bay Watershed—provided shapefiles representing all human-made water features on the mined and reclaimed landscapes in 2007. These human-made water features included stormwater retention ponds, golf course water features, reservoirs, and permanent mine holding ponds (Fig. 2), but did not include temporary mine holding ponds in the actively mined areas. We visually compared the human-made water features extracted from the two geospatial datasets against aerial imagery from 2007, adding missed human-made water features by heads-up digitizing. This comprised the 2007 AWF geospatial dataset. We then created the 1950s AWF geospatial dataset by modifying the 2007 AWF geospatial dataset while viewing aerial imagery from the 1950s, editing human-made water features as necessary by heads-up digitizing. All boundary modifications and new linework was digitized at a scale of 1:5000 using automated vertex generation every 20 m.

We analyzed wetlands and AWF in total (e.g., the total area of all wetlands) and as individuals (e.g., the mean area of the typical wetland). We followed Rains et al. (2013) and defined an individual wetland as any polygon (or "patch") with a unique hydrogeomorphic (*sensu* Brinson 1993) and vegetation structure class. Rains et al. (2013) identified three hydrogeomorphic classes in this study area: riverine, lacustrine, and slope-flat-depressional, combining slope, flat, and depressional classes into a single class because relief typically varies only slightly and typically below the minimum mapping unit of 0.8 ha for these features (Southwest Florida Water Management District 2009). Rains et al. (2013) further specified two vegetation structure classes: forested and non-forested. By this method, a single contiguous wetland environment could be separated into numerous smaller individual wetlands. For example, a single contiguous lacustrine-fringe wetland environment could be classified into multiple contiguous but individual wetlands (i.e., forested lacustrine, non-forested lacustrine), each with a minimum mapping unit of 0.8 ha (Southwest Florida Water Management District 2009). This did affect our results in terms of numbers, areas, and perimeter:area of the wetlands. We nevertheless did this for two reasons. First, this facilitated direct comparisons to the prior detailed wetland change analyses in the Tampa Bay Watershed (i.e., Rains et al. 2013). Second, there are many wetland environments in the Tampa Bay Watershed in which forested and non-forested riverine, lacustrine, and slope-flat-depressional wetland patches are contiguously connected. However, these individual wetland patches can be distinguished from one another by both hydrogeomorphic class (*sensu* Brinson 1993) and vegetation structure (e.g., forested, non-forested), which is at least in part a function of hydrologic characteristics (Nilsson et al. 2013, Balerna et al. 2023). By most conventional definitions, these individual wetland patches are themselves individual wetlands.

Change Analysis, Spatial Statistics, and Visualization

We utilized four software platforms to complete the analyses: QGIS geometry tools (QGIS.org, Zürich, Switzerland); MMQGIS, a QGIS plugin, for edge-to-edge distances); Microsoft Excel (Microsoft Corporation, Redmond, WA, USA); and Matlab 9.1 (Natick, MA, USA), for best-fit

Table 1 Change in total wetland, AWF, and combined wetland and AWF number, area, and perimeter, 1950s-2007

	n		Area (km^2)				Perimeter (km)			
	1950s	2007	1950s	2007	Change	Change (%)	1950s	2007	Change	Change (%)
Wetlands	33,973	26,861	1271	855	-416	-33%	27,195	20,939	-6256	-23%
AWFs	235	15,723	9	142	134	1554%	174	5863	5690	3272%
Combined[1]	34,201	42,584	1280	997	-283	-22%	27,369	26,803	-566	2%

Table 2 LULC types to which wetlands were lost and from which AWFs were gained between the 1950s-2007

	Wetland/Waterbody LULCs	Non-Wetland/Non-Waterbody LULCs
Wetlands	7%	93%
AWFs	22%	78%

trends analyses. We identified changes to individual wetland and AWF features by performing a spatial union between the 1950s and 2007 geospatial datasets, with the resulting geospatial datasets depicting the locations of changes (i.e., losses, gains) to individual wetland and AWF features. We analyzed these datasets by 12-digit Hydrologic Unit Codes (HUC12) of the National Hydrologic Database (NHD; USGS 2018), allowing a finer-grained understanding of the spatial asymmetry of the loss of wetlands and gain of AWFs. We quantified change in the area, perimeter, and spatial characteristics of wetlands, AWFs, and combined wetlands and AWFs. We chose area and perimeter, because both play crucial roles in multiple functions ranging from biogeochemical processing (e.g., Cheng and Basu 2017; Walton et al. 2020) to wildlife use (e.g., Ma et al. 2010; Straka et al. 2016). We also chose location and edge-to-edge distances because wetlands occur in networks (e.g., Xian and Crane 2005; Rains et al. 2016) and changes in network structure can change functions that emerge at the network scale (e.g., Cohen et al. 2016).

Results

Between the 1950s-2007, total wetland area and perimeter in the Tampa Bay Watershed decreased while total AWF area and perimeter in the Tampa Bay Watershed increased (Table 1). Total wetland area decreased from 1271 to 855 km^2, a loss of 33%, while total wetland perimeter decreased from 27,195 to 20,939 km, a decrease of 23%. Wetland area was disproportionately lost to non-wetland/non-water LULCs (Table 2) and was disproportionately lost in the headwaters (Fig. 3). (Here, we adopt the general definition of headwaters from the USGS [2018], which includes "the source and upper reaches of a stream" and those "parts of a river basin except the mainstream river and main tributaries.") Conversely, total AWF area increased from 9 to 142 km^2, a gain of 1554%, while total AWF perimeter increased from 174 to 5863 km, a gain of 3272%. AWF area was disproportionately gained from non-wetland/non-water LULCs (Table 2) and was disproportionately gained adjacent to Tampa Bay (Fig. 3). The loss of total wetland area was incompletely replaced by the gain in total AWF area, with total combined wetland and AWF area decreasing from 1280 to 997 km^2, for an overall combined loss of 23%. However, the loss of total wetland perimeter was almost completely replaced by the gain in total AWF perimeter, with total combined wetland and AWF perimeter decreasing from 27,369 to 26,803 km, for an overall combined loss of 2%.

Between the 1950s-2007, the mean areas of individual wetlands, AWFs, and combined wetlands and AWFs decreased (Table 3; Fig. 4). Mean ± SD wetland area decreased from 37,423 ± 494,442 to 31,839 ± 441,743 m^2, mean ± SD AWF area decreased from 36,880 ± 114,517 to 9043 ± 122,833 m^2, and mean ± SD combined wetland and AWF area decreased from 37,419 ± 492,831 to 23,422 ± 358,856 m^2. Similarly, between the 1950s-2007, the mean perimeter of individual wetlands, AWFs, and combined wetlands and AWFs decreased (Table 3; Fig. 4). Mean ± SD wetland perimeter decreased from 800 ± 4241 to 780 ± 3763 m, mean ± SD AWF perimeter decreased from 740 ± 930 to 373 ± 579 m, and mean ± SD combined wetland and AWF perimeter decreased from 800 ± 4227 to 629 ± 3016 m.

Between the 1950s-2007, the mean distance between individual wetlands increased while the mean distance between individual AWFs decreased (Fig. 5). In the 1950s, wetlands were typically < 128 m apart and often < 32 m apart; by 2007, wetlands were still typically < 128 m apart but rarely < 32 m apart. This trend was most prominent in the headwaters. Conversely, in the 1950s, AWFs were typically either not present or > 128 m apart; by 2007, AWFs were widespread and often < 128 m apart. This trend was most prominent adjacent to Tampa Bay. Combined, the distance between combined wetlands and AWFs generally increased in the headwaters and decreased adjacent to Tampa Bay.

Between the 1950s-2007, the distribution of the distances between wetlands was generally unchanged, while the distribution of the distances between AWFs changed (Fig. 6). In the 1950s, wetlands tended to be relatively close to one another, with wetland edges most frequently approximately

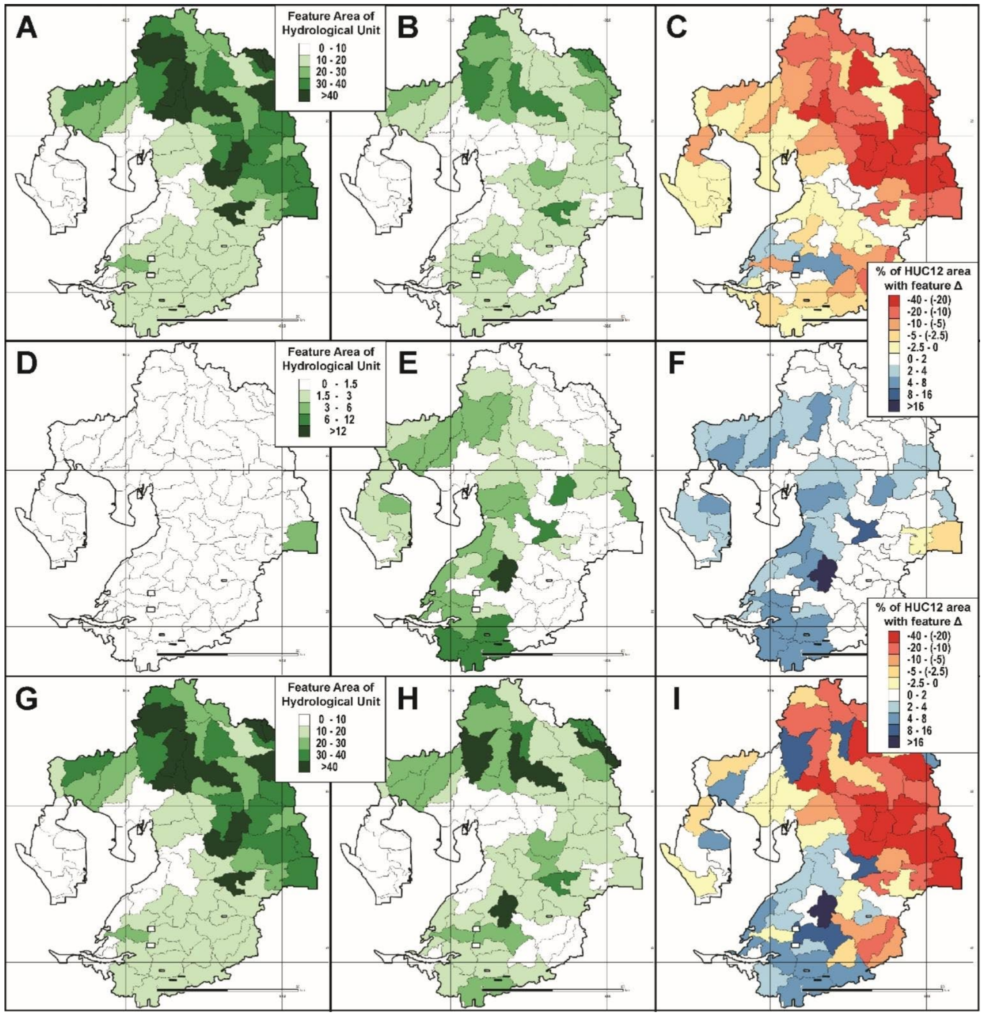

Fig. 3 Change in wetland, AWF, and combined wetland and AWF area, 1950s-2007, aggregated at the HUC 12 level. In the first two columns, feature area is expressed as a percentage of the total area of the HUC 12. In the last column, the change in feature area in each HUC 12 is expressed as a percent difference. A – wetland area, 1950s; B – wetland area, 2007; C – change in wetland area, 1950s-2007; D – AWF area, 1950s; E – AWF area, 2007; F – change in AWF area, 1950s-2007; G – combined wetland and AWF area, 1950s; H – combined wetland and AWF area, 2007; I – change in combined wetland and AWF area, 1950s-2007

Table 3 Change in the mean wetland, AWF, and combined wetlands and AWF area, perimeter, and perimeter:area, 1950s-2007

	n	Mean ± SD Area (m^2)	Mean ± SD Perimeter (m)	Mean P:A
Wetlands				
1950s	33,973	37,423 ± 494,442	800 ± 4241	0.02
2007	26,861	31,839 ± 441,743	780 ± 3763	0.02
AWFs				
1950s	235	36,880 ± 114,517	740 ± 930	0.02
2007	15,723	9043 ± 122,833	373 ± 579	0.04
Combined				
1950s	34,201	37,419 ± 492,831	800 ± 4227	0.02
2007	42,584	23,422 ± 358,856	629 ± 3016	0.03

150 m apart and rarely being more than 400 m apart, while AWFs tended to be equally likely to be within approximately 150, 400, or even up to 1000 m apart. In 2007, wetlands still tended to be relatively close to one another, with wetland edges still most frequently approximately 150 m apart and rarely being more than 400 m apart, but AWFs became more likely to be close to one another, with AWFs generally following the wetland distribution, being most frequently approximately 150 m apart and rarely more than 400 m apart.

Discussion

Between the 1950s-2007, the waterscape was substantially reorganized in the Tampa Bay Watershed. The overall trend has been toward fewer, smaller wetlands and more, smaller AWFs. The loss of wetland area has far exceeded the gain in AWF area, though the loss of wetland perimeter has nearly been equaled by the gain in AWF perimeter. The loss of wetlands and the gain of AWFs have predominantly occurred in different geographic locations within the Tampa Bay Watershed, creating a spatial asymmetry between losses and gains. Overall, the physical structure of the waterscape of today is different than the physical structure of the waterscape of the past.

Loss in wetland area was partially but incompletely replaced by gain in AWF area (Table 1; Rains et al. 2013). The loss of wetland area was widespread but was especially

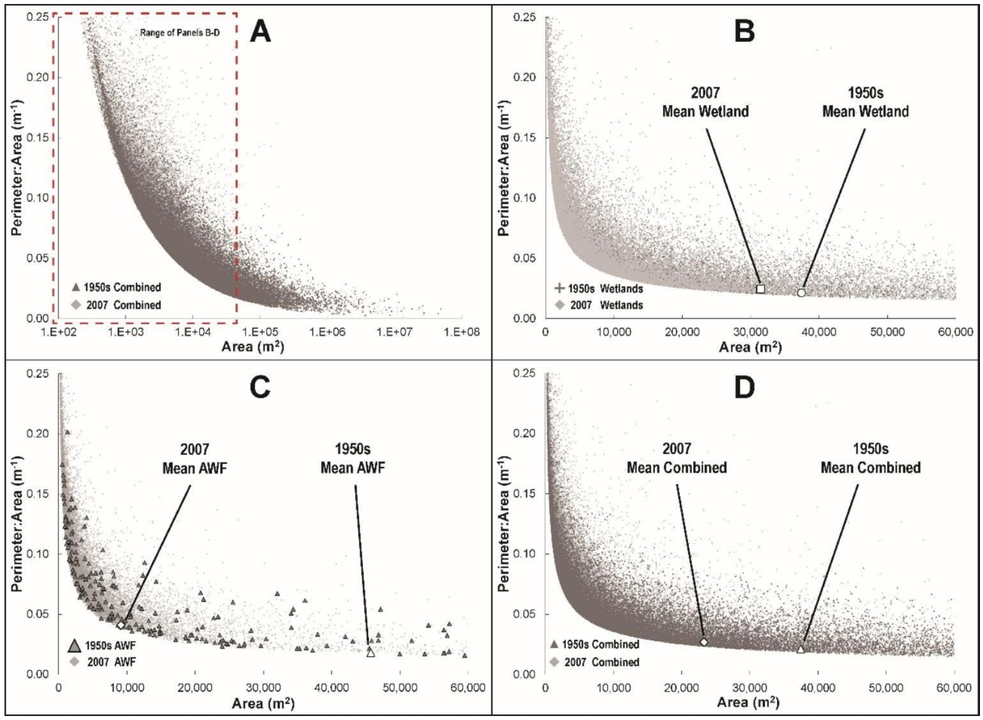

Fig. 4 Change in wetland, AWF, and combined wetland and AWF perimeter:area v area, 1950s-2007. A – perimeter:area v area for all combined wetlands and AWFs, 1950s and 2007; B – perimeter:area v area for the subset of wetlands in the panel A inset, 1950s and 2007; C – perimeter:area v area for the subset of AWFs in the panel A inset, 1950s and 2007; D – perimeter:area v area for the subset of combined wetlands and AWFs in the panel A inset, 1950s and 2007

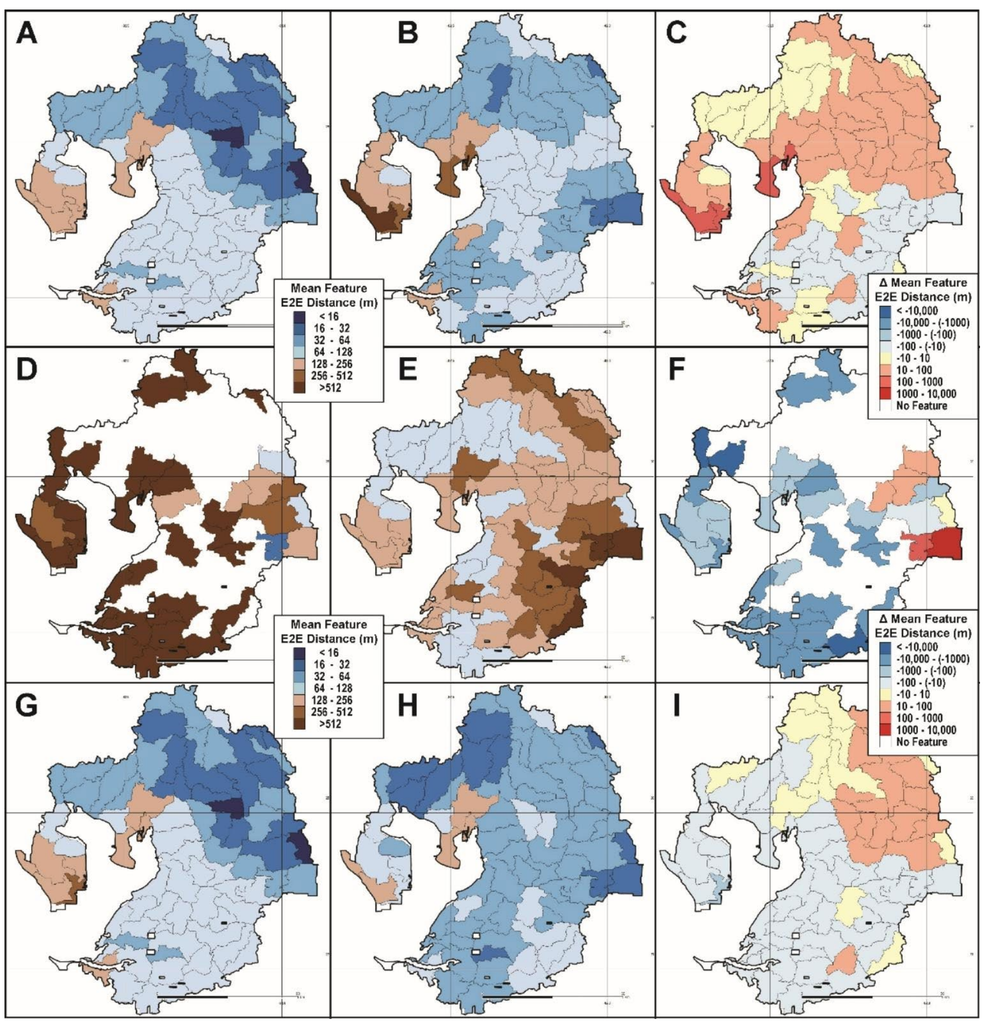

Fig. 5 Change in mean distance between wetlands, AWFs, and combined wetlands and AWFs, 1950s-2007, aggregated at the HUC 12 level. A – mean distance between wetlands, 1950s; B – mean distance between wetlands, 2007; C – change in mean distance between wetlands, 1950s-2007; D – mean distance between AWFs, 1950s; E – mean distance between AWFs, 2007; F – change in mean distance between AWFs, 1950s-2007; G – mean distance between combined wetlands and AWFs, 1950s; H – mean distance between combined wetlands and AWFs, 2007; I – change in mean distance between combined wetlands and AWFs, 1950s-2007

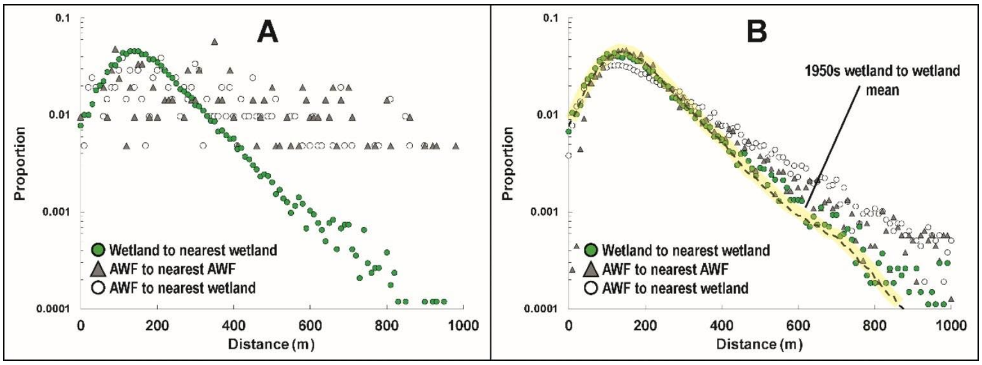

Fig. 6 Change in the distribution of distances between wetlands, AWFs, and combined wetlands and AWFs, 1950s-2007. A – In the 1950s, wetlands and AWFs had different distributions, with wetlands likely to be close to one another and unlikely to be far from one another and AWFs equally likely to be close to one another or far from one another. B – In 2007, wetlands and AWFs had similar distributions, with both likely to be close to one another and unlikely to be far from one another

concentrated in the headwaters (Fig. 3). These spatial trends are consistent with development since the 1950s, which has occurred primarily in the suburban periphery of the Tampa-St. Petersburg-Clearwater metropolitan area. This includes suburban development along Interstate 4, which was constructed in 1959 to connect the Tampa-St. Petersburg-Clearwater and Orlando metropolitan areas (e.g., Xian and Crane 2005). This also includes phosphate mining which has occurred in the Central Phosphate District, partially located in the east-central portion of the Tampa Bay Watershed (e.g., Brown 2005). Meanwhile, AWFs became a prominent landscape element (Table 1). This gain was widespread but was especially concentrated adjacent to Tampa Bay (Fig. 3). These spatial trends are consistent with infilling of the Tampa-St. Petersburg-Clearwater metropolitan area (e.g., Xian and Crane 2005). Overall, one-third of the wetland area was lost and just one-third of that lost wetland area was replaced by gained AWF area, resulting in an overall loss of 23% of the combined wetland and AWF area (Table 1).

Wetland area was disproportionately lost to non-wetland and non-waterbody LULCs (Table 2; Rains et al. 2013). Just 7% of the lost wetland area was lost to wetland or waterbody LULCs, including natural waterbodies (e.g., forested wetlands to open water) or AWFs (e.g., forested wetlands to stormwater retention ponds). Meanwhile, AWF area was disproportionately created from what had been non-wetland or non-waterbody LULCs (Table 2). Just 22% of the gained AWF area was gained from wetland and or waterbody LULCs (e.g., forested wetlands to stormwater retention ponds). These results are qualitatively similar to statewide results that can be inferred from the National Wetlands Inventory's newly developed Difference Product Line, which indicate that open water area was disproportionately gained from non-wetland or non-waterbody LULCs between 1984 and 2016 (US Fish and Wildlife Service 2023).

Wetlands and AWFs also changed in both total number and mean size. Wetlands became less numerous and smaller (Table 3; Fig. 4). This implies both the complete loss of entire wetlands and perhaps also the partial loss of wetlands by encroachment into the wetland margins, the latter a form of "nibbling" (*sensu* Lee and Gosselink 1988). Meanwhile, AWFs became far more numerous but smaller (Table 3; Fig. 4). These changes reflect a change from predominantly reservoirs prior to the 1950s to the more-recent mixture of stormwater retention ponds, golf course water features, and reservoirs by 2007. Together, there are now more, smaller combined wetlands and AWFs (Table 3; Fig. 3). The example here may be indicative a broader trend, with evidence suggesting that urban waterbodies converge on moderate sizes and simpler shapes throughout the U.S., presumably as smaller waterbodies are lost and larger waterbodies are physically reshaped around their margins (Steele and Heffernan 2014; Steele et al. 2014).

Lost wetland perimeter was largely replaced by gained AWF perimeter from the strict standpoint of the total perimeter length in the landscape (Table 1). However, lost wetland perimeter was unlikely replaced by gained AWF perimeter from the standpoint of the total functional capacity of total perimeter in the landscape. Wetland and AWF perimeter differ in the Tampa Bay Watershed, with wetland edge typically gently sloped (Haag and Lee 2010) and AWF edge typically constructed with slopes at 1:4 and occasionally up to 1:2 under certain circumstances (Hillsborough County 2021). The more gently sloped edge of the wetlands is more conducive to a gradual vegetation and hydrological gradient that provides better support for many functions, including biogeochemical processing (Mayer et al. 2007; Creed et al. 2013) and wading bird foraging (Bancroft et al. 2002; Binkley et al. 2019). This implies that the near equal replacement of perimeter length has not resulted in an equal replacement of the functions provided by wetland edge.

Wetlands and AWFs also changed in individual and combined distribution. Wetlands became further apart (Fig. 5), which might have naturally followed from the fact that wetlands also became less numerous and smaller (Table 3; Fig. 4). However, wetlands remained generally close to one another, in part because wetlands occur in localized landscape positions defined by specific climatic, geologic, and topographic characteristics (e.g., Johnson et al. 2010; Stepchinski et al. 2023). Meanwhile, AWFs became closer to one another (Fig. 5), which naturally followed from the fact that AWFs also became far more numerous (Table 3; Fig. 4). More strikingly, AWFs went from being equally likely to be close to or far from one another to being generally close to one another (Fig. 6). It is not entirely clear why AWFs became generally close to one another, though stormwater retention ponds are typically in developed areas (Beckingham et al. 2019) and golf course water features are always on golf courses, and developed areas and golf courses are non-uniformly distributed and commonly clustered themselves. Whatever the case, AWF distributions now mirror the more natural wetland distributions.

The changes in wetlands and AWFs are driven by differing authorization and motivation, because all LULC change is driven by concentrated political, institutional, cultural, natural, and spatial drivers (Plieninger et al. 2016). Wetland protections have been authorized by overlapping federal and state statutes and related regulations. At the federal level, wetlands are protected under Clean Water Act Sect. 404, which is authorized under 33 CFR Part 323. Under those protections, an applicant is required to conduct an alternatives analysis to arrive at the least environmentally damaging practicable alternative, a process that includes documenting efforts to avoid, minimize, and mitigate impacts to wetlands resulting from the proposed action. In Florida, wetlands are further protected under the Environmental Resource Permit (ERP) program, which is authorized under FS 373 Part IV. Under those protections, an applicant also must seek to avoid and minimize impacts to wetlands, and specific provisions are set forth to determine the amount of mitigation needed to offset such impacts to wetlands. Crucially, both allow unavoidable impacts to wetlands with sufficient mitigation, including the use of offsite mitigation. Therefore, wetland mitigation is often spatially decoupled from development. In Florida, this has resulted in widespread transfer of wetland area and function from project sites to offsite mitigation sites, including mitigation banks (Goldberg and Reiss 2016). Meanwhile, the creation of prominent types of AWFs, including stormwater retention ponds and golf course water features, is authorized and/or motivated by a variety of statutes and related regulations and other unrelated market forces. In Florida, stormwater management falls under the ERP program, which again is authorized under FS 373 Part IV. A central emphasis is on onsite stormwater retention, including through the construction of onsite stormwater retention ponds (see Harper and Baker 2007; see also Hillsborough County 2021). Meanwhile, the development of golf courses surged in the latter half of the 20th century, both nationwide (Napton and Laingen 2008) and in Florida (Haydu and Hodges 2002). These golf courses commonly include water features, often integrated into onsite stormwater management plans (Hurdzan 2006; Florida Department of Environmental Protection 2012). Therefore, unlike wetland loss, AWF gain is commonly spatially coupled to development. Given these differing authorizations and motivations, a complete and symmetric replacement of wetlands by AWFs would likely have been purely coincidental.

Humans prefer to live near freshwater, with approximately 50% and 90% of the global population living within one and 10 km of freshwater environments, respectively (Bin 2005; Kummu et al. 2011). Studies of home values suggest we prefer living adjacent to open water rather than vegetated wetlands (Mahan et al. 2000). However, not all open water is created equal, and further studies of home values suggest we prefer to not live adjacent to stormwater retention ponds unless they are integrated into a mixed-use, park-like setting (Lee and Li 2009). As we reshape our environments to match our preferences, we inadvertently create consequences to the larger ecological waterscape. In the natural state, waterscapes are structurally and behaviorally complex (Peipoch et al. 2015). These complexities support ecosystem functions necessary to maintain ecological resilience (Odum 1962; Gunderson and Holling 2001) and provide the natural capital necessary to produce the ecological services that maintain human well-being (Boyd and Banzhaf 2007; Kleindl et al. 2018). However, as we reshape the waterscape to meet our preferences, we simultaneously reduce complexity at local and landscape scales leading to ecological simplification (*sensu* Peipoch et al. 2015). Such simplification may be common, with urban areas throughout the U.S. converging in terms of the numbers, areas, and shapes of their waterbodies (Steele and Heffernan 2014; Steele et al. 2014). Crucially, it is not just the presence or absence of aquatic features within the waterscape but, rather, the presence or absence and spatial characteristics of those features within the waterscape that control function (e.g., Callahan et al. 2015; Callahan et al. 2017), potentially including functions that only emerge at scale and in the aggregate (e.g., Cohen et al. 2016; Rains et al. 2016; Thorslund et al. 2018; Stepchinski et al. 2023). AWFs do perform some functions and provide some related ecological services at high levels, with evidence suggesting that stormwater retention ponds meet the goals of flood storage throughout Florida (Harper and Baker 2007). However, AWFs are not direct substitutes for wetlands due to structural differences in characteristics such as landscape position, basin morphology, hydroperiod, soil, and vegetation (Rooney et al. 2015; Beckingham et al. 2019; Hess et al. 2022), which is likely reflected in a change in both the functions performed and the related ecological services provided at the local and landscape scales.

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Data Availability The supporting geospatial datasets are available on Figshare: https://doi.org/10.6084/m9.figshare.22138046.v1.

Declarations

Competing Interests Mark Rains is currently serving as the Chief Science Officer for the State of Florida. The views expressed in this article are his own and do not necessarily reflect the views of the State of Florida.

References

Balerna JA, Kramer AM, Landry SM, Rains MC, Lewis DB (2023) Synergistic Effects of Drought and Groundwater extraction on Freshwater Wetland Inundation. J Environ Manage 337. https://doi.org/10.1016/j.jenvman.2023.117690

Bancroft GT, Gawlik DE, Rutchey K (2002) Distribution of wading birds relative to Vegetation and Water Depths in the Northern Everglades of Florida, USA. Waterbirds 25:265–277. https://doi.org/10.1675/1524-4695(2002)025[0265:DOWBRT]2.0.CO;2

Beckingham B, Callahan T, Vulava V (2019) Stormwater Ponds in the Southeastern U.S. Coastal Plain: Hydrogeology, Contaminant Fate, and the need for a Social-Ecological Framework. Front Environ Sci 7

Bin O (2005) A semiparametric hedonic model for valuing wetlands. Appl Econ Lett 12:597–601. https://doi.org/10.1080/13504850500188505

Binkley EE, Dorn NJ, Cook MI (2019) Feeding on the edge: foraging White Ibis target inter-habitat prey fluxes. J Field Ornithol 90:235–247. https://doi.org/10.1111/jofo.12302

Boyd J, Banzhaf S (2007) What are ecosystem services? The need for standardized environmental accounting units. Ecol Econ 63:616–626

Brinson MM (1993) Changes in the functioning of wetlands along environmental gradients. Wetlands Vol 13:65–74

Brown MT (2005) Landscape restoration following phosphate mining: 30 years of co-evolution of science, industry and regulation. Ecol Eng 24:309–329. https://doi.org/10.1016/j.ecoleng.2005.01.014

Callahan MK, Rains MC, Bellino JC et al (2015) Controls on temperature in salmonid-bearing headwater streams in two common hydrogeologic settings, kenai peninsula, alaska. J Am Water Resour Assoc 51:84–98. https://doi.org/10.1111/jawr.12235

Callahan MK, Whigham DF, Rains MC et al (2017) Nitrogen subsidies from hillslope alder stands to streamside wetlands and headwater streams, kenai peninsula, alaska. J Am Water Resour Assoc 53:478–492. https://doi.org/10.1111/1752-1688.12508

Cheng FY, Basu NB (2017) Biogeochemical hotspots: role of small water bodies in landscape nutrient processing. Water Resour Res 53:5038–5056. https://doi.org/10.1002/2016WR020102

Cicchetti G, Greening H (2011) Estuarine Biotope Mosaics and Habitat Management Goals: an application in Tampa Bay, FL, USA. Estuaries Coasts 34:1278–1292. https://doi.org/10.1007/s12237-011-9408-4

Cohen MJ, Creed IF, Alexander L et al (2016) Do geographically isolated wetlands influence landscape functions? Proc Natl Acad Sci 113:1978–1986. https://doi.org/10.1073/pnas.1512650113

Creed IF, Miller J, Aldred D et al (2013) Hydrologic profiling for greenhouse gas effluxes from natural grasslands in the prairie pothole region of Canada. J Geophys Research: Biogeosciences 118:680–697. https://doi.org/10.1002/jgrg.20050

Creed IF, Lane CR, Serran JN et al (2017) Enhancing protection for vulnerable waters. Nat Geosci 10:809–815. https://doi.org/10.1038/ngeo3041

Dahl TE (1990) Wetlands losses in the United States, 1780's to 1980's. Report to the Congress. National Wetlands Inventory, St. Petersburg, FL (USA), U.S. Fish & Wildlife Service

Dahl TE (2005) Florida's wetlands: an update on status and trends, 1985 to 1996. U.S. Fish & Wildlife Service, Washington, D.C

Dahl TE (2006) Status and trends of wetlands in the conterminous United States, 1998 to 2004. 112

Dahl TE (2011) Status and Trends of Wetlands in the Conterminous United States 2004 to 2009. U.S. Department of the Interior; Fish and Wildlife Service, Washington, D.C

Downing DM, Winer C, Wood LD (2003) Navigating through clean water act jurisdiction: a legal review. Wetlands 23:475–493 10.1672/0277-5212(2003)023[0475:NTCWAJ]2.0.CO;2

Florida Department of Environmental Protection (2012) Best Management Practices for the Enhancement of Environmental Quality on Florida Golf Courses. 136

Frayer WE, Hefner JM (1991) Florida wetlands: status and trends, 1970's to 1980's /. U.S. Fish & Wildlife Service, Atlanta, GA

Fretwell JD, Williams JS, Redman PJ (1996) National water summary on wetland resources. Natl Water Summary Wetland Resour 2425:439. https://doi.org/10.3133/wsp2425

Goldberg N, Reiss KC (2016) Accounting for Wetland loss: Wetland Mitigation Trends in Northeast Florida 2006–2013. Wetlands 36:373–384. https://doi.org/10.1007/s13157-016-0749-4

Gunderson LH, Holling CS (eds) (2001) Panarchy: understanding transformations in Human and Natural Systems. Island Press

Haag KH, Lee TM (2010) Hydrology and Ecology of Freshwater Wetlands in Central Florida - A Primer. Hydrology and Ecology of Freshwater Wetlands in Central Florida - A Primer. https://doi.org/10.3133/cir1342

Harper HH, Baker DH (2007) Evaluation of current stormwater design criteria within the state of Florida. Florida Department of Environmental Protection. Florida Department of Environmental Protection, Tallahassee, FL

Haydu JJ, Hodges AW (2002) Economic impacts of the Florida golf course industry. University of Florida, Institute of Food and Agricultural Sciences, Gainsville, FL

Hefner JM (1986) Wetlands of Florida 1950s to 1970s. In: Estevez ED, Miller J, Moris J, Hamman R (eds) Managing cumulative effects in Florida wetlands. Omni Press, Madison, WI, pp 23–31

Hefner JM, Brown JD (1984) Wetland trends in the Southeastern United States. Wetlands 4:1–11. https://doi.org/10.1007/BF03160482

Hess KM, Sinclair JS, Reisinger AJ et al (2022) Are stormwater detention ponds protecting urban aquatic ecosystems? A case study using depressional wetlands. Urban Ecosyst 25:1155–1168. https://doi.org/10.1007/s11252-022-01208-9

Hillsborough C (2021) Stormwater Management Technical Manual. https://www.hillsboroughcounty.org/library/hillsborough/media-center/documents/public-works/ttm/ttm-2021/stormwater-management-technical-manual-october-2021.pdf. Accessed 1 Nov 2022

Hurdzan MJ (2006) Golf course architecture: evolutions in design, construction, and restoration technology, 2nd edn. J. Wiley & Sons, Hoboken, N.J

Johnson WC, Werner B, Guntenspergen GR et al (2010) Prairie Wetland Complexes as Landscape Functional Units in a changing climate. Bioscience 60:128–140. https://doi.org/10.1525/bio.2010.60.2.7

Kleindl W, Stoy P, Binford M et al (2018) Toward a Social-Ecological Theory of Forest Macrosystems for Improved Ecosystem Management. Forests 9:200. https://doi.org/10.3390/f9040200

Knochenmus LA (2006) Regional evaluation of the hydrogeologic framework, hydraulic properties, and chemical characteristics of the intermediate aquifer system underlying southern west-central Florida. U.S. Geological Survey, Washington, D.C

Kummu M, de Moel H, Ward PJ, Varis O (2011) How close do we live to Water? A Global Analysis of Population Distance to Freshwater Bodies. PLoS ONE 6:e20578. https://doi.org/10.1371/journal.pone.0020578

Lee LC, Gosselink JG (1988) Cumulative impacts on wetlands: linking scientific assessments and regulatory alternatives. Environ Manage 12:591–602. https://doi.org/10.1007/BF01867538

Lee JS, Li M-H (2009) The impact of detention basin design on residential property value: case studies using GIS in the hedonic price modeling. Landsc Urban Plann 89:7–16. https://doi.org/10.1016/j.landurbplan.2008.09.002

Ma Z, Cai Y, Li B, Chen J (2010) Managing Wetland Habitats for Waterbirds: an International Perspective. Wetlands 30:15–27. https://doi.org/10.1007/s13157-009-0001-6

Mahan BL, Polasky S, Adams RM (2000) Valuing urban wetlands: a property price approach. Land Econ 100–113

Mayer PM, Reynolds SK Jr, McCutchen MD, Canfield TJ (2007) Meta-analysis of Nitrogen removal in riparian buffers. J Environ Qual 36:1172–1180. https://doi.org/10.2134/jeq2006.0462

McIntyre NE, Collins SD, Heintzman LJ et al (2018) The challenge of assaying landscape connectivity in a changing world: a 27-year case study in the southern Great Plains (USA) playa network. Ecol Ind 91:607–616. https://doi.org/10.1016/j.ecolind.2018.04.051

Miller JA (1997) The geology of Florida. In: Jones DA, Randazzo R (eds) The hydrogeology of Florida. University PRess of Florida, Gainsville, FL, pp 69–88

Napton DE, Laingen CR (2008) Expansion of Golf Courses in the United States*. Geogr Rev 98:24–41. https://doi.org/10.1111/j.1931-0846.2008.tb00286.x

Nilsson KA, Rains MC, Lewis DB, Trout KE (2013) Hydrologic Characterization of 56 Geographically Isolated Wetlands in West-Central Florida Using a Probabilistic Method. Wetland Ecology and Management 21:1–14 https://doi.org/10.1007/s11273-012-9275-1

Nowicki RS, Rains MC, LaRoche JJ, Pasek MA (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 RS, Rains MC, LaRoche JJ et al (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

Odum EP (1962) Relationships between structure and function in the ecosystem. Japanese J Ecol 12:108–118

Peipoch M, Brauns M, Hauer FR et al (2015) Ecological simplification: human influences on Riverscape Complexity. Bioscience. https://doi.org/10.1093/biosci/biv120

Plieninger T, Draux H, Fagerholm N et al (2016) The driving forces of landscape change in Europe: a systematic review of the evidence. Land Use Policy 57:204–214. https://doi.org/10.1016/j.landusepol.2016.04.040

Rains MC, Landry S, Rains KC et al (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 MC, Leibowitz SG, Cohen MJ et al (2016) Geographically isolated wetlands are part of the hydrological landscape. Hydrol Process 30:153–160. https://doi.org/10.1002/hyp.10610

Rooney RC, Foote L, Krogman N et al (2015) Replacing natural wetlands with stormwater management facilities: Biophysical and perceived social values. Water Res 73:17–28. https://doi.org/10.1016/j.watres.2014.12.035

Serran JN, Creed IF (2016) New mapping techniques to estimate the preferential loss of small wetlands on prairie landscapes. Hydrol Process 30:396–409. https://doi.org/10.1002/hyp.10582

Serran JN, Creed IF, Ameli AA, Aldred DA (2018) Estimating rates of wetland loss using power-law functions. Wetlands 38:109–120. https://doi.org/10.1007/s13157-017-0960-y

Sinclair WC (1974) Hydrogeologic characteristics of the surficial aquifer in northwest Hillsborough County, Florida. Florida Bureau of Geology, Tallahassee, FL

Southwest Florida Water Management District (2009) 2009 SWFWMD Photointerpretation Key. https://www.sfwmd.gov/sites/default/files/documents/2009_pi-key.pdf. Accessed 1 Jan 2023

Southwest Florida Water Management District (2008) 2007 land use/cover classifications [Computer File, GIS shapefiles]. http://www.swfwmd.state.fl.us/data/gis/layer_library/metadata/lu07.html. Accessed 1 Dec 2008

Stedman S, Dahl TE, the eastern United States (2008) Status and trends of wetlands in the coastal watersheds of. 1998 to 2004.

National Oceanic and Atmospheric Administration. National Marine Fisheries Service and US Department of the Interior, Fish and Wildlife Service, Washington, D.C.

Steele MK, Heffernan JB (2014) Morphological characteristics of urban water bodies: mechanisms of change and implications for ecosystem function. Ecol Appl 24:1070–1084. https://doi.org/10.1890/13-0983.1

Steele MK, Heffernan JB, Bettez N, Cavender-Bares J, Groffman PM, Grove JM, Hall S, Hobbie SE, Larson K, Morse JL, Neill C, Nelson KC, O'Neil-Dunne J, Ogden L, Pataki DE, Polsky C, Roy Chowdhury R (2014) Convergent Surface Water Distributions in U. S Cities Ecosystems 17:685–697. https://doi.org/10.1007/s10021-014-9751-y

Stepchinski LM, Rains MC, Lee LC, Lis RA, Nutter WL, Rains KC, Stewart SR (2023) Hydrologic connectivity and Flow Generation from California Vernal Pool, Swale, and Headwater Stream Complexes to downstream Waters. Wetlands 43. https://doi.org/10.21203/rs.3.rs-2082455/v1

Straka TM, Lentini PE, Lumsden LF et al (2016) Urban bat communities are affected by wetland size, quality, and pollution levels. Ecol Evol 6:4761–4774. https://doi.org/10.1002/ece3.2224

Tampa Bay Water (2022) Tampa Bay Regional Drinking Water Sources. In: Tampa Bay Water. https://www.tampabaywater.org/tampa-bay-regional-drinking-water-sources. Accessed 1 Nov 2022

Thorslund J, Cohen MJ, Jawitz JW et al (2018) Solute evidence for hydrological connectivity of geographically isolated wetlands. Land Degrad Dev 29:3954–3962. https://doi.org/10.1002/ldr.3145

Tihansky B, Knochenmus LA (2001) Karst features and hydrogeology in West-central Florida—A Field Perspective. U.S. Geological Survey, St. Petersburg, Florida

US Fish and Wildlife Service (2023) National Wetlands Inventory Difference Product Line. https://www.fws.gov/project/national-wetlands-inventory-difference-product-line. Accessed 1 Jan 2023

USGS (U.S. Geological Survey) (2018) National Hydrography Dataset, October 2018. https://www.usgs.gov/national-hydrography/access-national-hydrography-products. Accessed 1 Nov 2018

USGS (U.S. Geological Survey) Water School (2018) Water Science Glossary. https://www.usgs.gov/special-topics/water-science-school/science/water-science-glossary Accessed 13 June 2023

Van Meter KJ, Basu NB (2015) Signatures of human impact: size distributions and spatial organization of wetlands in the Prairie Pothole landscape. Ecol Appl 25:451–465. https://doi.org/10.1890/14-0662.1

Walton CR, Zak D, Audet J et al (2020) Wetland buffer zones for nitrogen and phosphorus retention: impacts of soil type, hydrology and vegetation. Sci Total Environ 727:138709. https://doi.org/10.1016/j.scitotenv.2020.138709

Xian G, Crane M (2005) Assessments of urban growth in the Tampa Bay watershed using remote sensing data. Remote Sens Environ 97:203–215. https://doi.org/10.1016/j.rse.2005.04.017