4. Aquatic Life

4.1. Submerged Aquatic Vegetation (SAV)

4.1.1. Description

Dating back to 1773, records indicate that extensive SAV beds existed in the river (**Bartram 1928**). Since that time, people have altered the natural system by dredging, constructing seawalls, contributing chemical contamination, and sediment and nutrient loading (**DeMort 1990**; **Dobberfuhl 2007**). SAV found in the LSJRB (see Table 4.1) are primarily freshwater and brackish water species. Commonly found species include tape grass *(Vallisneria americana)*, water naiad *(Najas guadalupensis)*, and widgeon grass *(Ruppia maritima).* Tape grass forms extensive beds when conditions are favorable. Water naiad and widgeon grass form bands within the shallow section of the SAV bed. Tape grass is a freshwater species that tolerates brackish conditions, water naiad is exclusively freshwater and wigeon grass is a brackish water species that can live in very salty water (**White et al. 2002; Sagan 2010)**. *Ruppi*a does not form extensive beds. It is restricted to the shallow, near shore section of the bed and has never formed meadows as extensive as *Vallisneria* even when salinity has eliminated *Vallisneria* and any competition, or other factors change sufficiently to support *Ruppia* (**Sagan 2010**).

Other freshwater species include: muskgrass *(Chara sp.)*, spikerush *(Eleocharis sp.)*, water thyme *(Hydrilla verticillata*; an invasive non-native weed), baby's-tears *(Micranthemum sp.)*, sago pondweed *(Potamogeton pectinatus)*, small pondweed *(Potamogeton pusillus)*, awl-leaf arrowhead *(Sagittaria subulata)*, horned pondweed *(Zannichellia palustris*), and coontail *(Ceratophyllum demersum)* (**IFAS 2007**; **Sagan 2006**; **USDA 2013**; **Trent 2021**). **DeMort 1990** surveyed four locations for submerged macrophytes in the LSJR and indicated that greater consistency in species distributions occurred south of Hallows Cove (St. Johns County) with tape grass being the dominant species. North of this location, widgeon grass and sago pondweed were the dominant species until 1982-1987, when tape grass coverage increased 30%, and is now the most dominant species encountered.

The greatest distribution of SAV in Duval County is in waters south of the Fuller Warren Bridge (**Kinnaird 1983b**; **Dobberfuhl 2002**; **Dobberfuhl and Trahan 2003**; **Sagan 2004**; **Sagan 2006**; **Sagan 2007**; **Goldberg et al. 2018**). Submerged aquatic vegetation in the tannin-rich, black water LSJR is found exclusively in four feet or less of water depth. Poor sunlight penetration prevents the growth of SAV in deeper waters. **Dobberfuhl 2007** confirmed that the deeper outer edge of the grass beds occurs at about three feet in the LSJRB. Rapid regeneration of grass beds occurs annually in late winter and spring when water temperatures become more favorable for plant growth and the growing season continues through September (**Dobberfuhl 2007**; **Thayer et al. 1984**). SAV beds, especially *Vallisneria*, are present year-round and are considered “evergreen” in Florida (**Sagan 2010**).

Sunlight is vital for good growth of submerged grasses. Sunlight penetration may be reduced because of increased color, turbidity, pollution from upland development, and/or disturbance of soils. Deteriorating water quality has been shown to cause a reduction in grass beds (**Linhoss et al. 2015**). This leads to erosion and further deterioration of water quality.

In addition to the amount of light, the frequency and duration of elevated salinity events in the river can adversely affect the health of SAV (**Jacoby 2011**). In lab studies, **Twilley and Barko 1990** showed that tape grass grows well from 0-12 parts per thousand of salinity and can tolerate water with salinities up to 15-20 parts per thousand for short periods of time. Also, SAV requires more light in a higher salinity environment because of increased metabolic demands (**Dobberfuhl 2007**). Finally, evidence suggests that greater light availability can lessen the impact of high salinity effects on SAV growth (**French and Moore 2003**; **Kraemer et al. 1999**).

Dobberfuhl 2007 noted that, during drought conditions, there is an increase in light availability that likely causes specific competition between the grasses and organisms growing on the surface of the grasses (Table 4.1). Many of these epiphytic organisms block light and can be detrimental to normal growth of the tape grass. As a result, this fouling causes an increase in light requirements for the SAV (**Dunn et al. 2008b**).

Table 4.1 Submerged aquatic vegetation in the Lower St. Johns River.


(Photo: SJRWMD)

Tape grass (*Vallisneria americana*)

- Teeth on edge of leaves
- Leaves flat, tape-like; 0.5-4 cm wide
- Leaves taper at tip
- No obvious stem
- Height: 4-90 cm
  (a small one can be confused with *Sagittaria subulata*)


(Photo: SJRWMD)

Water naiad (*Najas guadalupensis*)

- Leaf whorls not tightly packed
- Leaf pairs/whorls separated by large spaces on stem
- Leaves opposite, usually in pairs, sometimes in whorls of three
- Leaves with teeth (must look closely); 2 mm wide


(Photo: SJRWMD)

Widgeon grass (*Ruppia maritima*)

- Leaves alternate, tapering at end
- Leaves thread-like; 0.5 mm wide
- Height: 4-20 cm


(Photo: Kerry Dressler)

Muskgrass (*Chara sp.*)

- Leaf whorls separated by conspicuous spaces
- Leaf not forked
- Leaves stiff and scratchy to touch
- Height: 2-8 cm


(Photo: SJRWMD)

Spikerush (*Eleocharis sp.*)

- No teeth on leaves
- Leaves round, pencil-like; 1-3 mm wide
- Leaves as broad at tip as at base
- Height: 1-5 cm


(Photo: Kerry Dressler)

Water thyme (*Hydrilla verticillata*)

- Leaf whorls tightly packed
- Leaves opposite, in whorls of four to eight leaves
- Leaves with conspicuous teeth, making plant scratchy to the touch
- Leaf tip pointed; leaves 2-4 mm wide
- Height: 5-15 cm


(Photo: SJRWMD)

Baby's-tears (*Micranthemum sp.*)

- Leaf whorls not tightly packed
- Leaf opposite, in whorls of three to four leaves
- No teeth on leaves
- Leaf tip rounded; 2-4 mm wide
- Height: 2-15 cm


(Photo: SJRWMD)

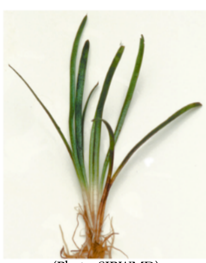
Sago pondweed (*Potamogeton pectinatus*)

- Leaves alternate; 0.5-4.5 cm wide
- No teeth on leaves
- Leaves long and narrowing with pointed tips
- Stems thread-like
- Height: 5-20 cm


(Photo: SJRWMD)

Small pondweed (*Potamogeton pusillus*)

- Leaves alternate; 0.5-3 mm wide
- No teeth on leaves
- Leaves long and narrow with blunted or rounded tips
- Stems thread-like
- Height: 5-20 cm


(Photo: SJRWMD)

Awl-leaf arrowhead (*Sagittaria subulata*)

- No teeth on leaves
- Leaves triangular, spongy; 3-8 mm wide
- Leaves taper at tip
- Height: 1-5 cm


(Photo: SJRWMD)

Horned pondweed (*Zannichellia palustris*)

- Leaves opposite
- No teeth on leaves
- Long narrow leaves with blunted tips
- Stems thread-like
- Often seen with kidney-shaped fruit
- Height: 1-8 cm

4.1.2. Significance

SAV provides nurseries for a variety of aquatic life, helps to prevent erosion, and reduces turbidity by trapping sediment. Scientists use SAV distribution and abundance as major indicators of ecosystem health (**Dennison et al. 1993**). SAV is important ecologically and economically to the LSJRB. SAV persists year-round in the LSJRB and forms extensive beds, which carry out the ecological role of "nursery area" for many important invertebrates, and fish. Also, aquatic plants and SAV provide food for the West Indian manatee *Trichechus manatu* (**White et al. 2002**). Manatees consume from 4-11% of their body weight daily, with *Vallisneria americana* being a preferred food type (**Bengtson 1981**; **Best 1981**; **Burns Jr et al. 1997**; **Lomolino 1977**). Fish and insects forage and avoid predation within the cover of the grass beds (**Batzer and Wissinger 1996**; **Jordan et al. 1996**). Commercial and recreational fisheries, including largemouth bass, catfish, blue crabs and shrimp, are sustained by healthy SAV habitat (**Watkins 1992**). **Jordan 2000** mentioned that SAV beds in LSJRB have three times greater fish abundance and 15 times greater invertebrate abundance than do adjacent sand flats. **Sagan 2006** noted that SAV adds oxygen to the water column in the littoral zones (shallow banks), takes up nutrients that might otherwise be used by bloom-forming algae (see Section 2.4 Algal Blooms) or epiphytic alga, reduces sediment suspension, and reduces shoreline erosion.

Over the years, dredging to deepen the channel for commercial and naval shipping in Jacksonville, has led to salt-water intrusion upstream. The magnitude of this intrusion over time has not been well quantified (See Section 1.2.3 Ecological Zones). Further deepening is likely to impact salinity regimes that could be detrimental to the grass beds. This is especially important if harbor deepening were to occur in conjunction with freshwater withdrawals for the river (**SJRWMD 2012a**). On April 13, 2009, the Governing Board of the SJRWMD voted on a permit to allow Seminole County to withdraw an average of 5.5 million gallons of water a day (mgd) from the St. Johns River. Seminole County's Yankee Lake facility would eventually be able to withdraw up to 55 mgd. This initial permit from Seminole County represents the beginning of an Alternative Water Supply (AWS) program that would result in the withdrawal of water from the St. Johns and Ocklawaha Rivers (**St. Johns Riverkeeper 2009**). The impact of water withdrawal on salinity was investigated by a team of researchers from the SJRWMD, and the final recommended sustainable withdrawal from the Water Supply Impact Study was 155 MGD. The National Research Council peer review committee provided peer review, and the final report was made available in early 2012 (**NRC 2012**).

4.1.3. Data Sources & Limitations

The SJRWMD conducted year-round sampling of SAV from 1998 to 2011 at numerous stations (about 152 stations along line transects of St. Johns River (1.25 miles apart) (**Hart 2012**). This monitoring program, which included water quality data collected at some of the SAV sites, was suspended due to budget cuts, so no new data were available from 2012-2014. Sampling resumed on a more limited basis in 2015/2016 to include 56 stations from Jacksonville to Black Creek, Hallows Cove, and Federal Point. In 2017, this increased to 61 stations, 81 (2018) and 112 (2019). The increase in site sampling in 2018 and 2019 was mostly because priority 1 sites were largely devoid of grass and quick to survey. Therefore, there was extra time to survey more sites, and the uptick in sites surveyed will not likely remain as the SAVs regrow (**Trent 2020**). COVID -19 restrictions limited the ability to sample in 2020 to 44 sites north of Hallows Cove (Table 4.2). Data collection focused on continuous line-intercept data at about half of these sites annually from June to August. The intercept data were supplemented with 0.25m^2 quadrat data collected at 10 m evenly spaced intervals along the transects. Quadrat data were used to determine water depth, sediment type, species composition, SAV percent cover and average canopy height (**PSSOP 2015**). The terminology developed in the late 1990's included GT 'Ground Truthing' and LT 'Light Truthing' site names (Appendix 4.1.7.1.A-E). When the sites were first selected, the LTs were not fully surveyed in the beginning with data collected every meter, and some every 5 meters along a transect. Since 2015, all of the SAV sites are surveyed fully in the same way. The original name designations remain to keep track and compare data over time (**Trent 2020**).

Table 4.2 Summary of SAV sampling sites in LSJRB 2015-2020.

All Sites:

Year	No. bare no grasses	Not sampled	Total sampling sites	% Bare
2015	6	2	56	11
2016	7	2	56	13
2017	12		61	20
2018	16		81	20
2019	30	6	112	27
2020	11	4	44	25

Note: Sampling was reduced in 2020 due to COVID-19 restrictions. Not included above are 6 sites in Doctors Lake (2 or 33% were bare compared to 50% in 2019); Julington Creek 2 sites (1 or 50% were bare, both had SAV in 2019); and 5 sites between Hallows Cove to Shands Bridge all had SAV in 2019/2020 (Appendix 4.1.7.1.A-E).
Source: (**Trent 2020**)

This type of field sampling provides information about inter-annual relative changes in SAV by site and region. Data evaluated in this report are for the years 1989, 2000 through 2011, and 2015 through 2020. In 2019, ten additional sites were sampled in Lake George, but these were not sampled in 2020. For maps of the individual transect locations, see Appendix 4.1.7.1.A-E.

The parameters used as indicators of grass bed condition were (1) mean bed length (includes bare patches) and grass bed length (excludes bare patches), (2) total percent cover by SAV (all species), and (3) *Vallisneria* percent cover. The data were broken down into six sections of the St. Johns River as follows: (1) Fuller Warren to Buckman, (2) Buckman to Hallows Cove, (3) Hallows Cove to Federal Point, (4) Federal Point to Palatka, (5) Palatka to Mud Creek Cove, and (6) Crescent Lake and in 2019 (7) Lake George (Appendix 4.1.7.1.A-D). The most recent data have been updated in this report and includes a couple of the most intense El Niño years (1998, and 2015); the former was followed by one of the most intense drought periods (1999-2001) in Florida history. Both of these weather phenomena exaggerate the normal seasonal cycle of water input/output into the river. In addition, a series of shorter droughts occurred during 2005-2006 and 2009-2010, 2016 and 2019. In early 2017, there was an intense drought followed by intense storms (August-September) and in 2018, more storms (Appendix 4.1.7.1.E.) Normally, grass bed length on western shorelines tends to be longer than on eastern shorelines; and this is likely because of less wave action caused by the prevailing winds and broader shallower littoral edges compared to the east bank. Therefore, the shore-to-shore differences are most pronounced in Clay County-western shore sites and St. Johns County-eastern shore sites (**Dobberfuhl 2009**). For a list of grass species encountered within each section and a comparison of the variation among grass bed parameters, including canopy height and water depth, see Appendix 4.1.7.1.A-D.

Because of the importance of color and salinity, rainfall and salinity levels were examined. Rainfall data were provided by SJRWMD (**Rao et al. 1989**; **SJRWMD 2021c**) (Figure 4.1), the National Hurricane Center (**NOAA 2021a**), and the Climate Prediction Center (**NOAA 2013**) (see Appendix 4.1.7.1.E for Rainfall, Hurricanes, and El Niño). Salinity data from 1991 to 2020 were provided by the Environmental Quality Division of the COJ. Water quality parameters are measured monthly at ten stations in the mainstem of the St. Johns River at the bottom (5 m), middle (3 m), and surface (0.5 m) depths. Additional data on salinity from 1994 to 2011 came from the SJRWMD and correspond with five specific SAV monitoring sites (Appendix 4.1.7.1.F Salinity). These data are discussed further in Section 4.4 Threatened & Endangered Species. Note that "spot sampling" cannot be used to adequately match water quality parameters and grass bed parameters; because plants like *Vallisneria* integrate conditions that drive their responses. To evaluate such responses, "high-frequency" data are required (**Jacoby 2011**). Moreover, information is limited about duration and frequency of elevated salinity events in the river and how that relates to the frequency and duration of rainfall. Also, there is limited information about the ability of SAV growing in different regions of the river to tolerate varying degrees of salinity. In 2009, the SJRWMD began to conduct research to evaluate this question by transplanting tape grass from one area to other areas in the river, thus exposing it to varying degrees of salinity for varying periods of time (**Jacoby 2011**). These same concerns are echoed by the Water Science and Technology Board's review of the St. Johns River Water Supply Impact Study (**NRC 2011**, p. 5) – see a list of select findings under Section 4.1.5 Future Outlook.

4.1.4. Current Status & Trend

The status and trend were based on the significance of evaluated grass bed parameters using Kendall's Tau correlation analysis. For the period 1989, and 2000 through 2007, the section of the St. Johns River north of Palatka had varying trends in all the parameters that usually increase and decrease according to the prevailing environmental conditions. For the period 2001-2011, the data showed a declining trend in grass bed parameters – this is in spite of some recovery in grass beds condition in 2011. In addition, salinity was negatively correlated with percent total cover and the proportional percent of tape grass (Appendix 4.1.7.2.A-B). The degree to which this occurred was greater north of the Buckman Bridge compared to south of the bridge. The ability of grasses to recover from storm-related impacts depends on how robust they are in the first place (**Gurbisz et al. 2016**). As a result, recovery seems to be quicker south of Buckman Bridge than north of the bridge.

<u>North of the Buckman Bridge:</u> The mean grass bed length (includes bare patches) decreased from 139 m (1998) to 22 m (2011). Surveys were suspended due to budget cuts from 2012 to 2014. When annual sampling resumed in the area during 2015, there were 12 GT (Ground-Truthing) sampling sites and mean grass bed length had recovered to 50 m. Following 2017 drought and later hurricane Irma, the bed length decreased to 20 m (17 sites total, of which 13 or 76% represented sites without grass), but recovered in 2019 to a length of 40 m. Then in 2020, the mean bed length decreased to 15m (16 sites total, of which 11 or 69% represented sites without grass). The mean SAV bed length (excluding bare patches) declined from 111 m (1998) to 8 m (2011), and from 37 m (2015) to 4 m (2020). The total percent coverage declined from 64% (1998) to 11% (2011) and from 47% (2015) to 7% (2020). The percentage of tape grass declined over time from 71% (1998) to 14% (2011), and from 61% (2015) to less than 1% (2019). In spite of the beginnings of a new recovery after the recent storms, in general, the recoveries in the indices over time have been below past highs. In addition, species diversity decreased over time, but the predominant species is *Vallisneria americana*, then *Zannichellia palustris* and *Ruppia maritima* and there appear to be more grasses on the east bank of the river versus the west bank (see Table 1-3 in Appendix 4.1.7.1.A). In addition, anecdotal observations from manatee aerial surveys of the area in January, May, and August of 2018, May 2019, and monthly during 2020 indicated that grass bed coverage north of the Buckman Bridge (Bolles School to Buckman Bridge-east bank, and some parts from NAS JAX to Buckman-west bank) was sparse (2019), and devoid of grasses in 2020. This was most likely due to the lack of rainfall in early 2017 that resulted in increased salinity conditions in that part of the river contributing to the decline in grass bed coverage; followed by major storms from 2017-2020 leading to increased turbidity in the water that hampered recovery.

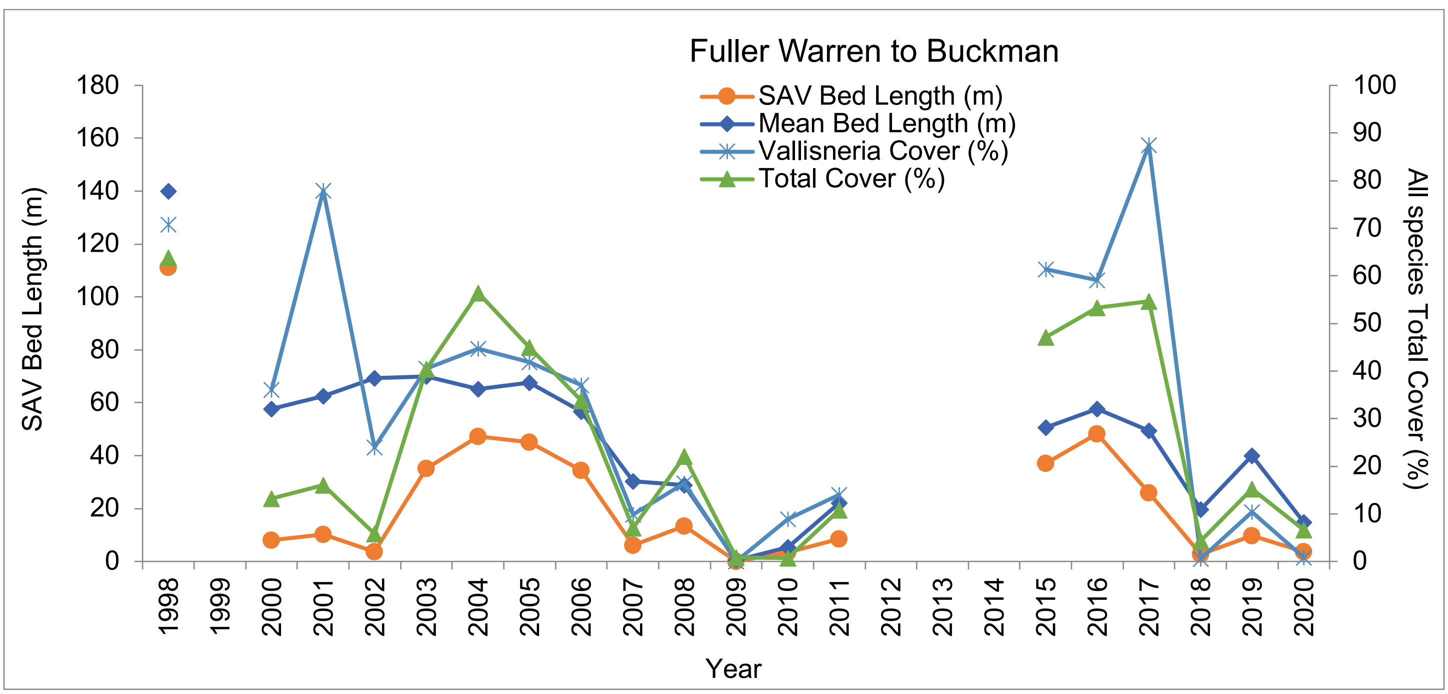


Figure 4.1 Grass bed indices for the St. Johns River in the area from Fuller Warren Bridge to Buckman Bridge. Dots represent the means (Data source ***SJRWMD 2021b****).*

South of the Buckman Bridge to Hallows Cove: Note that this analysis does not include Doctors Lake with 6 sites (2 sites or 33% without grass in 2020, 50% in 2019) or Julington Creek (1 site or 50% without grass in 2020, both with grass in 2019). The mean grass bed length (includes bare patches) decreased from 106 m (1998) to 88 m (2011), with a maximum of 146 m in 2004 when four hurricanes skirted Florida, providing above average rainfall and fresher conditions prevailed. Surveys were suspended due to budget cuts from 2012 to 2014. When annual sampling resumed in the area during 2015, there were 18 GT (Ground-Truthing) sampling sites. Mean grass bed length in 2015 recovered to 91 m, and in 2020, it was 47 m (28 sites total of which 1 or 4% represented of sites with no grass). The mean SAV bed length (excluding bare patches) increased from 64 m (1998) to 73 m (2011) and decreased from 74 m (2015) to 42 m (2019), then increased to 60 m in 2020. The total percent coverage decreased from 61% (1998) to 58% (2011) and from 83% (2015) to 80% (2020). The percentage of tape grass declined over time from 81% (1998) to 73% (2011), and from 67% (2015) to 21% (2019), then increased to 57% in 2020. In spite of the beginnings of a new recovery after the recent storms, in general, the recent recoveries in the indices over time have been below past highs. In addition, species diversity decreased over time, but the predominant species is *Vallisneria americana* and then *Ruppia maritima.* In addition, there are more grasses on the west bank of the river versus the east bank (see Table 4-6 in Appendix 4.1.7.1.A). Moreover, anecdotal observations from manatee aerial surveys of the area in January, May, August 2018, May 2019, and monthly during 2020 indicated reduced grass bed coverage south of the Buckman Bridge (to Black Creek-east bank, and Switzerland-west bank). This area still supports relatively more grass beds compared to the mostly bare north likely due to lower salinity and generally fresher conditions prevailing. Reduced coverage was likely due to decreased water clarity from increased storm activity in late 2017 to 2020.

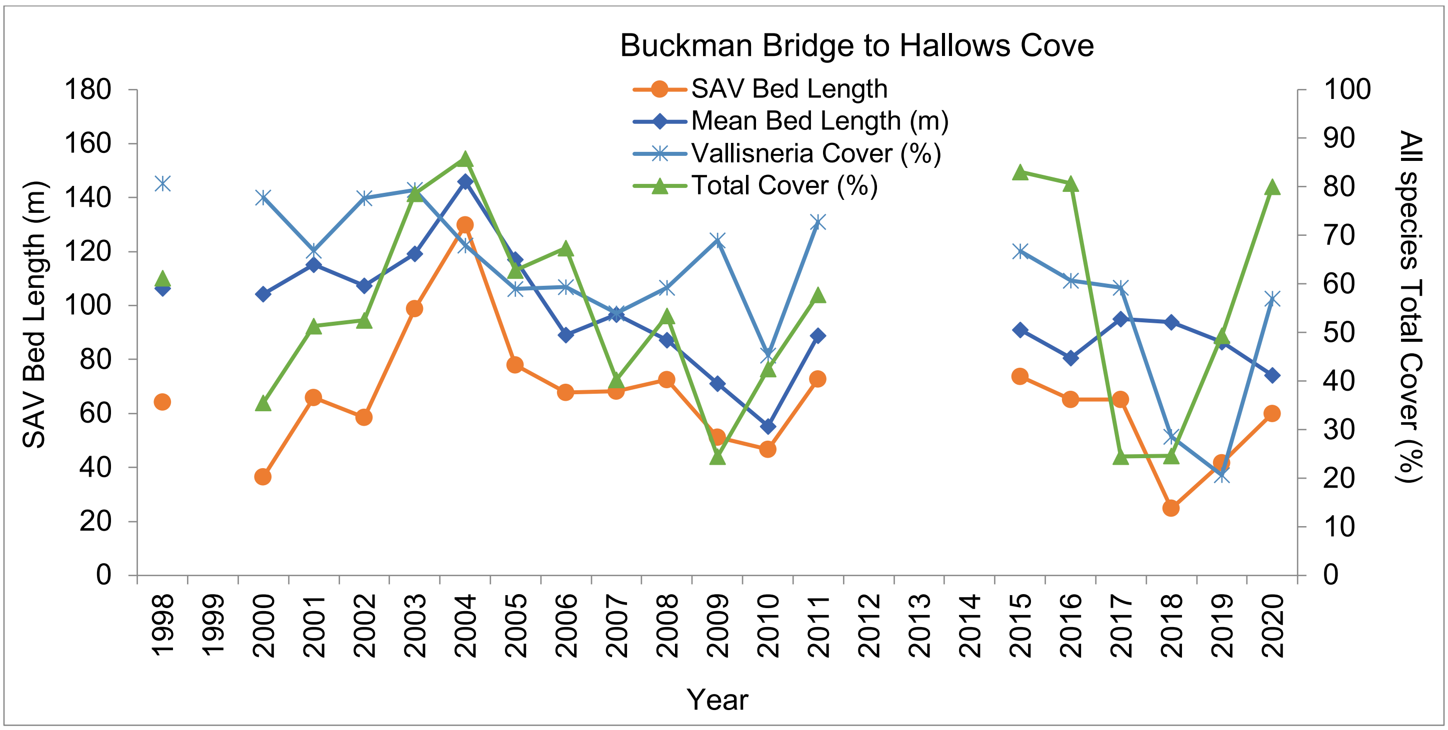


*Figure 4.2 Grass bed indices for the St. Johns River in the area from Buckman Bridge to Hallows Point.Dots represent the means (Data source **SJRWMD 2021b**)*

South of the Hallows Cove to Federal Point: This section of the river was not analyzed in 2020 due to few sites that were in the northern most reaches of this section (5 sites south of Hallows Cove to Shands Bridge). This section of the river generally tends to show less decline than adjacent sections of the river and more speedy recoveries. The mean grass bed length (includes bare patches) increased from 78 m (1998) to 80 m (2011) and there were 13 GT (Ground-Truthing) sampling sites. Surveys were suspended due to budget cuts from 2012 to 2015. When annual sampling resumed in the area during 2018, there were 28 sampling sites (11 GT + 17 LT, none bare) and this increased to 34 in 2019 (13 GT + 21 LT, with 2 sites or 6% without vegetation). Mean grass bed length decreased from historic levels to 24 m in 2018, and in 2019, it increased to 49 m. The mean SAV bed length (excluding bare patches) increased from 46 m (1998) to 61 m (2011). In 2018, it was 55 m, but increased to 67 m in 2019. The total percent coverage increased from 58% (1998) to 80% (2011) and from 24% (2018) to 48% (2019). The percentage of tape grass was relatively stable over time with 64% (1998) and 62% (2011). In 2018, it was 55%, but increased to 67% (2019). In spite of recovery after the recent storms, in general, the bed length has decreased; however, the amount of SAV has remained relatively stable. Total percent coverage has decreased, but the proportion of tape grass has remained stable, if not slightly increased. In addition, although this section has the most species diversity of all the sections sampled, species diversity has decreased over time. Nevertheless, the predominant species that have increased over time are *Vallisneria americana* mostly, followed by *Chara spp., and Eleocharis spp.* The species that have decreased in abundance over time included *Ruppia maritima, Najas guadalupensis*, *Sagittaria subulata*, *Zannichellia palustris*, *Ceratophyllum demersum*, and *Hydrilla verticillata.* In addition, there are more grasses on the west bank of the river versus the east bank (see Table 1-3 in Appendix 4.1.7.2.B).

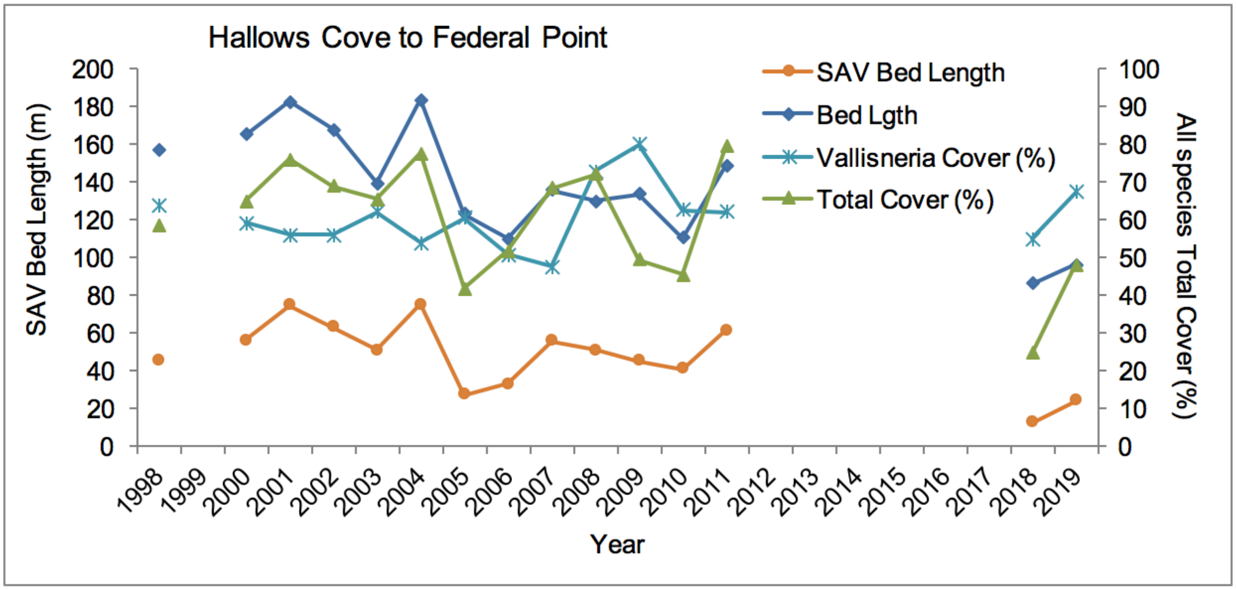


*Figure 4.3 Grass bed indices for the St. Johns River in the area from Hallows Point to Federal Point. Dots represent the means (Data source **SJRWMD 2021b**).*

<u>South of the Federal Point to Palatka:</u> This section of the river was not sampled in 2020 due to COVID-19 restrictions and is not included in the high priority 1 sites (as previously mentioned, these sites were able to be collected in 2019 because of reduced SAV coverage in priority 1 sites). The mean grass bed length (includes bare patches) decreased from 43 m (1998) to 22 m (2011) but had a high of 60 m in 2004. There were 6-8 GT (Ground-Truthing) sampling sites during this period. Surveys were suspended due to budget cuts from 2012 to 2015. In addition to GT and LT site names, some SAV sites were named 'H' or Historic sites in the past. When annual sampling resumed in the area during 2019, there were 15 sampling sites (5GT + 9 LT + 1 H, with 6 or 40% without grass). Mean grass bed length decreased from historic levels to 26 m. The mean SAV bed length (excluding bare patches) decreased from 25 m (1998) to 15 m (2011), with a high of 37 m in 2004. Then decreased to 6 m in 2019. The total percent coverage remained stable from 53% (1998) to 54% (2011) and decreased to 20% (2019). The percentage of tape grass was relatively stable over time with 62% (1998) and decreasing to 39% (2011). In 2019, it was 65%. In spite of recovery after the recent storms, in general, the bed length has decreased; the amount of SAV has decreased. Total percent coverage has decreased, but the proportional percent of tape grass has remained stable. In addition, species diversity has decreased over time. Currently, the dominant species is *Vallisneria americana* mostly, followed by small amounts of *Eleocharis spp.*, *Najas guadalupensis*, and *Sagittaria subulata.* Prior to 2007, there were up to eleven different species with *Vallisneria* and *Najas* dominating (see Table 1-3 in Appendix 4.1.7.2.C).

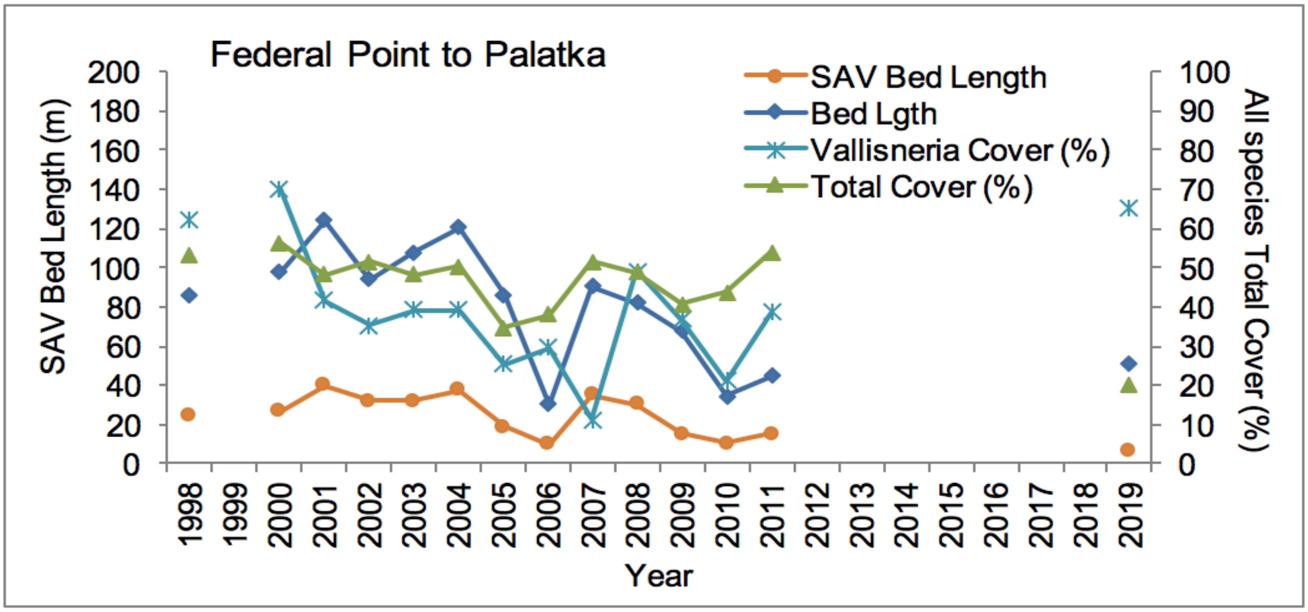


*Figure 4.4 Grass bed indices for the St. Johns River in the area from Federal Point to Palatka. Dots represent the means (Data source **SJRWMD 2021b**).*

<u>South of the Palatka to Mud Cove Creek:</u> This section of the river was not sampled in 2020 due to COVID-19 restrictions and is not included in the high priority 1 sites (as previously mentioned, these sites were able to be collected in 2019 because of reduced SAV coverage in priority 1 sites). The river is narrower here, and flow significantly affects water clarity. The mean grass bed length (includes bare patches) decreased from 17 m (2000) to 8 m (2011) but decreased to 4 m in 2004. There were 18-20 GT (Ground-Truthing) sampling sites during this period. Surveys were suspended due to budget cuts from 2012 to 2015. When annual sampling resumed in the area during 2019, there were 3 GT sampling sites (all without grass). Mean grass bed length decreased from historic levels to 0 m. The mean SAV bed length (excluding bare patches) decreased from 10 m (1998) to 6 m (2011), 0.75 m in 2004, then decreased to 0 m in 2019. The total percent coverage decreased from 55% (1998) to 39% (2011), 7% (2004) and was 0% (2019). The percentage of tape grass decreased from 67% (1998) to 58% (2011) and was 0% (2019). Prior to 2007, there were up to eight different species with *Vallisneria americana* mostly, followed by *Najas guadalupensis*, *Sagittaria subulata*, *Zannichellia palustris*, and *Ceratophyllum demersum* dominating (see Tables 4-6 in Appendix 4.1.7.2.C).

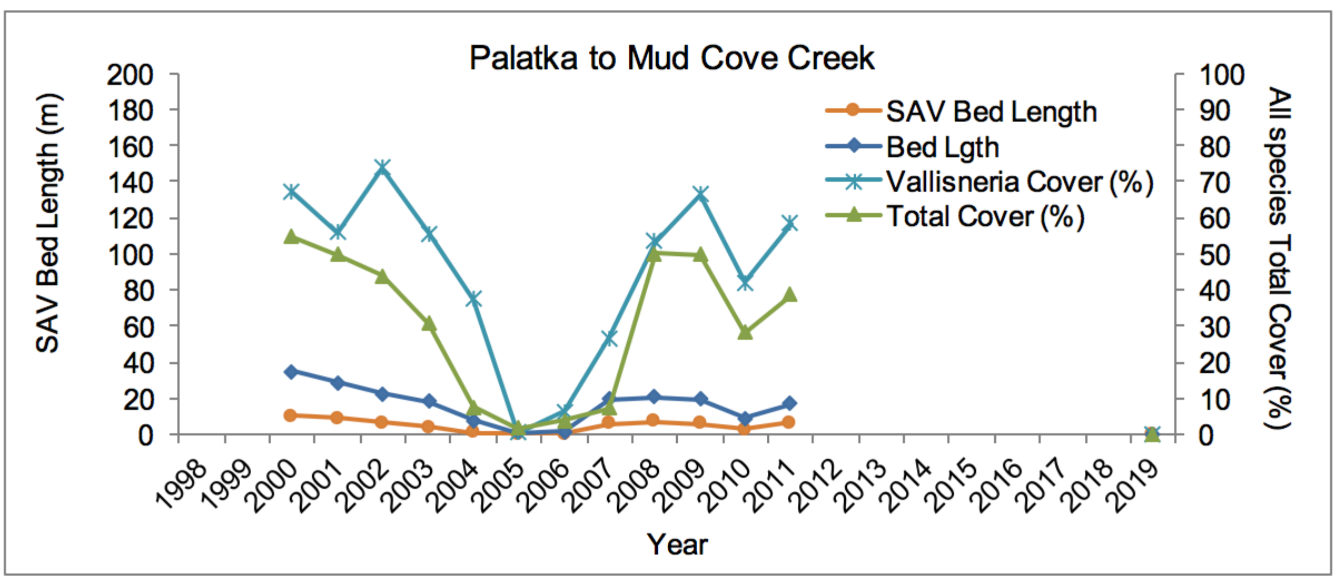


*Figure 4.5 Grass bed indices for St. Johns River in the area from Palatka to Mud Cove Creek. Dots represent the means, 2019 represented by three sites all were devoid of grass (Data source **SJRWMD 2021b**).*

Crescent Lake: This location was not sampled in 2020 due to COVID-19 restrictions and is not included in the high priority 1 sites (as previously mentioned, these sites were able to be collected in 2019 because of reduced SAV coverage in priority 1 sites). This system tends to be highly variable, with no grasses in 2019. The mean grass bed length (includes bare patches) increased from 33 m (2001) to 62 m (2011) but was from 0-16 m (2003-2006). There were 4 GT (Ground-Truthing) sampling sites during this period. Surveys were suspended due to budget cuts from 2012 to 2018. When annual sampling resumed in the area during 2019, there were 5 sampling sites (2 GT + 1 H + 2 LT, all without grass). Mean grass bed length decreased from historic levels to 0 m. The mean SAV bed length (excluding bare patches) increased from 26 m (2001) to 48 m (2011), 0-2 m (2003-2006), and was 0 m in 2019. The total percent coverage was relatively stable from 51% (1998) to 55% (2011), 0-6% (2003-2006) and 0% (2019). The proportional percentage of tape grass increased from 41% (2001) to 100% (2011), 0-2% (2003-2006), and was 0% in 2019. Prior to 2007, there were up to four different species with *Vallisneria americana* mostly, followed by *Najas guadalupensis*, *Hydrilla verticillata,* and *Chara spp.* dominating (see Table 1-3 in Appendix 4.1.7.2.D).

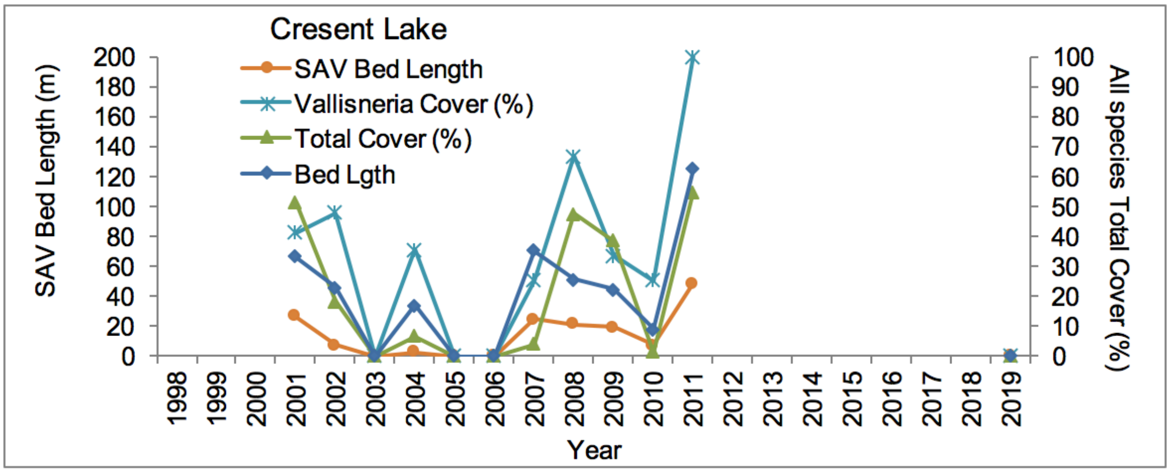


Figure 4.6 Grass bed indices for Crescent Lake. Dots represent the means, 2019 represented by five sites all were devoid of grass (Data source ***SJRWMD 2021b****).*

Lake George: This location was not sampled in 2020 due to COVID-19 restrictions and is not included in the high priority 1 sites (as previously mentioned, these sites were able to be collected in 2019 because of reduced SAV coverage in priority 1 sites). The District began a new round of sampling in Lake George in 2019 consisting of 9 HS sampling stations (HS – new site terminology from Lake George). The mean grass bed length (includes bare patches) was 63 m (2019); the mean SAV bed length (excluding bare patches) was 28 m (2019); the total percent coverage was 37% (2019); and the proportional percentage of tape grass was 51% (2019). There were five different species with *Vallisneria americana* mostly, followed by *Chara spp., Ruppia maritima, Sagittaria subulata, and Najas guadalupensis* dominating. Furthermore, grass beds are more substantial on the west bank compared to the east bank, but the east bank has about double the proportional percent of tape grass coverage (see Tables 1-3 in Appendix 4.1.7.1.E).

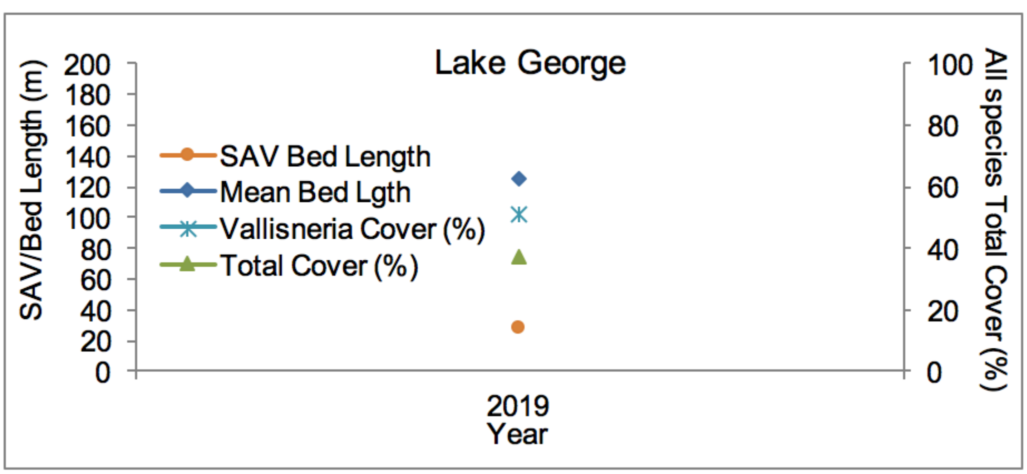


Figure 4.7 Grass bed indices for Lake George. Dots represent the means, 2019 represented by nine sites (Data source ***SJRWMD 2021b****).*

Although still below 1998 levels, the 2015 to 2018/2019 data from SJRWMD indicate that grass beds in the northern section of St. Johns River recovered some compared to 2011 levels because of more fresh conditions (Figure 4.1). Drought in early 2017 caused salinity to increase, and then storms in late 2017, 2018, 2019, and 2020 caused poor water clarity, negatively influencing the grass bed regrowth. Reports from residents along the river described severe mechanical ripping of SAVs, and plants washing up on adjacent properties (especially after Irma). This sudden loss of SAV may also be a reason for the current slow recovery (**Trent 2021**).

However, it is important to note that these data are limited, and more years of data are required to see how well the grasses will recover from what was an anomalous weather pattern in the last three years and reduced sampling effort due to COVID -19 restrictions. The grass beds from the Buckman Bridge to Hallows Cove and Federal Point appear to have undergone significant changes in the last three years compared to 2011. The 2019 data indicated that the grasses were regenerating again in spite of a limited number of sample sites (5 in 2020). There was a declining trend in all the parameters (2001-2007) south of Palatka and in Crescent Lake. Moreover, from 2007-2009, the data suggested an increasing trend in all parameters. In 2010, data showed a declining trend, but in 2011 the trend was increasing again. However, over the longer-term (2001-2011, and 2020) there was a declining trend in grass bed length (Appendix 4.1.7.2.D-E). There were no new data for these areas of the river in 2015-2018, and 2020.

The availability of tape grass decreased significantly in the LSJRB during 2000-2001. This may be because the severe drought during this time caused higher than usual salinity values which contributed to as much as 80% mortality of grasses (**Morris and Dobberfuhl 2012**). Factors that can adversely affect the grasses include excess turbidity, nutrients, and phytoplankton (see Section 2.5 Algae Blooms). In 2003, environmental conditions returned to a more normal rainfall pattern. As a result, lower salinity values favored tape grass growth. In 2004, salinities were initially higher than in 2003 but decreased significantly after August with the arrival of heavy rainfall associated with four hurricanes that skirted Florida (Hurricanes Charley, Francis, Ivan, and Jeanne). Grass beds north of the Buckman Bridge regenerated from 2002-2006 and then declined again by as much as 50% (**Morris and Dobberfuhl 2012**) in 2007 due to the onset of renewed drought conditions (**White and Pinto 2006b**). Drought conditions ensued from 2009-2010, leading to a further decline in the grass beds. From 2012-2015, rainfall was normal and stable, favoring grass bed growth again in the northern sections of the river. Under normal conditions, SAV in the river south of Palatka and Crescent Lake is dynamic (highly variable) and significantly influenced by rainfall, runoff, and water color (**Dobberfuhl 2009**). The 2017 year was unusual in that a severe drought occurred early in the year, which adversely affected the grass beds. Then in September, major Hurricane Irma and, in 2018, another series of storms, including another major Hurricane Michael, significantly affected the State of Florida. Massive amounts of freshwater input to the river resulted, likely reduced water clarity for many months and preventing grass beds from recovering. Taking everything into account, the current **STATUS** of SAV is ***Unsatisfactory***, and the **TREND** is ***Uncertain***.

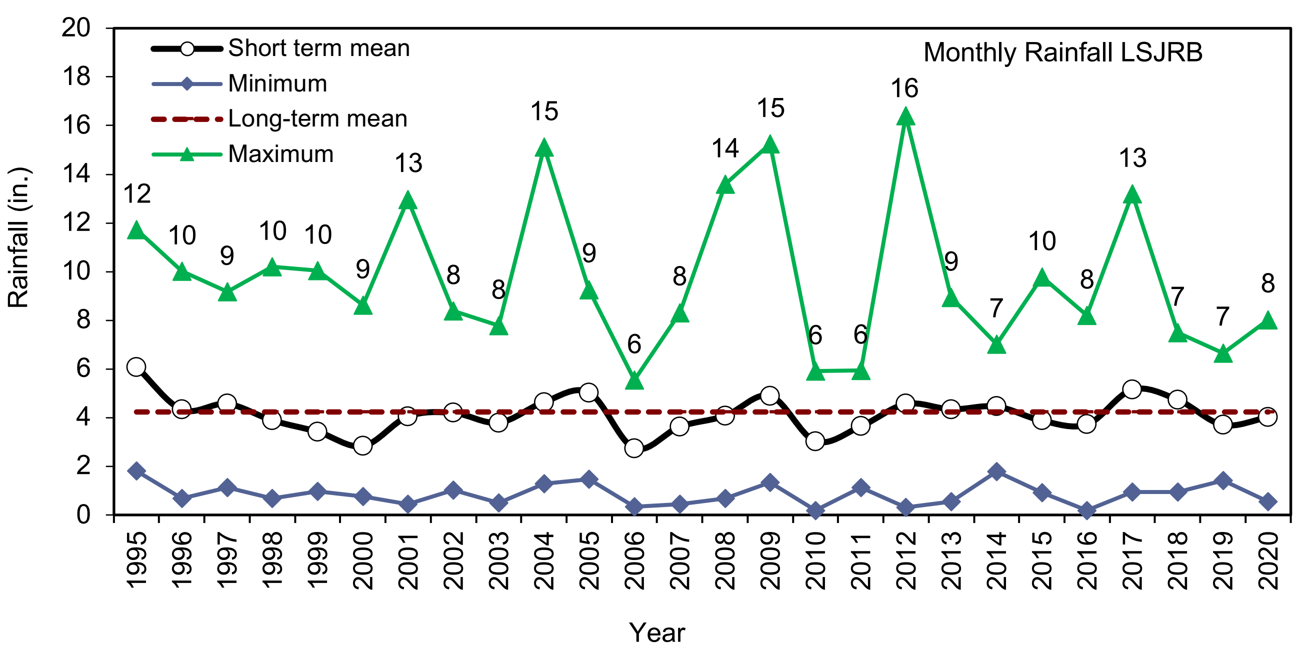


Figure 4.8 Monthly rainfall maximum, minimum, long-term and short-term annual means for LSJRB. Data are for the period June 1995 to December 2020 (solid lines). Average of monthly rainfall for periods 1951-1960 and 1995-2020 were not significantly different (dotted line) (Data source: ***SJRWMD 2021c****).*

Continuation of long-term monitoring of SAV is essential to detect changes over time. Grass bed indices, along with water quality parameters, should be used to determine the current state of health. They can then be used to identify restoration goals of the SAV habitat, which will preserve and protect the wildlife and people who rely on the habitat for either food, shelter and their livelihood. Further indices of the health and status of grass beds should be developed that express the economic value of the resource as it pertains to habitat ecosystem services, fisheries and other quality-of-life indices such as aesthetics, recreation, and public health. Maintaining water clarity is essential to the survival of grasses particularly from storms (**Linhoss et al. 2015**), but also persistent algae blooms. The grass beds monitoring should continue to be expanded especially in light of efforts to further deepen the port channel, and the pending environmental and habitat changes that are likely to ensue because of global warming, rising sea levels, El Niño events, and storms.

Learning more about SAV response to drought and/or periods of reduced flow can provide crucial understanding as to how water withdrawals (including broader water supply policy), dredging, and the issue of future sea level rise will affect the health of the ecosystem by adversely altering salinity profiles.

Freshwater withdrawals, in addition to harbor deepening, have likely contribute to the changes in salinity regimes in the LSJRB over time, but the size of the most recent impacts are predicted to be minimal based on the 2012 Water Supply Impact Study (**SJRWMD 2012b**). The study found that the maximum sustainable upstream surface water withdrawal, and extent of impact to SAV in the LSJR was to be negligible relative to the normal inter-annual variation in the primary drivers of SAV colonization, water color, and salinity intrusion, which in turn are driven by precipitation and runoff. If a sufficient change in salinity regimes occurs, it is likely to cause a die-off of the grass bed food resources for the manatee. This result would decrease carrying capacity of the environment's ability to support manatees (**Mulamba et al. 2019**). As a result, the cumulative effects of freshwater withdrawals on these and other flora and fauna should be monitored to assess the impacts of water supply policy (**NRC 2012**).

Select findings of the St. Johns River Water Supply Impact Study: Final Report (**NRC 2012**):

- "The workgroups did not appear to consider the possibility of "back-to-back extreme events in their analyses, e.g., two or three years of extreme drought in a row, which the Committee considers to be reasonably likely future situations." p. 97
- "They also tended to present mean responses to perturbations of a given driver with little or no consideration of the variance in that response. Although mean values are considered the most likely responses from a statistical perspective, in analyzing potential environmental impacts of changes in driver variables it is important to consider ranges (or variances) of responses. Although such responses may be less likely than mean values, they may not have negligible probabilities and they also could be much more detrimental than the mean responses. The Committee remains concerned that the District did not consider such conditions sufficiently in their otherwise thorough analyses." p. 97
- "Several critical issues that are beyond the control of the District or were considered to be outside the boundaries of the WSIS limit the robustness of the conclusions. These issues include future sea-level rises and increased stormwater runoff and changes in surface water quality engendered by future population growth and land-use changes. As discussed in Chapter 2, the predicted effects of some of these issues on water levels and flows in the river are greater in magnitude than the effects of the proposed surface water withdrawals, but they have high uncertainties. In addition, the relatively short period (ten years) of the rainfall record used for the hydraulic and hydrodynamic modeling and the assumption that it will apply to future climatic conditions is a concern. The Committee recognizes that changing climatic conditions globally are rendering long-term historic records less and less useful in making extrapolations to future rainfall patterns, particularly for time periods in the more distant future (e.g., 25-50 years from now). The District should acknowledge this limitation in its final report and should plan to run its models with more recent rainfall records in an adaptive management mode." p. 100-101
- "The Committee continues to be somewhat concerned with the basis for the final conclusion that water withdrawals of the magnitude considered in the WSIS will not have many deleterious ecological effects. In large part, this conclusion was based on the model findings that increased flows from the upper basin projects and from changes in land use (increases in impervious urban/suburban areas) largely compensated for the impacts of water withdrawals on water flows and levels. Although the upper basin projects should be viewed as a positive influence insofar as they will return land to the basin (and water to the river) that belonged there under natural conditions, the same cannot be

said about increased surface runoff from impervious urban- and suburbanization. The generally poor quality of surface runoff from such land uses is well known. Uncertainties about future conditions over which the District has no control (e.g., climate change, sea level rise, land use) also lead to concerns about the reliability of the conclusions." p. 100-101

- "The WSIS should have included a water quality workgroup that addressed the effects of changing land use on runoff and return flow water quality throughout the basin. It is clear that future needs for additional water supplies in the St. Johns River basin will be driven by population increases that also will result in land-use changes—essentially increases in urban/suburban land cover—and increases in the production of wastewater effluent. Both of these changes are highly likely to affect surface water quality in the basin. The District argued that these considerations were beyond their scope and authority and that existing regulations such as NPDES permits and stormwater regulations would be sufficient to prevent water quality degradation. Although the Committee accepts the District's argument that it lacks authority to control land use and population growth, it does not accept the view that this means the District has no responsibility to consider these issues in a study on the environmental impacts of surface water withdrawals." p. 104
- "District scientists found that the lack of basic data (e.g., certain kinds of benthos and fish information) and the inadequacy of basic analytical tools (e.g., on wetland hydrology and biogeochemical processes) limited what they were able to achieve and conclude. Some of these deficiencies could be overcome by future work of District scientists, and these needs should be addressed in the District's medium- and long-term planning for future studies." p. 104

4.2. Wetlands

Figure 4.9 A variety of weltands can be found along the St. Johns River Basin, including marshes in the brackish, tidal coastal areas (left), and cypress-lined, freshwater river swamps to the south of Jacksonville, Florida (right) (Photos: Heather P. McCarthy).

4.2.1. Description

Some of the most biologically diverse and productive systems on earth, wetlands are partially or periodically inundated with water during all or part of the year (**Myers and Ewel 1990**). The term *wetland* is broadly used to describe an area that is transitional between aquatic and terrestrial ecosystems. Within the LSJRB, these ecosystems include both coastal and freshwater wetlands. ***Coastal wetlands*** include all wetlands that are influenced by the tides within the St. Johns River watershed as it drains into the Atlantic Ocean (**Stedman and Dahl 2008**). The term *wetland* also includes non-vegetated areas like tidal sand or mud flats, intertidal zones along shorelines, intermittent ponds and oyster bars. ***Freshwater wetlands*** are typically inland, landlocked or further upstream in the Middle and Upper Basins of the St. Johns River. Wetland ecosystems described in this section are typically broken down into vegetation types based on physiognomy, or growth form of the most dominant plants: 1) forested wetlands and 2) non-forested wetlands. ***Forested wetlands*** are usually freshwater and include swampy areas that are dominated by either hardwood or coniferous trees. ***Non-forested wetlands*** can be marine, estuarine or freshwater, and include areas that are dominated by soft-stemmed grasses, rushes and sedges. Non-forested wetlands include wet prairies and mixed scrub-shrub wetlands dominated by willow and wax myrtle. The

SJR represents, in Florida, one of the rivers with the highest headwater to stream length ratios, with 5.3 headwaters per km of river and a total of 886 headwaters (**White and Crisman 2016**). Headwater wetlands are associated with grassland/prairie, hardwood forest, and pine flatwood habitats (**White and Crisman 2016**).

4.2.2. Significance

Wetlands perform a number of crucial ecosystem functions including assimilation of nutrients and pollutants from upland sources. The estimated nitrogen removal of 187,765 Mt per year by SJR wetlands is valued at >$400 million per year for nitrogen and the estimated phosphorous removal of 2,390 Mt per year is valued at >$500 million per year (**Craft et al. 2015**). **Widney et al. 2018** estimated that forested swamps (2340 km^2), cypress forests (1311 km^2), and inland freshwater marshes (2572 km^2) in the SJR watershed removes 80,000 MT of nitrogen and more than 2000 MT of phosphorous by burial and provides a minimum savings of $240 million and $17 million to removed nitrogen and phosphorous, respectively, using estimated costs from wastewater treatment facilities. Additionally, wetlands can help to minimize local flooding, and, thereby, reduce property loss (**Brody et al. 2007**). Basins with as little as 5% lake and wetland areas may have 40-60% lower flood peaks than comparable basins without such hydrologic features (**Novitski 1985**). In Florida between 1991 and 2003, 48% of permits issued were within the 100-year floodplain, suggesting potential costs for recovery (**Brody et al. 2008**). Wetlands also provide nursery grounds for many commercially and recreationally important fish; refuge, nesting, and forage areas for migratory birds; shoreline stabilization; and critical habitat for a wide variety of aquatic and terrestrial wildlife (**Groom et al. 2006**; **Mitsch and Gosselink 2000**).

4.2.3. The Science and Policy of Wetlands in the US: The Past, the Present, and the Future

Since the 1970s when wetlands were recognized as *valuable* resources, accurately describing wetland resources and successfully mitigating for the destruction of wetlands have been ongoing pursuits in this country. During the last few decades wetland science and policy have been driven by a) calculating wetland loss, and b) determining how to compensate for the loss. The result has been adaptive management and evolving regulations.

Wetland mitigation was not initially a part of the Section 404 permitting program as outlined in the original 1972 Clean Water Act, but “was adapted from 1978 regulations issued by the Council on Environmental Quality as a way of replacing the functions of filled wetlands where permit denials were unlikely” (**Hough and Robertson 2009**). However, it was not until 1990 that the USACE and EPA defined mitigation. It was defined as a three-part, sequential process: 1) permit-seekers should first try to ***avoid*** wetlands; 2) if wetlands cannot be avoided, then permit-seekers should try to ***minimize*** impacts; and 3) if wetland impacts cannot be avoided or minimized, then permit-seekers must ***compensate*** for the losses.

4.2.3.1. The Past: A Focus on Wetland Acreage

During the 1980s-1990s, assessments of wetland losses (and the mitigation required as compensation) typically focused on ***acres*** of wetlands. In 1988, President G.H. Bush pledged “no net-loss” of wetlands. This pledge was perpetuated by President Clinton in 1992, and President G.W. Bush in 2002 (**Salzman and Ruhl 2005**). To ascertain whether this goal was being achieved or not, the USFWS was mandated to produce status and trends reports using the National Wetlands Inventory data. In 1983, the first report, *Status and Trends of Wetlands and Deepwater Habitats in the Conterminous United States, 1950s to 1970s,* calculated a net annual loss of wetlands during this time period equivalent to 458,000 acres per year (**Frayer et al. 1983**). In 1991, the second report, *Status and Trends of Wetlands in the Conterminous United States, mid-1970s to mid-1980s,* reported a decline in the rate of loss to 290,000 acres per year (**Dahl and Johnson 1991**). In 2000, the USFWS released the third report, *Status and Trends of Wetlands in the Conterminous United States 1986 to 1997,* which concluded the net annual loss of wetlands had further declined to 58,500 acres per year (**Dahl 2000**).

4.2.3.2. The Present: A Focus on Wetland Functions

In 2006, the fourth report by the USFWS, *Status and Trends of Wetlands in the Conterminous United States 1998 to 2004,* calculated for the first time a *net gain* of wetlands in the US equivalent to 32,000 acres per year (**Dahl 2006**). This result was publicized, celebrated, scrutinized, and criticized.

The central shortfall of the USFWS analyses was that wetland functions were not considered. This shortfall was briefly addressed in a footnote in the middle of the 112-page report: “One of the most important objectives of this study was to monitor gains and losses of all wetland areas. The concept that certain kinds of wetlands with certain functions (e.g., human-constructed ponds on a golf course) should have been excluded was rejected. To discriminate based on qualitative considerations would have required a much larger and more intensive qualitative assessment. The data presented do not address functional replacement with loss or gain of wetland area" (**Dahl 2006**). The results of the 2006 report solidified the acceptance among scientists and policymakers that the simplistic addition and subtraction of wetland acres do not produce a wholly accurate portrayal of the status of wetlands. In short, any comprehensive evaluation of the status of wetlands needs to include a thorough consideration of what types of wetlands are being lost or gained and the ecosystem functions those wetlands provide.

Toward this end, publications began to emphasize that the USFWS's reported net gain of wetlands in the US must be viewed alongside some important caveats and exceptions (**CEQ 2008**). For instance, some important types of wetlands were declining, although the overall net gain was positive. In 2008, USFWS and NOAA released an influential report entitled *Status and Trends of Wetlands in the Coastal Watersheds of the Eastern United States 1998-2004* (**Stedman and Dahl 2008**). This report calculated an annual loss of coastal wetlands at a rate of 59,000 acres per year (prior to Hurricanes Katrina and Rita in 2005). The report states: "The fact that coastal watersheds were losing wetlands despite the national trend of net gains points to the need for more research on the natural and human forces behind these trends and to an expanded effort on conservation of wetlands in these coastal areas" (**CEQ 2008**). The report emphasizes the important functions of coastal wetlands and the need for more detailed tracking of wetland gains and losses.

The positive trends reported in the earlier report did not persist. The *Status and Trends of Wetlands in the Coastal Watersheds of the Conterminous United States 2004 to 2009* states: "Wetland losses in coastal watersheds have continued to outdistance wetland gains, by an estimated 360,720 acres between 2004 and 2009 due primarily to silviculture and development. **This rate of loss increased by 25 percent since the previous reporting period of 1998 to 2004**" (**Dahl and Stedman 2013**).

4.2.3.3. The Present: A Focus on Wetland Mitigation Banking

The last decade has also been marked by the growing popularity of ***wetland mitigation banking***. To offset the impacts of lost wetlands caused by a permitted activity, the SJRWMD or USACE (with the consent of DEP) may allow a permit-holder to purchase compensatory mitigation credits from an approved mitigation bank per the Compensatory Mitigation Rule (USACE, 2008a). Wetland mitigation banks are designed to compensate for unavoidable impacts to wetlands that occur as a result of federal or state permitting processes (**NRC 2001**). By 2008, it was reported that mitigation banking accounted for >30% of all regulatory mitigation arising from the Section 404 permitting process (**Ruhl, et al. 2008**). This is not a surprise as the USACE actively supports the use of mitigation banks: "Mitigation banks are a "performance-based" form of wetland and stream replacement because, unlike in-lieu fee mitigation and permittee-responsible mitigation, the tradable aquatic resource restoration credits generated by banks are tied to demonstrated achievement of project goals. Thus, the rule establishes a preference for the use of credits from mitigation banks when appropriate credits are available" (**USACE 2008**). A maximum number of *potential* credits are available for purchasable mitigation banks, provided that each mitigation bank has existing documents for its milestones met in the scheduled restoration, enhancement, preservation, and/or creation plan (**SJRWMD 2010c**). Credits are *release*d as criteria for ecological performance are met, and these newly released credits are *withdrawn* from the currently *available* credits as they are sold to permit applicants (Table 4.3, **SJRWMD 2010c**).

Although more successful than previous approaches, mitigation banking has its own set of inherent problems and inadequacies. As **Salzman and Ruhl 2005** explain, "different types of wetlands may be exchanged for one another; wetlands in different watersheds might be exchanged; and wetlands might be lost and restored in different time frames." According to **Salzman and Ruhl 2005**, "Despite all its potential shortcomings, wetland mitigation banks certainly remain popular. Credits in Florida are now trading anywhere from $30,000-$80,000 per acre. There clearly is demand and banks are still being created to supply it." Of course, the price that a permit-holder pays per mitigation credit varies by bank and time.

For example, in October 2007, SJRWMD approved the Florida Department of Transportation (FDOT) to purchase 55 mitigation bank credits from the East Central Florida Mitigation Bank at a purchase price of $32,000 per credit with up to ten additional credits for $38,000 each for unexpected impacts (**SJRWMD 2007b**). The Mitigation Banking Group, which manages 16 mitigation banks in Florida, encourages mitigation banking: "developer using a mitigation bank will have reduced permitting time… and… On site mitigation often becomes a burden on development sites, causing a development to be planned around the mitigation. Buying credits from a mitigation bank allows the developer to maximize his usable land and put that space to its highest and best use" (**MBG 2021**). Approximately, the cost for Freshwater Forested/Palustrine Forested mitigation is on average, $120,000 per state credit (**MBG 2021**).

To facilitate mitigation banking within northeast Florida, the SJRWMD has delineated mitigation basins. In most cases, mitigation credits can only be purchased within the same mitigation basin as the permitted project where wetland loss is expected. The SJRWMD mitigation basins closely resemble, but do not exactly align with the USGS drainage basins.

Within the LSJRB, the following SJRWMD mitigation basins include: Northern St. Johns River and Northern Coastal, Tolomato River and Intracoastal Nested, Sixmile and Julington Creeks Nested, Western Etonia Lakes, St. Johns River (Welaka to Bayard), and Crescent Lake (**SJRWMD 2010c**).

The definition and use of mitigation bank service areas are explained below according to the SJRWMD (**SJRWMD 2010c**):

A mitigation bank's service area is the geographic area in which mitigation credits from the bank may be used to offset adverse impacts to wetlands and other surface waters. The service area is established in the bank's permit. The mitigation service areas of different banks may overlap. With three exceptions, mitigation credits may only be withdrawn to offset adverse impacts of projects located in the bank's mitigation service area. The following projects or activities are eligible to use a mitigation bank even if they are not completely located in the bank's mitigation service area:

- *a) Projects with adverse impacts partially located within the mitigation service area;*
- *b) Linear projects, such as roadways, transmission lines, pipelines; or*
- *c) Projects with total adverse impacts of less than one acre in size.*

Before mitigation credits for these types of projects may be used, SJRWMD must still determine that the mitigation bank will offset the adverse impacts of the project and either that:

- *a) On-site mitigation opportunities are not expected to have comparable long-term viability due to such factors as unsuitable hydrologic conditions or ecologically incompatible existing adjacent land uses; or*
- *b) Use of the mitigation bank would provide greater improvement in ecological value than on-site mitigation.*

In the LSJRB, 13 mitigation banks managed by the USACE and 13 mitigation banks managed by the SJRWMD were active in 2020 (Tables 4.2 and 4.3; **DEP 2017b**; **ERDC 2017**). Six mitigation banks listed by the ACOE are inactive, and another two are approved with no credits issued. Four mitigations managed by the SJRWMD were inactive in 2020. These mitigation banks are typically located in rural areas with palustrine habitats. Permits for new mitigation banks that are pending with the USACE include Sunnyside, Little Creek Florida, Normandy, Lake Swamp MB Expansion, Sandy Creek, and Lower St. Johns Mitigation Banks.

Further investigation is needed to determine the quality and longevity of mitigated wetlands and their ability to actually perform the ecosystem functions of the wetlands they "replace." An increasing proportion of these mitigation wetlands represent uplands/wetlands preserved on average >30 miles from project site (**Brody et al. 2008**), including many acres in wetland mitigation banks. If preserved wetlands represent already functional wetlands, then they do not replace the ecosystem services lost to development. Currently, there is no accounting of the specific locations of each impacted wetland. In addition, given the connectivity of aquifer and ground water via fracture lines, those activities that uptake water in one location may prevent the watershed from being recharged during precipitation events and exacerbate drought effects on wetland systems (**Bernardes et al. 2014**).

Restored and created wetlands generally do not reach ecosystem functioning present in reference wetlands. Based on a meta-analysis from published studies of 621 wetlands, **Moreno-Mateos et al. 2012** reported that ecosystem services were not returned with restoration efforts in either created or restored wetlands. The size of the wetland (>100 ha) recovered more quickly than smaller wetlands (0.1, 1, and 10 ha). Wetlands only reached on average 74% of biogeochemical functioning **after 100 years**. In addition, plants and vertebrate diversities in restored/created wetlands remained lower than reference wetlands after 100 years. By comparison, macroinvertebrates reached references assemblages between 5 and 10 years. In comparing different types of wetlands, riverine and tidal wetlands recovered more quickly (up to 30 years) as compared to depressional wetlands that did not reach reference conditions (**Moreno-Mateos et al. 2012**).

Wetlands at the mitigation banks are not necessarily reaching a measure of success relative to reference conditions. Difficulties in restoring wetlands may be related to past activities on the property and indirect effects due to surrounding land use. For example, land use at Loblolly, Tupelo, and Sundew mitigation banks were previously agricultural, managed pasturelands, and mixed agriculture and/or low intensity urban, respectively (**Reiss et al. 2014**). **Reiss et al. 2007** investigated success and compliance of 29 wetland mitigation banks in Florida. Barberville, Loblolly, Sundew, and Tupelo were included in their study (Tables 4.3 and 4.4). These mitigation banks did not include a target for success criteria or a reference condition (either a reference database and/or comparison sites, **Reiss et al. 2009**) to measure success (e.g., wildlife needs). With respect to exotic and nuisance cover, final success criteria for state permit requires <10% exotic and nuisance cover (except for Barberville: 5% exotic, 10% nuisance). **Reiss et al. 2007** recommend that monitoring should also encompass flora and fauna, and not just exotic and nuisance species. At the time of their study, Barberville was a 'long ways off' from final success due to pines having to be replanted. Loblolly and Tupelo had started plantings and was described as not communicating so well in providing the monitoring and management status reports. Sundew was also described as not communicating so well with reports (**Reiss et al. 2007**). **Reiss et al. 2007** argue that functional equivalency in wetland mitigation banking remains questionable without a clear method to assess ecosystem function. LDI scores within the mitigation banks indicate that wetland function may be impossible to achieve (**Reiss et al. 2014**).

Table 4.3 Wetland mitigation banks permitted by the USACE serving the LSJRB, Florida (Source: ERDC 2018). Values in parentheses indicate credits reported in 2017 River Report, if any changes were reported. For inactive mitigation banks- *(year) indicates when last active.

MITIGATION BANK NAME	ACREAGE	CREDIT TYPE	CREDIT BALANCE			
			AVAILABLE	WITHDRAWN	RELEASED	POTENTIAL
Barberville Mitigation Bank* (2010)	366	Palustrine emergent, palustrine forested	2.8 (No change)	13.1 (No change)	15.9 (No change)	63.7 (No change)
Brandy Branch* (2017)	762	Palustrine forested	12.6 (No change)	3.0 (No change)	15.6 (No change)	130.5 (No change)
Brick Road* (2018)	2,945	Palustrine emergent, palustrine forested	62.0 (No change)	2.5 (No change)	64.6 (No change)	504.0 (No change)
Farmton	24,323	Palustrine	4530.7 (4570.6)	545.9 (506.0)	5076.6 (No change)	5465.7 (No change)
Fish Tail Swamp	5,327	Palustrine forested	171.1 (157.4)	118.8 (112.5)	290.0 (269.9)	860.1 (No change)
Greens Creek	1,353	Palustrine forested	29.5 (34.4)	72.8 (67.9)	102.3 (No change)	291.9 (No change)
Highlands Ranch* (2019)	1,581	Palustrine forested	28.6 (31.4)	6.6 (3.8)	35.2 (No change)	70.4 (No change)
Lake Swamp	1,890	Palustrine	30.1 (32.1)	(106.3 104.3)	136.4 (130.6)	215.3 (No change)
Loblolly	6,240	Palustrine forested	1572.9 (1575.8)	442.0 (439.0)	2014.8 (No change)	2507.5 (No change)
Longleaf	3,021	Palustrine emergent, palustrine forested	514.0 (530.4)	512.7 (496.3)	1026.7 (No change)	1026.7 (No change)
Mill Creek (new)	2,159	Palustrine forested	0	0	0	208.9
Nochaway	3,987	Palustrine forested	0	0	0	366.0
North Florida Saltwater Marsh* (2017)	92.36	Estuarine intertidal, emergent	11.7 (No change)	0 (No change)	11.7 (No change)	49.6 (No change)
Northeast Florida Wetland	386	Palustrine	256.0 (287.1)	387.0 (355.9)	643.0 (No change)	643.0 (No change)
Peach Drive* (2015)	57.3	Palustrine forested	27.7 (No change)	20.0 (No change)	47.6 (No change)	47.6 (No change)
Star 4	950.4	Palustrine forested	113.8 (116.6)	13.4 (10.6)	127.2 (no change)	182.1 (182.5)
St. Johns	3580.0	Palustrine forested	64.1 (65.0)	0.9 (0.2)	65.0 (No change)	488.0 (No change)
St. Marks Pond	935	Palustrine forested, palustrine emergent	24.5 (22.5)	13.6 (6.8)	38.0 (29.3)	58.5 (No change)
Sundew	2,105	Palustrine emergent, palustrine forested	211.0 (224.8)	39.5 (27.7)	251.1 (250.5)	931.4
Town Branch	432	Palustrine forested	16.1 (14.6)	9.6 (6.8)	25.7 (21.3)	56.3 (No change)
Tupelo	1,525	Palustrine forested	405.9 (406.3)	217.8 (217.3)	623.6 (No change)	623.6 (No change)

Table 4.4 Wetland mitigation banks permitted by SJRWMD serving the Lower St. Johns River Basin, Florida (Source: DEP 2013f; SJRWMD 2016c; DEP 2017b; and SJRWMD 2018). Values in parentheses indicate credits reported in 2019 River Report, if any changes were reported. * indicates no change in activity.

MITIGATION BANK NAME	ACREAGE	CREDIT TYPE	CREDIT BALANCE		
			AVAILABLE	RELEASED	POTENTIAL
Brick Road			83.4	118.2	295.4
Greens Creek	4,201	Forested freshwater	9.7 (25.4)	307.7 (302.6)	405.6 (no change)
Fish Tail Swamp			33.9	169.4	322.9
Highlands Ranch Mitigation	1,575	Forested freshwater	37.9 (52.7)	103.4 (no change)	200.5 (no change)
Loblolly	6,247	Forested freshwater, general wetlands	25.4 (30.1)	1388.7 (no change)	1650.9 (no change)
Longleaf	3,020	Forested freshwater	-0.92 (0.05)	375.0 (374.9)	375.0 (395.0)
Lower St. Johns*	990	Forested freshwater	8.04 (no change)	128.7 (no change)	140.1 (no change)
Nochaway*	4,076	Forested freshwater	8.2 (no change)	183.9 (no change)	459.7 (no change)
Normandy	1,033	Forested freshwater, general wetlands	22.2 (18.4)	61.1 (43.6)	174.5 (no change)
North Florida Saltwater Marsh	93	Estuarine intertidal, emergent	3.5 (4.0)	11.9 (no change)	47.7 (no change)
Northeast Florida	774	General wetlands	4.9 (11.2)	394.9 (no change)	394.9 (no change)
Sandy Creek	504	General wetlands	21.0 (22.4)	22.4 (no change)	89.8 (no change)
St. Johns	3,579.6	Forested freshwater	26.5 (70.7)	120.1 (no change)	480.3 (no change)
St. Marks Pond	759	Forested freshwater, herbaceous freshwater	7.7 (8.1)	118.6 (107.8)	157.6 (no change)
Star 4	950	Forested freshwater	42.8 (29.8)	170.3 (153.3)	171.7 (no change)
Sundew	2,107	Forested freshwater, herbaceous freshwater	29.8 (11.8)	449.2 (424.3)	621.7 (no change)
Sunnyside*	385	Forested freshwater	18.1 (no change)	18.1 (no change)	56.8 (no change)
Town Branch	431	Forested freshwater	10.4 (5.8)	47.7 (43.1)	64.2 (no change)
Tupelo*	1,524	General wetlands	7.6 (no change)	459.7 (no change)	459.7 (no change)

4.2.3.4. The Future: A Focus on Wetland Services

Wetland policies now focus on ecosystem services (**Ruhl et al. 2008**). As applied to wetlands, the *science of ecosystem functions* investigates how wetlands function as nursery grounds, shelter, or food for wildlife. The emerging *science of ecosystem services* examines how wetlands serve human populations. As explained by **Ruhl et al. 2008**, recent research documents that "wetlands can provide important services to local populations, such as air filtering, micro-climate regulation, noise reduction, rainwater drainage, pollutant treatment, and recreational and cultural values." Ecosystem services research aims to develop cost-effective methods to quantify wetland alterations. For example, wetland mitigation banking has led to a predominance of wetland banks in rural areas (**Ruhl and Salzman 2006**). In this case, the services provided by wetlands are taken from urban to rural environments. These services, like sediment capture, groundwater recharge, water filtration, and flood mitigation, have economic value associated with them. Calculating the dollar value of such services is a challenging, but not impossible, endeavor (Figure 4.10). The economic value of wetlands to retain stormwater surges or buffer shorelines was clear after Hurricanes Katrina and Rita hit the Gulf Coast of the US, where coastal wetlands have been substantially diminished (**Stedman and Dahl 2008**).

Brody et al. 2007 examined wetland permits granted by the USACE in Florida between 1997 and 2001 and determined that "one wetland permit increased the average cost of each flood in Florida by $989.62."

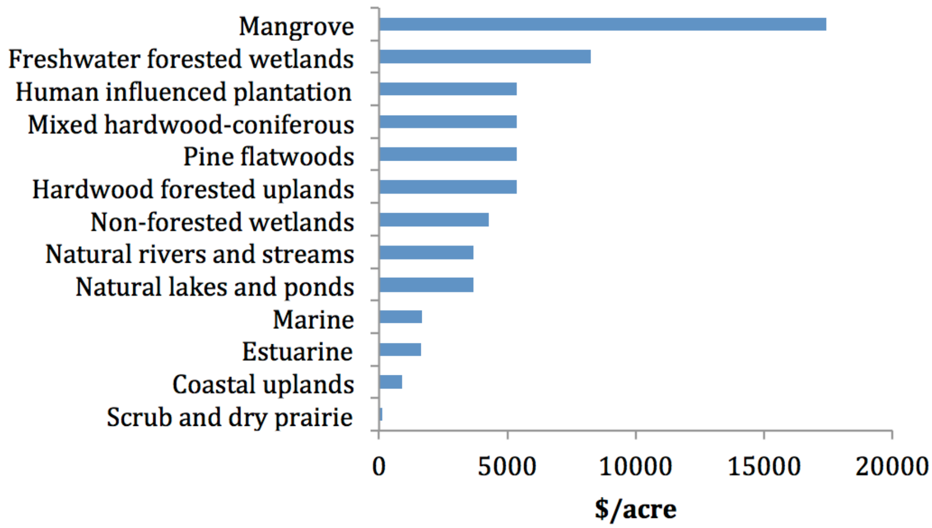


*Figure 4.10 Estimated value of ecosystem services by habitat (Source: **Brown and Shi 2014**).*

Likewise, the economic value of wetland-dependent recreation in northeast Florida is estimated in the range of $700 million per year (**Kiker and Hodges 2002**). The wetland-dependent activities with the greatest economic value to northeast Florida are recreational saltwater fishing ($301.6 million per year), followed by wildlife viewing ($226.5 million per year). Based on survey results, Florida residents and tourists value outdoor recreation (>95% of 3,961 Florida residents and 2,306 tourists participated in outdoor recreation) and specifically saltwater beach activities (63%), wildlife viewing trips (49%), and fishing (46%) (**DEP 2013g**). In Florida, 2.9 million people fished, hunted, or viewed wildlife in 2006 (**USDOI and USDOC 2008**). The number of pleasure vessels recorded in Duval, St. Johns, Clay, Putnam, and Flagler is >500,000 vessels (**SRR 2012**). Bird watchers spent an estimated $3.1 billion and fishers $4.3 billion in 2006 (**USDOI and USDOC 2008**). Canoeing and kayaking have become more popular, representing 14% of recreational activities in 2002 and 26% in 2011 (**DEP 2013g**). If these kinds of services are negatively impacted, the economic and social repercussions can be substantial.

The USACE and EPA published a landmark overhaul of US wetland regulations in April 2008 (**USACE 2008**). Not only did the rule consolidate the regulatory framework and require consideration of wetland functions, according to **Ruhl et al. 2008**, "the new rule also for the first time introduces ecosystem services into the mitigation decision-making standards, requiring that 'compensatory mitigation…should be located where it is most likely to successfully replace lost…services.'"

4.2.4. Data Sources on Wetlands in the LSJRB

4.2.4.1. Data Sources for Wetland Spatial Analyses

Ten GIS (Geographic Information System) maps that contain data on wetlands vegetation were available and analyzed. The GIS maps were created by either the Department of Interior USFWS or the SJRWMD from high-altitude aerial photographs (color infrared or black-and-white photos) with varying degrees of consideration of soil type, topographical and hydrologic features, and ground-truthing. In this analysis, each parcel of land or water was outlined and assigned a category, creating distinct polygons for which area (i.e., number of acres) can be calculated. These areas were used to calculate total wetlands and total acres within the LSJRB for each year available (Table 4.5). On average, wetland area represented 23.8% of total LSJRB acreage (Table 4.5). Wetlands in the LSJRB can be viewed using the National Wetlands Inventory: Wetlands Mapper: https://www.fws.gov/wetlands/data/Mapper.html.

Table 4.5 Comparison of wetland maps – Lower St. Johns River Basin, Florida.

GIS MAP ANALYZED	TOTAL WETLAND AREA IN LSJRB (ACRES)	TOTAL LAND & WATER AREA IN LSJRB (ACRES)
SJRWMD-corrected National Wetlands Inventory map (produced from 1971-1992 lumped data, processed by SJRWMD in 2001, 2003)	727,631	849,512 ACRES INCLUDING DEEPWATER. Non-wetland upland acres not specified in this map.
SJRWMD Wetland & Deep Water Habitats map (based on National Wetlands Reconnaissance Survey maps from 1972-1980, processed 1996 by SJRWMD, dated 2001)	870,576	3,110,209
SJRWMD Wetlands & Vegetation Inventory map (based on District's Wetlands Mapping Project 1984-2002, finished 2002, accuracy of wetland boundaries estimated at 80-95%)	441,072	2,208,172
SJRWMD Land Use/Land Cover map (based on 1973 data)	440,048	2,100,552
SJRWMD Land Use/Land Cover map (based on 1990 data)	435,662	2,605,247
SJRWMD Land Use/Land Cover map (based on 1995 data)	450,595	1,910,422
SJRWMD Land Use/Land Cover map (based on 2000 data)	444,467	1,851,447
SJRWMD Land Use/Land Cover map (based on 2004 data)	455,308	1,868,003
SJRWMD Land Use/Land Cover map (based on 2009 data)	452,315	1,903,789
SJRWMD Land Use/Land Cover map (based on 2014 data)	451,689	1,938,279

4.2.4.2. Data Sources for Wetland Permit Analyses

Within the LSJRB, there are two governmental entities that grant permits for the destruction, alteration, and mitigation of wetlands: 1) SJRWMD, and 2) US Army Corps of Engineers (USACE). The differing regulatory definitions of wetlands used by Federal and State agencies are outlined in Appendix 4.2.A. At the regional level, the SJRWMD has posted a comprehensive online database of all mitigation bank ledgers (**SJRWMD 2010c**). At the national level, the USACE and EPA have made available a single online database to track mitigation banking activities called the *Regional Internet Bank Information Tracking System (RIBITS)* (**ERDC 2015**). Concurrently, the EPA and USACE have developed a GIS-enabled database to *spatially* track and map permits and mitigation bank transactions, which will interface and complement the RIBITS database (**Ruhl et al. 2008**).

The wetland permit analysis conducted for this report reveals how the acreage of wetlands has changed over time according to the historical wetland permits granted through the SJRWMD Environmental Resource Permitting Program.

4.2.5. Limitations

4.2.5.1. Limitations of Wetland Spatial Analyses

The identification of vegetation type from an aerial photograph is an imperfect process. The metadata associated with the SJRWMD Wetlands & Vegetation Inventory map estimates the margin of error in wetlands delineation from aerial photographs to vary according to the type of vegetation being identified and range from 5-20% (**SJRWMD 2010b**). The metadata states: "The main source of positional error, in general, is due to the difficulty of delineating wetland boundaries in transitional areas. Thematic accuracy: correct differentiation of wetlands from uplands: 95%; correct differentiation of saline wetlands from freshwater or transitional wetlands: 95%; correct differentiation of forested, shrub, herbaceous, or other group forms: 90%; correct differentiation of specific types within classes: 80%. Accuracy varies for different locations, dates, and interpreters."

In addition to interpretational errors, wetland maps cannot accurately reflect wetlands habitats that vary seasonally or annually (e.g., the spatial extent of floating vegetation or cleared areas can be dramatically different depending on the day the aerial photo was taken). Aerial photographs pieced together to create wetlands maps may be of different types (high altitude vs. low altitude, color infrared, black-and-white, varying resolutions, and varying dates). Sometimes satellite imagery is used to create wetlands maps, which is considered less accurate for wetland identification (**USGS 1992**).

Analyses are further limited by inconsistencies and shortcomings in the wetland classification codes used (e.g., wetland codes used in the SJRWMD Land Use/Land Cover map of 1973 were markedly different than codes used since 1990). Additionally, wetland classification codes do not always address whether a wetland area has been diked/impounded, partially drained/ditched, excavated, or if the vegetation is dead (although the National Wetlands Inventory adds code modifiers to address the impacts of man). Further, wetland mapping classification categories often do not differentiate between natural and manmade wetlands. For example, naturally occurring freshwater ponds may be coded identically with ponds created for stormwater retention, golf courses, fishing, aesthetics, water management, or aquaculture. Some maps classify drained or farmed wetlands as uplands, while others classify them as wetlands. An unknown number of additional discrepancies may exist between maps. Lastly, most of the spatial information in wetlands maps has not been ground-truthed or verified in the field but is based on analyses of aerial photographs and other maps.

4.2.5.2. Limitations of Wetland Permit Analyses

A shortcoming of the records of wetlands impacted through regulatory permitting processes is that they do not address total wetland acres in the region. Additionally, acreages recorded as mitigated wetlands do not always represent an actual gain of new wetland acres (e.g., mitigation acres may represent preexisting wetlands in a mitigation bank or formerly existing wetland acres that are restored or enhanced). Thus, a true net change in wetlands (annually or cumulatively) cannot be calculated from permit numbers with certainty.

Further, changing environmental conditions require that field verification of mitigated wetlands occur on a regular basis over long time periods. The actual spatial extent, functional success, health of vegetation, saturation of soil, water flow, etc. of mitigated wetlands can change over time. On-ground site visits can verify that the spatial extent of anticipated wetlands impacted (as recorded on permits) equals actual wetlands impacted and confirm the ecological functionality of mitigated wetlands.

4.2.6. Current Status (UNSATISFACTORY)

Although wetlands maps do not reveal with any statistical certainty how many acres of wetlands in the LSJRB have been gained or lost over time, there are reliable historical records in the literature that estimate how many wetland acres have been lost *throughout the state of Florida* over time. A literature search was conducted to compile comparable and quantifiable estimates of historical wetland change in Florida over time. Because data occurring within just the LSJRB could not be extracted from statewide data, information for the whole state of Florida was evaluated and compiled in Appendix 4.2.B.

Prior to 1907, there were over 20 million acres of wetlands in Florida, which comprised 54.2% of the state's total surface area. By the mid-1950s, the total area of wetlands had declined to almost 15 million acres. The fastest rate of wetland destruction occurred between the 1950s and 1970s, as the total area of wetlands dropped down to 10.3 million acres. Since the mid-1970s, total wetland area in Florida appears to have risen slightly. Net increases in total statewide wetlands are attributed to increases in freshwater ponds, such as manmade ponds created for fishing, artificial water detention or retention, aesthetics, water management, and aquaculture (**Dahl 2006**). The average of all compiled wetlands data in Florida revealed that the state retained a total of 11,371,900 acres by the mid-1990s (occupying 30.3% of state's surface area). This translates into a cumulative net loss of an estimated 8,940,607 acres of wetlands in Florida since the early 1900s (a loss of 44% of its original wetlands). From the 2015 Florida Cooperative Land Cover Data, wetlands represented 11,069,804 acres in Florida, a reduction of 302,096 acres or 2.7% from the mid-1990s (**Volk et al. 2017**).

The current **STATUS** of wetlands in the LSJRB remains ***Unsatisfactory*** because of the continued stressors to wetlands, as indicated by the decrease of 627 acres between 2009 and 2014 data (Table 4.5). Currently, wetlands represent 23.3% of total LSJRWMD area (Table 4.5). In comparing wetland acreage between 2009 and 2014, losses >500 acres per community were for wet prairies, mixed wetland hardwoods, bay swamp, and cypress (Table 4.6). Gains > 500 acres per community were for freshwater marshes, mixed scrub-shrub wetlands, wetland forested mixed, and emergent aquatic vegetation (Table 4.6).

Federal, state, local, and privately managed wetlands can be found in Florida conservation lands that include national parks, state forests, preserves and parks, wildlife management areas, mitigation banks, and conservation easements (Figure 4.11, **FNAI 2019**). Comparing acreage of conservation lands between 2019 and 2020, greatest increases occurred in privately-managed lands by 3,727 acres with 3,580 acres managed by the St. Johns Mitigation Bank. Mitigating lands comprise 51% of privately-managed lands (Figure 4.11; **FNAI 2019**).

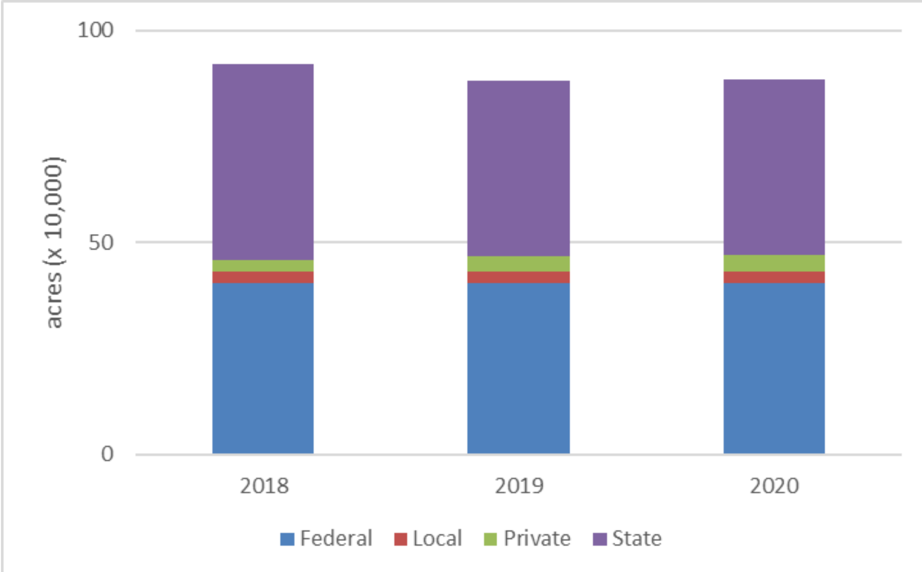


Figure 4.11 Florida Conservation Lands in the LSJRB (Source: ***FNAI 2019****).*

From a study of 20 conserved natural areas in Florida, ecosystem services were valued at $5,052 per acre (**Brown and Shi 2014**). For example, Pumpkin Hill Creek Preserve State Park was estimated in providing $6,169 per acre (**Brown and Shi 2014**). According to the North Florida Land Trust compares the cost of purchasing 112,346 acres ($216,516,934) to ecosystem value ($413,430,739 per year), highlighting the importance of setting aside these priority areas for conservation (**NFLT 2018**).

Table 4.6 Comparison in wetland acreage between 2009 and 2014 - Lower St. Johns River Basin, Florida (SJRWMD 2017a).

WETLAND CATEGORY	2009 (ACRES)	2014 (ACRES)
Mixed wetland hardwoods	156,274	152,801
Wetland forested mixed	111,155	113,682
Mixed scrub-shrub wetland	68,296	70,498
Hydric pine flatwoods	30,850	30,884
Cypress	27,543	26,723
Saltwater marshes	17,965	18,074
Wet prairies	16,276	12,449
Freshwater marshes	11,271	12,097
Bay swamp	9,640	8,721
Emergent aquatic vegetation	2,271	4,960
Cabbage palm hammock	648	741
Non-vegetated wetland	121	56
Pond pine	6	4

Stressors to wetland communities include land use, nutrients, pollutants, and invasive species. In addition, changes in populations of endangered/sensitive species can be indicators of stressed wetlands. Below is a discussion of these stressors affecting the LSJRB:

LAND USE. Land use is a powerful predictor of wetland condition (**Reiss and Brown 2007**). In Florida, countless non-tidal wetlands <5 ha that were formerly in agricultural fields and pasture lands have since been developed for residential and commercial uses (**Reiss and Brown 2007**). For example, in 1960, the population density was 43 people/km^2 as compared to 183 people/km^2 in 2000 near Deland, FL (**Weston 2014**). Landscape Development Intensity (LDI) is an index that associates nonrenewable energy use (electricity, fuels, fertilizers, pesticides, irrigation) to wetland condition. Palustrine wetlands surrounded by multi-family residential, high-intensity commercial, and central business district had LDI scores of 9.19 to

10.00 as compared to pine plantation, recreational open space (low intensity) and pastures of 1.58 to 4.00 (**Reiss and Brown 2007**).

High LDI values can be predicted for areas in the LSJRB with multi-family residential and commercial land use. Residential land is prevalent along waterways, representing 29% of total acreage within 50 m of a waterway. Surface drainage basins with residential land use can be plagued by fecal coliform (e.g., Cedar River and Black Creek) (**SRR 2014**). Leaking septic tanks, stormwater runoff, and wastewater treatment plants contribute to fecal coliform. Commercial activities also ranked with high LDI values (**Reiss and Brown 2007**). In the LSJRB, Georgia-Pacific, power plants, shipping and maritime activities, and the US Department of Defense contribute to PAH, PCB, mercury, and nitrates in Rice Creek, Cedar River, and Ortega River (**SRR 2016**). Additional sources of PCB contamination are from waste oil spills and accidental release of locomotive waste, such as hydraulics and lubricants into drainage ditches (**Flowe 2016**).

The extent of the surface drainage basin can exacerbate land use pressures (e.g., stormwater runoff). For example, the surface drainage basin Etonia Creek that includes the polluted Rice Creek covers 355 miles^2 (**Bergman 1992**). Connected surface drainage basins with a history of elevated fecal coliform levels and low oxygen include Julington Creek, Sixmile Creek, and Arlington River, covering approx. 260 miles^2 (**Bergman 1992**). Agriculture, although with a lower LDI (**Reiss and Brown 2007**) can contribute to nitrogen and phosphorous loading as is recorded from Deep Creek and Dunns Creek and cover approx. 100 miles^2 of surface drainage basin (**Bergman 1992; SRR 2014**).

NUTRIENTS. Wetlands take up terrestrial nutrients, thereby preventing or minimizing the amount of nutrients from reaching waterways. Understanding nutrient loading rates as a function of land use is critical to determining if nutrient uptake rates by wetlands can meet current demands. Stormwater runoff from residential and agricultural land use can contribute more nitrogen and phosphorous than other land use categories. For example, residential areas can release 2.32 mg N/L and 0.52 mg P/L as compared to agriculture (3.47 mg N/L and 0.61 mg P/L, respectively) and undeveloped/rangeland/forest (1.15 mg N/L and 0.055 mg P/L), respectively (**Harper and Baker 2007**). From a 2003-2009 study of water quality collected from 59 groundwater wells in the LSJRB, a relationship was evident between land use and groundwater (**Ouyang et al. 2012**). From the shallow groundwater system, septic tank land use had greater values of nitrate/nitrite concentrations than in agricultural lands (7.4 mg/L nitrate/nitrite and 0.04 mg/L, respectively). By comparison, calcium, sodium, chlorine, and sulfate had more than twice the values in agricultural lands (agriculture: 85.9, 148.8, 318.8, and 233.1 mg/L; septic tank land use: 34.5, 23.2, 36.5, and 58.8 mg/L, respectively; **Ouyang et al. 2014**). Managed plantations that use nitrogen and phosphorous excessively can be a source of nutrient loading to nearby tributaries that can be measured from weeks to years following application in sediments and water column (**Shepard 1994**). In addition, nutrient-laden waters from wastewater treatment spray fields can travel via the aquifer and contribute to nutrient loading far from the source, as has been recorded in Wakulla Spring from Tallahassee's wastewater reuse facility (**Kincaid et al. 2012**). Also, sanitary sewer overflows (SSO) can contribute to nutrient enrichment if not contained. **JEA 2020a** reported 120 SSO records with a total of 2,030,747 gallons released. In one event, 792,000 gallons was released on the Nocatee Parkway. Common reasons for SSOs were equipment or pipe failure, blockage, or contractor error.

Table 4. Records of Sanitary Sewer Overflows reported by JEA (JEA 2020a).

Year	< 11 gallons	11-100 gallons	101-1,000 gallons	1,001-10,000 gallons	>10,000 gallons	Total (average) gallons
2018	21	24	31	12	9	1,181,483 (12,307)
2019	5	26	33	9	11	959,267 (11,420)
2020	13	34	41	19	13	2,030,747 (17,065)

Nitrogen values remain high in a number of tributaries, but have been declining in the LSJR, although no data were collected in 2017 (**SRR 2018**). By comparison, total phosphorous concentrations in the tributaries is rated as unsatisfactory as compared to satisfactory and improving in the freshwater and marine/estuarine sections of the SJR (**SRR 2018**).

Sediments retain nitrogen and phosphorous. During periods of anoxic conditions due to algal blooms, Malecki et al. reported that 21% of total P load and 28% of total N load came from the sediments in the LSJR (**Malecki et al. 2004**). Dissolved reactive phosphorous released from the sediments was 37 times lower (0.13 mg per m^2 per day) than during aerobic conditions (4.77 mg per m^2 per day) (**Malecki et al. 2004**).

The presence of nutrients in combination with herbicides such as atrazine has been shown to have negative impacts on the native *Vallisneria americana* (**Dantin et al. 2010**). Submerged aquatic vegetation (SAV; e.g., *Vallisneria americana*) provides food and refuge for shrimp, blue crabs, and a variety of other fauna.

POLLUTANTS. Arsenic is present in LSJRB sediments. In Naval Station Mayport, arsenice-contaminated spoils from dredging of the basin were used to fill in wetlands and low-lying areas (**Fears 2010**). Arsenic contamination has also been documented in golf course soils (5.3 to 250 ppm, with an average of 69.2 ppm) due to herbicide applications to turf grass (81 golf courses from the northeast, 1086 surveyed in Florida; **Ma et al. 2000**). Leaching of arsenic is further exacerbated by the presence of phosphorous, commonly applied in fertilizer. Many of these golf courses have waterbodies or are near wetlands, streams, and rivers (**Ma et al. 2000**). **Ouyang et al. 2014** reported greater arsenic values in the groundwater associated with agriculture (4.3 μg/L) and wastewater sprayfield (5.6 μg/L) land use as compared to undeveloped forest lands and septic tank land use (0.6 and 1.3 μg/L, respectively) in the LSJB. In addition, superfund sites are at risk of inundation from floods, extreme high tide, and storm surge, resulting in possible contamination (**GAO 2020**). For example, Kerr-MCGee Chemical Corp, Fairfax St. Wood Treaters, and Pickettville Rd. Landfill are susceptible to natural disasters (**GAO 2020**).

Wading birds and other fauna that forage in wetlands are at risk of bioaccumulation of heavy metals. For example, mercury has been reported in the Broward and Trout Rivers. **Ouyang et al. 2012** estimated an average annual mercury load of 0.36 g ha^{-1} $year^{-1}$ within the Cedar and Ortega watershed (254 km^2). St. Johns River Power Park and Northside Generating Station have reduced their mercury atmospheric emissions by 71% between 2001 and 2013 (**SRR 2016**). However, an increase of 250% in metal discharge was reported for electric utility since 2001, in particular zinc, nickel, cobalt, and manganese (**SRR 2016**). Salt marshes are sinks for metals (**Leendertse et al. 1996**). **Giblin et al. 1980** found that metals in *Spartina alterniflora* detritus were taken up by fiddler crabs, and metals can be concentrated in bivalves near contaminated sites (**Leendertse et al. 1996**). **Burger et al. 1993** reported mean lead concentrations of 3,640 ppb dry weight in young wood storks from Dee Dot colony, demonstrating the availability of lead contamination and bioaccumulation from prey items.

HYDROLOGIC MANIPULATION. Many of the mitigation banks in the LSJRB were formerly pine plantations. Hydrology in forest plantations is typically modified to minimize surface waters (**Shepard 1994**) that can then impact non-tidal wetland diversity and sediment and nutrient loading to nearby waterways. Erosion in plantations adds to suspended sediments in drainage waters and connecting waterways (**Shepard 1994**). In lowland forested habitats, stormwater is retained in the forest and runoff occurs after the groundwater table reaches the surface (**Sun et al. 2000**). When trees are harvested, the groundwater table rises particularly during dry periods, a phenomenon that can continue over a period of years (**Sun et al. 2000**). The decrease in evapotranspiration rates with the loss of trees is responsible for this rise in the water table (**Shepard 1994**).

Bernardes et al. 2014 raised the issue of water withdrawal affecting wetlands in northeastern Florida. Depressional wetlands are typically relict sinkholes. The Florida aquifer system is crisscrossed with fractures along which groundwater can travel. Mine pits create ponds where aquifer and groundwater accumulate and thus deprives other areas of water for recharging and supporting vegetation. Where mining-related withdrawal has occurred, wetlands in nearby mitigation banks and conservation areas have dried out with the potential of becoming sinkholes. For example, the DuPont Trail Ridge Mine is in close proximity to many of the mitigation banks listed in Table 4.3 and conservation areas (e.g., Camp Blanding, Cecil Field) that wetland permittees use to mitigate wetland alteration. Water quality, hydroperiods, and water availability would be impacted (**Bernardes et al. 2014**).

INVASIVE SPECIES. The most damaging invasive plant species have the capacity to do one or more of the following: reproduce and spread successfully, compete successfully against native species, proliferate due to the absence of herbivore or pathogen that can limit their populations, and alter a habitat (**Gordon 1998**). Invasive species can modify a wetland habitat by changing geomorphology (erosion, soil elevation, water channel), hydrology (water table depth, surface flow), biogeochemical cycling (nutrient pathways, water chemistry, nitrogen fixation), and disturbance regime. *Eichhornia crassipes* and *Pistia stratiotes* are reported to impact siltation rates, *Panicum repens* stabilizes edges of waterways, *Hydrilla verticillata* slows water flow where abundant, and *E. crassipes*, *P. stratiotes*, and *H. verticillata* alter water chemistry (dissolved oxygen, pH, phosphorous, carbon dioxide, turbidity, and water color) (**Gordon 1998**).

Where invasive plant species are dominant, native weedy species typically proliferate (**Gordon 1998**). In a 2002 study of 118 depressional non-tidal wetlands in Florida, macrophyte diversity and the percentage of native perennial species in urban environments were lower than in locations away from urban environments (**Reiss 2006**). Species that were considered the most tolerant to disturbance intensity in depressional marshes included *Alternanthera philoxeroides*, *Cynodon dactylon*, *Mikania scandens*, *Panicum repens*, and *Schinus terebinthifoilius* (**Cohen et al. 2004**). From a survey of 74 non-tidal depressional wetlands in Florida, greater plant species richness was associated with more disturbed sites and fewer species in undisturbed and oligotrophic conditions (**Murray-Hudson et al. 2012**). Ruderal or weedy species are likely to tolerate changes in the wetland-upland boundary and variability in soil saturation and water depth and extent. The authors also showed that the outer zone adjacent to the upland border of a depressional wetland with high numbers of exotics would also have a high number of exotics throughout the wetland. This pattern was true for sensitive species as well, indicating that the condition of the wetland could be predicted by the richness of suites of species along the outer band of the wetland (**Murray-Hudson et al. 2012**). In addition, urban stormwater ponds can retain and contribute to the proliferation of invasive species (**Sinclair et al. 2020**). The number of invasive species present was related to whether the ponds were managed and routinely dry (**Sinclair et al. 2020**).

ENDANGERED/SENSITIVE SPECIES. Urbanization, habitat encroachment and increased recreational activities can negatively impact breeding populations of amphibians, reptiles, and birds. Development that alters and/or fills headwaters and streams negatively impacts habitat connectivity for many stream and wetland-dependent organism in the SJR watershed (**White and Crisman 2016**). Animals that require a variety of wetland types would be negatively impacted by chemical pollutants and turbidity that limits prey availability. Sensitive species associated with wetlands include the Striped newt (*Notophthalmus perstriatus*) that is listed as a candidate species for protection; and the flowering plants Chapman rhododendron (*Rhododendron chapmanii*) listed as endangered, Okeechobee gourd (*Cucurbita okeechobeensis* ssp. *okeechobeensis*), and Rugel's pawpaw (*Deeringothamnus rugelii*) that are listed as endangered in counties of the LSJRB (**USFWS 2018**). In 2018 and 2019, American Flamingo (*Phoenicopterus ruber*) and to reclassify the striped newt from unlisted to Threatened (**USFWS 2019**). Other threatened and endangered species are found in Section 4.4. Under review, the candidate Black Creek crayfish (*Procambarus pictus*) is found in Doctors Lake and Rice Creek (**USFWS 2018**). Doctors Lake is listed as impaired due to nutrient loading, and Rice Creek had been listed as impaired due to dioxin levels from Georgia Pacific discharge (**SRR 2018**).

Urbanization and subsequent habitat loss and alterations can result in negative interactions between humans and wildlife. For example, the Wildlife Service is called in to disperse or dispatch a variety of animals. Between the years 2006 and 2011, gulls, egrets, and herons represented 57% of the 4,407,393 animals that the agency dispersed through a variety of measures in Florida (e.g., firearms, pyrotechnics, pneumatics, and electronics) (**Levine and Knudson 2012**). **Cooper and Vanderhoff 2015** recorded greater numbers of the brown pelican at Mayport during autumn through spring months and along the river at Jacksonville University during winter and spring months, from a study conducted in September 2012 to August 2013. By comparison, numbers reported to eBird, a database monitored by the National Audubon Society and the Cornell Lab of Ornithology, were greatest during winter months. Comparing the Christmas Bird Counts (CBC) in years 2000 and 2013 to 2020 from a marsh near Clapboard Creek, annual counts were generally greatest in 2015 and 2016 (Table 4.7). The CBC 2020 counts were the lower than the average for the brown pelican, American oystercatcher, laughing gull, wood stork, and roseate spoonbill between 2013-2020. The brown pelican, American oystercatcher, laughing gull were the lowest and the snowy egret, little blue heron, and roseate spoonbill were the greatest in 2020 compared to earlier years (Table 4.7). The brown pelican has shown an exponential decline since 2011 at the CBC location ($R = 0.88$, $p < 0.05$). Laughing gulls and black skimmers have been declining since 2017. Changes in counts may represent habitat modifications in nearby areas.

Table 4.7 Christmas bird counts of selected species from Jacksonville marsh site in 2000, 2013-2020. SSC - species of special concern; ST - state listed, threatened; FT- federal listed, threatened; FE - federal listed, endangered (Source: 1). Red font indicates the count was less than the 2013-2020 mean.

SPECIES	STATUS	2000	2013	2014	2015	2016	2017	2018	2019	2020	2013-2020 Mean ± SE
Brown pelican	SSC	634	1100	800	600	700	700	411	500	370	648 ± 84
American oystercatcher	SSC	13	8	6	8	8	3	6	7	3	6 ± 1
Laughing gull		512	800	1100	1600	1700	1200	1000	600	400	1050 ± 160
Bald eagle		18	32	21	40	36	42	20	20	36	31 ± 3
Piping plover	FT	24	6	18	12	13	6	10	17	15	12 ± 2
Snowy egret	SSC	307	175	400	450	342	500	128	264	478	342 ± 50
Wood stork	FE	120	120	100	260	140	100	55	185	115	134 ± 22
Black skimmer	SSC	8	350	600	1000	649	4100	450	350	300	975 ± 454
Tricolored heron	SSC	128	50	100	175	100	200	46	74	118	108 ± 20
Little blue heron	SSC	54	80	100	200	165	300	144	120	358	183 ± 35
White ibis	SSC	352	200	900	800	1000	1000	1200	1300	710	889 ± 120
Roseate spoonbill	SSC	1	6	13	4	5	0	7	0	61	12 ± 7
Osprey	SSC	82	100	100	100	100	68	90	75	101	92 ± 5

Least terns are migratory birds that require sandy or gravel habitats with little vegetation for nesting. Rooftop nesting sites have become more common due to habitat loss. Large rooftop populations have been recorded at NAS Jacksonville (**Jackson 2013**). In Florida, Wildlife Service Agency had been called upon to disperse 273 least terns in 2011, indicating negative interactions with humans (**Levine and Knudson 2012**).

Wood storks (endangered) nest in the LSJR and feed on fish among other animals, requiring 450 lbs of fish per pair during the nesting season (**SRR 2018**). They require shallow pools that dry up to help concentrate fish prey. During extended periods of drought, wood stork numbers decrease. Currently, populations are considered to be improving (**SRR 2020**).

4.2.7. Current Trends in Wetlands in the LSJRB (WORSENING)

The following trends in wetlands within Florida and certain sections of the LSJRB are also notable:

- In Florida, the conversion of wetlands for agriculture, followed by urbanization, has contributed to the greatest wetland losses (**Dahl 2005**).
- The Upper Basin (the marshy headwaters of the St. Johns River) has experienced substantial historical wetland loss, and by 1983, it was estimated that only 65% of the original floodplain remained (**SJRWMD 2000**).
- **Hefner 1986** stated that "over a 50-year period in Northeast Florida, 62 percent of the 289,200 acres of wetlands in the St. Johns River floodplain were ditched, drained, and diked for pasture and crop production (**Fernald and Patton 1984**)."
- According to **DEP 2002**, "the 1999 District Water Management Plan notes seven to 14 percent losses of wetlands in Duval County from 1984 to 1995, according to National Wetlands Inventory maps."
- In 2012-2013, the SJRWMD reported a loss of 380.7 wetland acres as compared to 14.5 acres created, 2,268.6 acres preserved, and 660.1 acres enhanced (**DEP 2014d**).
- Duval Country is characterized with very high runoff values (57-331) mm, a ratio of urban runoff relative to county area) due to increases urbanization (**Chen et al. 2017**).

Development pressures that result in wetland loss and function indicate a WORSENING trend in total wetland acreage within the LSJRB, as indicated by changes in acreage between 2009 and 2014 (Table 4.8). Jacksonville is the largest city in Florida with a current population is 920,577 or 1,232 people/mi^2, having added 16,688 individuals since 2019 (**WorldPopulationReview 2020**). For example, an increase of 3,466 residential acres was recorded between 2009 and 2014 land use maps in the LSJRB.

Although the total wetland acreage cannot be statistically compared from year to year, the relative contribution of different wetland types can be statistically compared with an acceptable degree of reliability. These comparisons attempt to assess how the quality of wetlands in the LSJRB might have changed over time.

When wetland codes are grouped into two broad categories (forested wetlands and non-forested wetlands), significant trends are noted. There appears to have been a shift in the composition of wetland communities over time from forested to non-forested wetlands (Figure 4.12). Forested wetlands comprised 91% of the total wetlands in 1973, and constituted 74% of total wetlands in 2009, and 73% in 2014. **Brown and Shi 2014** estimated freshwater forested wetlands represent twice the ecosystem value as non-forested wetlands (Figure 4.1). Non-forested wetlands comprised 9% of the total wetlands in 1973, 26% in 2009, and 27% in 2014 (Figure 4.12). In the LSJRB between 2006-2013, forested wetlands represented 47-97% of permitted impacted wetland area per year (**Goldberg and Reiss 2016**).

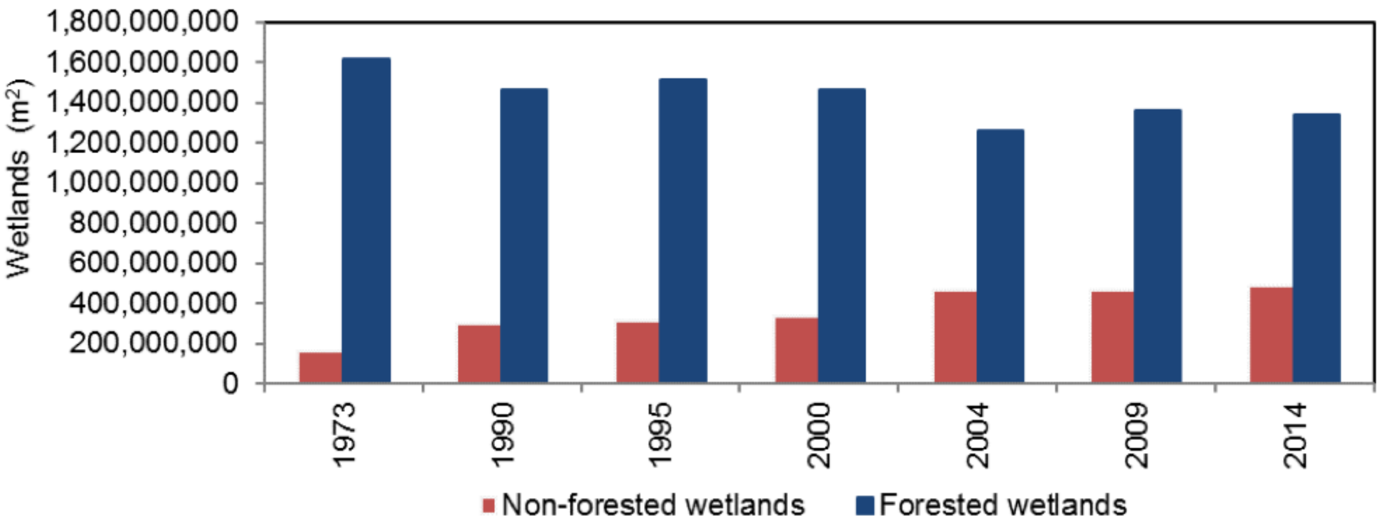


*Figure 4.12 Forested wetlands and non-forested wetlands in the Lower St. Johns River Basin based on land use/land cover maps (**SJRWMD 2017a**).*

4.2.8. Wetland Permit Trends in the LSJRB

The SJRWMD process environmental resource permits that may impact wetlands and surface waters (**SJRWMD 2017d**). In general, these projects were located in mixed hardwood wetlands. During 2020, 124 SJRWMD-processed permits were issued that required compensated mitigation, with a total of 601.5 impacted wetland acres. Between the years 2000 and 2020, the majority of issued environmental permits was for ≥11 average impacted wetland acres per project, based on SJRWMD permitting records (Figure 4.5). Incremental wetland conversions result in cumulative impacts at the landscape level.

Smaller wetlands are fragmented across the urban landscape and different habitats occur within and surrounding project sites (**Kelly 2001**) which then impacts wetland function and community composition (**Faulkner 2004**). If wetlands are few and far between, then travelling amphibians and other animals are exposed to pollutants and death on roadways (**Faulkner 2004**). Even smaller wetlands <0.2 ha contribute to local diversity (e.g., juvenile amphibians, **Semlitsch and Bodie 1998**). Permits for modifying small wetlands are the largest in numbers and yet the contribution of these wetlands to local diversity and function remains undocumented (Figure 4.13; **Semlitsch and Bodie 1998**). Permits are given to individuals and are site specific, but cumulative impacts due to the number of conversions at the landscape scale are not addressed. At the landscape level, these smaller and isolated wetlands are not as valued as riverine wetlands (**Brody et al. 2008**) and may not be protected by the Clean Water Act. Research is showing that these smaller wetlands can help take up nutrients via denitrification processes and thus reduce nitrogen and phosphorous, particularly in areas where there is heavy nutrient loading (e.g., agricultural and urban locations) (**Lane et al. 2015**). In addition, smaller wetlands contribute to the buffering of the local water table, in part due to the cumulative exchange along the perimeter of many smaller wetlands as compared to fewer but larger wetlands (**McLaughlin et al. 2015**).

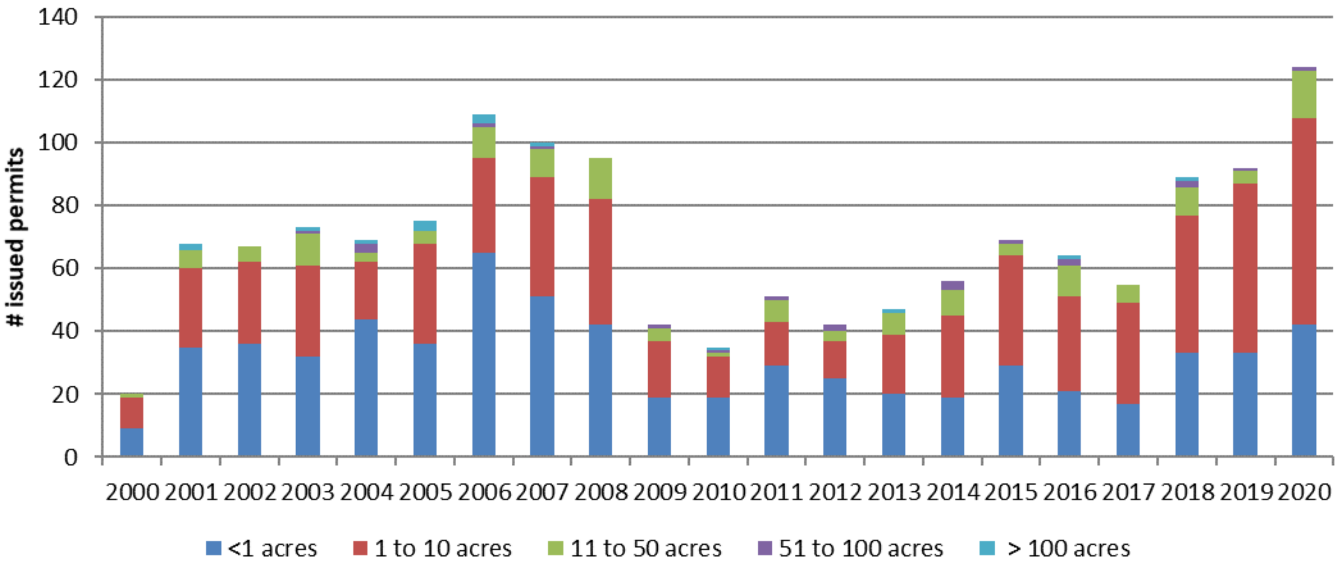


*Figure 4.13 Numbers of SJRWMD permits per project impacted wetland acreage from 2000 to 2020 (**SJRWMD 2017d**).*

Based on SJRWMD permit records, the methods used to mitigate wetlands have changed over time (Appendix 4.2.D). During the early 1990s, wetland areas were most commonly mitigated by the creation of new wetlands or through wetland restoration. During the 2000s, relatively few wetlands were created or restored with most mitigation occurring through the preservation of uplands/wetlands (Figure 4.14). In 2020, permittees applied for a total of 132.3 mitigation credits, with 85 for mitigation bank credits only and 13 for on-site only mitigation (**SJRWMD 2017d**). Eight permits included plans for enhancement and/or restoration.

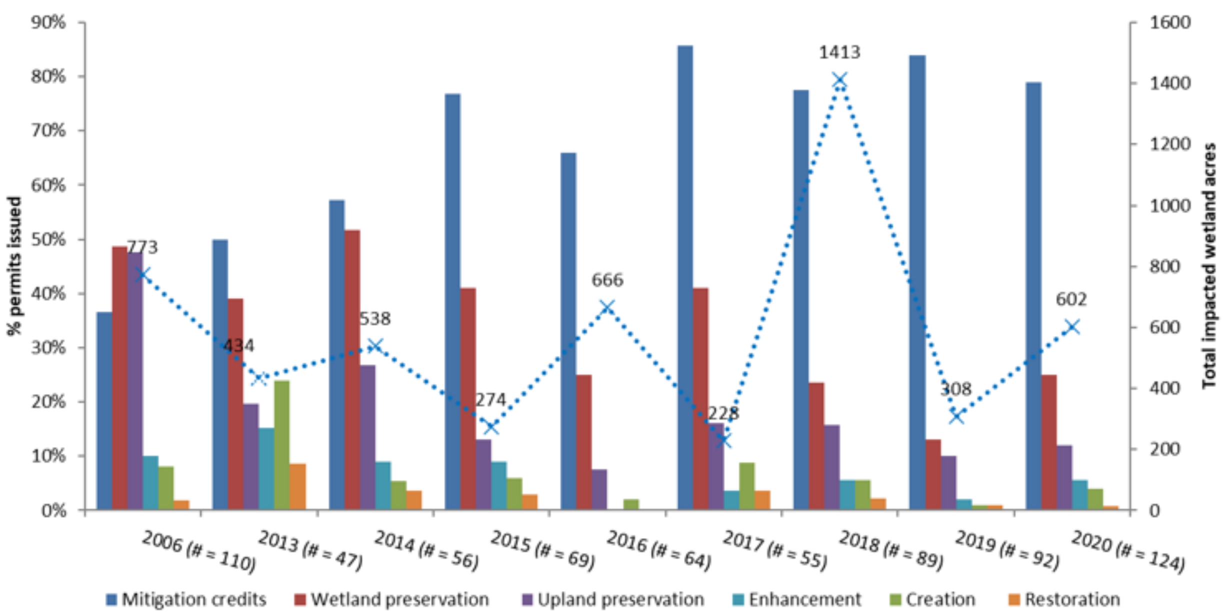


*Figure 4.14 Percentage (bars) of issued permits that opted for purchasing mitigation credits, wetland preservation, creation, upland preservation, or enhancement and total impacted wetland acres (line) in the years 2006, 2013 to 2020, indicating in parentheses the total number of permits issued for mitigated impacted wetlands. Because permittees may opt to use more than one type of mitigation for a project, total percentages per year will exceed 100% **(SJRWMD 2017d)**.*

For a complete analysis of wetlands impacted and mitigation in the LSJRB, data needed from the USACE would include the location, total acres, type of vegetation, maturation/stage of wetland, wetland functions replaced, and wetland services replaced. A similar data deficit was found by the NRC, which concluded that “data available from the Corps were not adequate for determining the status of the required compensation wetlands” (**NRC 2001**).

In 2020, the trend continued for purchase of credits to offset wetland dredge and fill activities rather than for creation, enhancement, or restoration (Figure 4.6, Table 4.8). The mean ratio of mitigation credit per acreage of impacted wetland was greater in 2006 (2.1, n = 29 permits) than in 2020 (0.4, n = 85 permits) for those projects that used mitigation banks as the only type of wetland mitigation. The Stillwater Development and Golf Course project had the greatest reported impacted wetland acreage with 51.53 acres with mitigation plans to purchase 17.28 mitigation bank credits. However, 2020 was a record year in the number of wetland acres preserved for mitigation. The Moncrief Creek Bridge Replacement permit included plans to preserve 222.65 wetland acres.

Table 4.8 Acreage comparison in wetland mitigation for permits issued in 2006, 2016-2020 that required mitigation (SJRWMD 2017d).

TOTAL ACREAGE # Permits	2006 110	2016 66	2017 56	2018 89	2019 92	2020 124
Impacted wetlands	774	754	221	1413	307	601.5
Wetland preservation	4853	2384	909	1482	416	1245.9
Upland preservation	1001	429	76	320	37	229.3
Creation	98	1	20	23	7	110.4
Enhancement/restoration	7539	0	15	19	15	334.6

4.2.9. Future Outlook

HIGH VULNERABILITY. In December 2018, the Trump Administration's EPA proposed a rule to exclude ephemeral and intermittent streams from the definition of the Waters of the United States, waters that are federally protected under the Clean Water Act (**UCS 2019**). If enacted, *at least* 18% of streams and 51% of wetlands will no longer be protected (**UCS 2019**). Given the number of isolated wetlands in the LSJR basin that are likely to be seasonally connected to waterways and/or the groundwater, the impact of this new definition is concerning.

Development, withdrawals, and flooding. The total spatial extent of wetlands negatively impacted through the SJRWMD permit process is increasing each fiscal year and is likely to increase with the improvement in the national and state economies. Urbanization at the landscape level has a direct impact on wetland communities. For example, between 2006-2013, approximately 73% of the 1,046 ha of impacted wetlands were located in Mid to High Development and 18% in Mid Development parcels (**Goldberg and Reiss 2016**).

Incremental filling of depressional ponds in addition to developing along waterways have the consequence of altering local hydrology, adding nutrients and heavy metals to the sediments and water column, bioaccumulation of heavy metals up the food web, and increasing the number and coverage of nuisance and invasive species. Isolated wetlands can retain 1,619 m^3 water/ha, on average, from models developed for Alachua County, FL, wetlands (**Lane and D'Amico 2010**). The potential for flooding, hydrologic alterations, and pressures on species diversity will continue with the loss of wetlands in the LSJRB.

Wetlands in the LSJRB will be affected in the future due to surface water withdrawals from the river as permitted by the SJRWMD. In order to fully understand and predict the potential effects, the SJRWMD released the *St. Johns River Water Supply Impact Study* in February 2012 after a peer review by the National Academy of Sciences — National Resource Council (**SJRWMD 2012a**). Using the Ortega River as a model system, the *St. Johns River Water Supply Impact Study* examined whether surface water withdrawals could potentially cause movement in the freshwater/saltwater interfaces along the river. Different types of wetland communities will be negatively impacted by future surface water withdrawals in the St. Johns River in conjunction with land use, surface water runoff, rainfall, navigational works, groundwater, and sea level rise.

In addition to development and withdrawals, tidal wetlands will be impacted by sea level rise, as well as land and property values (Table 4.10). According to **ClimateCentral 2019b**, between 1995-2004, 10 days of flooding occurred due to sea level rise and between 2005-2014, 16 days were reported in Jacksonville. In 2018, 5 days were attributed to sea level rise. Tidal wetlands in the river are unlikely to outpace sea level rise estimated at 3 mm/year (**Weston 2014**) due to inability of marsh vegetation to accrete organic material at faster rates. Delivery of fluvial suspended sediments is relatively low in the St Johns River, compared to other US rivers (**Weston 2014**). Turbidity in the mainstem is improving, indicating that sediment export to the tidal wetlands is low (see Turbidity section; **SRR 2016**). In 2015, the maximum turbidity value in 2015 was the lowest since 1997 (**SRR 2016**). Coastal wetlands may be less impacted by sea level rise. Contrary to expectations of coastal erosion with sea level rise and disruption of longshore drift with dredging activities, shorelines along Duval and St. Johns counties have been advancing since the 1800s (**Houston and Dean 2014**). On-shore sediment deposition is the likely mechanism and may help buffer erosion and sediment transport due to sea level rise in the future (**Houston and Dean 2014**).

Table 4.10 Predicted area and associated property values affected with a maximum inundation of two feet in Duval, Clay, St. Johns, Putnam, and Flagler County (ClimateCentral 2019b).

LAND AND PROPERTY VALUE	Duval	Clay	St. Johns	Putnam	Flagler
Land	17 $miles^2$	17 $miles^2$	24 $miles^2$	56 $miles^2$	24 $miles^2$
Protected land	6.4 $miles^2$	7.0 $miles^2$	10 $miles^2$	26 $miles^2$	3 $miles^2$
Property value	$1.2 billion	$156 million	$863 million	$39 million	$361 million
Local protected land	0.8 $miles^2$	0.3 $miles^2$	2.6 $miles^2$	-	0.9 $miles^2$
State protected land	0.6 $miles^2$	0 $miles^2$	1 $miles^2$	5.5 $miles^2$	1.8 $miles^2$
Federal protected land	0.1 $miles^2$	-	-	4.8 $miles^2$	-

The City of Jacksonville is exploring ways to mitigate and plan for floodplain management in the Jacksonville area, including Jacksonville Landing, northern San Marco, St. Johns Quarter (Riverside), Avondale, areas along Hogan Creek, McCoy Creek, Trout River and Ribault River **(MetroJacksonville 2017)**. For example, the 2019 and 2020 budget for Jacksonville includes $4.6 million and $5.45 million, respectively, to buyout and remove 39 homes in the flood-prone South Shores neighborhood, with the aim to provide a buffer from flooding (**Bauerlein 2020**). In the Jacksonville area, 18,891 people live in areas below 4 ft, covering 30 miles 2 (**ClimateCentral 2019a**).

The City of Jacksonville Special Committee on Resiliency recommended the following to be approved by the state (Lebron 2020):

- Ordinance 2019-331 to amend Chapter 652, Floodplain Management: Floodway setback and finished floor elevation in special flood hazard areas to be two feet above the base flood elevation;
- Ordinance 2019-375 to amend Chapter 656, Zoning Code and Chapter 654, Code of Subdivision Regulations: Impervious Surface Ratios;
- Ordinance change to create a wetland buffer that is an average of 25 feet and a minimum of 15 ft and changes to address soil permeability on filled lots, maintenance of drainage plans, and backyard drainage swales;
- Deploy additional tide gauges; and
- Provide affordable housing options that is outside located on land less prone to flooding.

Gov. Ron DeSantis has released billions of dollars to support project that prevents fertilizers from entering the waterways and capture stormwater to be used for irrigation (**Patterson 2019b**). For example, SJRWMD has contracted Sustainable Water Investment Group LLC to build a construct a wetland at a wastewater treatment facility at Doctors Lake. The company aims to capture phosphorous ten times more effectively than state standards. In return, the company will be 'paid for each pound of phosphorous' captured. In an effort to utilize stormwater as a source for urban irrigation, Clay County Utility Authority is researching the possibility of collecting and treating rainwater from the First coast Expressway (**Patterson 2019b**). Recently, the US EPA transferred the processing of wetland permits from the US ACOE to Florida's DEP, which may expedite construction in wetlands. ACOE's review of wetland permits have contributed to greater wetlands protections (**Patterson 2020a**).

On a more positive note, partial restoration of riparian corridors can have fairly immediate and positive impacts on nutrient levels and diversity of local flora and fauna (**Rossi et al. 2010**). The authors had planted riparian species of trees, shrubs, grass, and forbs to increase structural complexity in areas 3 x 9.5 m along first-order tributaries of the LSJR. After three months, sampling was conducted for two years. Macroinvertebrate diversity increased (Coleoptera and Lepidoptera), dominance of pollution-tolerant taxa decreased, and pollution-intolerant taxa (Odonta and Ephemeroptera) increased as compared to non-restored sites. In addition, soil nitrate was significantly less in the restoration sites than control sites and soil phosphorous decreased over time in restored sites due to nutrient uptake by the plants. The authors recommend incorporating restoration areas along urban stretches of the river to promote ecosystem function (**Rossi et al. 2010**). The Lasalle Bioswale Project showcases another way to minimize contaminants from entering waterways. Bioswales are vegetated areas that collect stormwater runoff. Plants and soil communities take up the pollutants and thereby treat pollutants found in stormwater runoff. This particular project was accomplished by the St. Johns Riverkeeper and partners (**St. Johns Riverkeeper 2013**). In addition, salt marsh restoration in Gamble Rogers and North Peninsula State Park has been successful, following 20 years of efforts that include removing invasive plant species and using recycled oyster shells to stabilize the banks and provide habitat (**Anderson 2020**).

In summary, the future outlook of the health of the LSJRB wetlands depends upon detailed, accurate, consolidated record-keeping of wetland impacts, the cumulative impact of parcel-by-parcel loss of wetland ecosystem functions and services, and the success of wetlands enhanced, created, or restored. Given the continued trend of mitigation via purchase of mitigation credits and off-site conservation areas in place of on-site mitigation and the implication of sea-level rise in combination with the development occurring in the LSJRB, the outlook for local wetlands in the LSJRB does not look promising.

4.3. Macroinvertebrates

4.3.1. Description

Benthic macroinvertebrates include invertebrates (animals without a backbone) that live on or in the sediment and can be seen with the naked eye. They include a large variety of organisms such as sponges, crabs, shrimp, clams, oysters, barnacles, insect larvae, and worms. Almost 400 species from 10 phyla have been identified in the LSJRB.

4.3.1.1. Sponges (Phylum Porifera)

Sponges are stationary filter feeding organisms consisting of over 5,000 species with about 150 freshwater species. They do not have organs or tissues, but the cells specialize in different functions. They reproduce both sexually and asexually (**Myers 2001c**). In the LSJRB, five taxa have been recorded and are found in fresh, marine, and estuarine waters (i.e., *Spongilla fragilis* and *Craniella laminaris*) (**Mattson et al. 2012**).

Sponge. Photo by Kimberly Mann

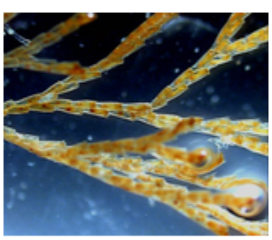
4.3.1.2. Sea Stars and Sea Cucumbers (Phylum Echinodermata)

Brittle Star (Family *Ophioderma)*
Photo by Christina Adams.

Sea Cucumber (*Cucumaria frondosa*)
http://www.sealifebase.fisheries.ubc.ca

There are approximately 7000 marine species. They can range in size from 1 cm to 2 m. Food habits vary among the different species, anything from filter feeders to scavengers to predators. Sea stars can regenerate missing arms, and sea cucumbers and urchins are also able to regenerate certain parts of their anatomy (**Mulcrone 2005**).

4.3.1.3. "Moss Animals" (Phylum Bryozoa)

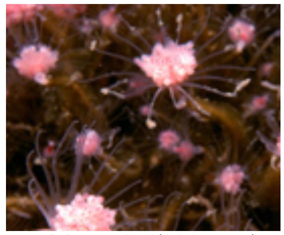


Genus *Bugula* from http://www.serc.si.edu

This group of animals lives in colonies (**Collins 1999**). They have tentacles which they use to filter phytoplankton out of the water (**Bullivant 1968**). Five non-native species have been recorded in the LSJRB (see Section 4.5 Non-native Aquatic Species; **Mattson et al. 2012**).

4.3.1.4. Jellyfish, Sea Anemones, and Hydrozoans (Phylum Cnidaria)

All the species in this phylum have stinging cells called nematocysts. They have two basic body forms – medusa and polyp. Medusae are the free-moving, floating organisms, such as jellyfish. Polyps are benthic organisms such as the hydrozoans (**Myers 2001a**). In the LSJRB, hydrozoans are more common than jellyfish and sea anemones. Eight taxa have been recorded in the LSJRB, with three taxa found in freshwater including *Corylophora lacustris* (**Mattson et al. 2012**). The non-native freshwater jellyfish *Craspedacusta sowerbyi* has been recorded in the LSJRB (see Section 4.5 Non-native Aquatic Species).



Tubularian Hydroid (*Tubularia crocea*)
Photo by Bob Michelson from
http://stellwagen.noaa.gov

Sea Anemone (Order Actiniaria) from
http://digitalmedia.fws.gov

Jellyfish (Class Scyphozoa) from
http://digitalmedia.fws.gov

4.3.1.5. Ribbon Worms (Phylum Nemertea)

Ribbon Worm (Genus *Tubulanus*)
Photo by Kare Telnes from
http://www.seawater.no/fauna/nemertea/

The common name "ribbon worm" relates to the length of many species with one species being 30 m. Marine species are more common than freshwater species (**Collins 2001**). Besides long length, these worms have an elongated appendage from the head called a proboscis that they use to capture prey. (**Collins 2001; Graf 2013**). One ribbon worm was recorded by **Evans et al. 2004** that was salt and pollution tolerant.

4.3.1.6. Snails, Mussels, and Clams (Phylum Mollusca)

The Mollusca are very diverse with >50,000 species, ranging in size from less than a millimeter to more than twenty meters long (giant squids). Over 150 taxa have been identified in the SJRB, including more than 3 invasive taxa (see Section 4.5 Non-native Aquatic Species) and others endemic to the SJR drainage (Elimia sp.) (**Mattson et al. 2012**). Representative taxa include *Mytilopsis leucophaeata*, *Gemma gemma*, *Littoridinops*, *Boonea impressa*, *Nassarius obsoletus*, and the non-native *Rangia cuneata* (**Cooksey and Hyland 2007b**). Six taxa were recorded by **Evans et al. 2004** from 2002-2003 collections in the LSJRB. Each taxon was pollution-tolerant and two taxa were gastropods and the other four were bivalves.

Snails (Class Gastropoda)
Photo by Kimberly Mann

American oyster (*Crassotrea virginica*)
Photo by Kimberly Mann

Mussel (Class Bivalvia) from
http://digitalmedia.fws.gov

4.3.1.7. "Peanut Worms" (Phylum Sipuncula)

Peanut Worm (Phylum Sipuncula) from
http://www.ucmp.berkeley.edu l

The common name "peanut worm" relates to their shape. Over 320 marine species have been described and they are found in sand, mud, and crevices in rocks and shells (**Collins 2000**).

4.3.1.8. "Horseshoe worm" (Phylum Phoronida)

Genus Phononpsis, Copyright Peter Wirtz
peterwirtz2004@yahoo.com l

Approximately 12 marine species have been identified with some species having horseshoe-shaped tentacles (**Collins 1995**). They are most common in shallow sediments. *Phoronis* has been recorded from Clapboard Creek (**Cooksey and Hyland 2007b**).

4.3.1.9. Insect larvae (Phylum Arthropoda, Supbphylum Crustacea, Class Insecta)


Insect larvae (Class Insecta) from
http://digitalmedia.fws.govl

Most insect larval forms look differently from their adult stage. Those larvae associated with aquatic habitats can be found under rocks and in the mud (**Myers 2001b**). Representative genera include *Coelonypus* and *Chrionomus* (**Cooksey and Hyland 2007b**). Sixteen taxa were recorded by **Evans et al. 2004** from 2002-2003 collections in the LSJRB. These taxa were found in freshwater, and six were pollution tolerant.

4.3.1.10. Isopods, Amphiphods, and "shrimp-like" crustaceans (Phylum Arthropoda, Subphylum Crustacea, Class Malacostraca, Superorder Peracarida)

It has been estimated that there are over 54,000 species in this group (**Kensley 1998**). They all possess a single pair of appendages (maxillipeds) extending from their chest (thorax) and mandibles. The maxillipeds assist in getting food to their mouth. For this superorder, the carapace (the exoskeleton protecting the head and some to all of the thorax is reduced in size and does not cover all of the thorax. The carapace is also used to brood eggs (**UTAS 2013**). Over 60 taxa have been recorded in the LSJRB (**Mattson et al. 2012**). In the LSJRB, eleven taxa were recorded, of which all were salt-tolerant, and four taxa were pollution-intolerant (**Evans et al. 2004**). Example taxa are *Paracaprella pusilla*, *Apocorophium lacustre*, and *Protohaustroius wigleyi* (**Cooksey and Hyland 2007b**). Two species are non-native to the SJRB (see Section 4.5 Non-native Aquatic Species).




Left: Isopod, photo by A. Slotwinski, from http://www.imas.utas.edu.au
Middle: Amphipod, photo by A. Slotwinkski, from http://www.imas.utas.edu.au
Right: Mysid ("shrimp-like"), photo by A. Slotwinski, from http://www.imas.utas.edu.au

4.3.1.11. Crabs and Shrimp (Phylum Arthropoda, Subphylum Crustacea, Class Malacostraca, Order Decapoda)


Blue Crab (*Callinectes sapidus*) from
http://digitalmedia.fws.gov


Shrimp (Order Decapoda)
Photo by Kimberly Mann

This is one of the most well-known groups since many people eat crabs, shrimps, and lobsters. Decapoda refers to the five pairs of legs. This group has an exoskeleton, which they periodically have to shed (molt) so they can continue to grow. Their body is divided into three sections – the head, thorax and abdomen. The head and thorax are fused together and covered by the carapace. In crabs, the abdomen is curved under the carapace (**Humann and Deloach 2011**). Approximately 55 taxa of crabs and shrimp have been reported in estuarine, marine, and freshwater in the LSJRB (Appendix 3.3.2a-3.3.3b). In the SJRB, five species are commercially and/or recreationally (**Mattson et al. 2012**) harvested. In 2002-3, **Evans et al. 2004** recorded two taxa in salt waters, of which *Rhithropanopeus harrisii* was pollution intolerant. Four species are non-native to the SJRB (see Section 4.5 Non-native Aquatic Species).

4.3.1.12. Barnacles (Phylum Arthropoda, Subphylum Crustacea, Class Malacostraca, Infraclass Cirripedia)


Gooseneck Barnacles,
http://www.digitalmedia.fws.gov l

There are approximately over 1,400 species. Size can range from a few centimeters to slightly greater than 10 cm. Barnacles are attached to a hard substrate or other organisms. The carapace completely encloses their soft body. They do not possess compound eyes or appendages. For most, their habitat is along rocky shoreline in the intertidal zone (**Newman and Abbott 1980**). Two taxa were recorded by **Evans et al. 2004** that were salt and pollution tolerant in the LSJRB. Five non-native taxa have been recorded in the LSJRB (see Section 4.5 Non-native Aquatic Species).

4.3.1.13. Worms (Class Polychaeta, Phylum Annelida)


Limnodrilus hoffmeisteri (Subclass Oligochaeta) from http://www.fcps.edu


Class Polychaete,
Photo by Kimberly Mann

This phylum consists of worms that have segmented bodies, including earthworms. Polychaete means "many bristles" and members of this class look like feathered worms. Over 200 taxa have been recorded in the SJRB (**Mattson et al. 2012**). Example taxa are *Streblospio benedicti*, *Mediomastus*, *Neanthes succinea*, *Nereis*, *Sabellaria vulgaris*, *Paraonis fulgens*, *Nephtys picta* (**Cooksey and Hyland 2007b**). *Streblospio benedicti* and *N. succinea* are pollution tolerant and representative of impaired environmental conditions (**Cooksey and Hyland 2007b**). Seventeen taxa were recorded by **Evans et al. 2004**, of which two taxa were pollution intolerant (*Orginiidae* sp. and *Scolopolos rubra*) and another two species that were freshwater tolerant (*Aulodrilus pigueti* and *Limnodrilus hoffmeisteri*) (**Evans et al. 2004**).

4.3.2. Significance

Benthic macroinvertebrates are an important component of the river's food web. Indeed, many of the adults of these species serve as food for commercially and recreationally important fish and invertebrate species. Their microscopic young can also be very abundant, providing food resources for smaller organisms, such as important larval and juvenile fish species. Benthic activities in the sediment or bioturbation can result in sediment turnover, changes in oxygen and nutrient availability, and distribution of grain size. The presence of stress-tolerant species can serve as an indicator of river health (**Pearson and Rosenberg 1978; Gray et al. 1979**). For more information on pollution in benthic invertebrates, see Section 5 Contaminants.

4.3.3. Data Sources

Macroinvertebrate community data used to assess long-term trends were obtained from the Florida Department of Environmental Protection (DEP), Florida's Inshore Marine and Assessment Program (IMAP), and the St. Johns River Water Management District (SJRWMD) from 1973-2000 with supplemental data from DEP's "Fifth Year Assessments" (**DEP 2013j**). No dataset has been compiled within the past three years.

4.3.4. Current Status (UNCERTAIN)

The current **STATUS** is rated as ***Uncertain*** due to lack of data.

4.3.5. Trend (UNCERTAIN)

Community shifts are expected in response to the natural changes in water quality, salinity, and temperature in addition to biological factors that can include recruitment and predation variability (**Cooksey and Hyland 2007b**). It is important to recognize that the mechanism by which many of these organisms may be affected is by either direct impact to adults or to the offspring that spend part of their time in the water column as plankton. During the planktonic stage of these organisms' lives, environmental gradients (i.e., salinity, temperature, dissolved oxygen) within the river can affect where young are and how they are transported to adult habitat.

The current trend is rated as UNCERTAIN. The lack of recent surveys and monitoring of benthic macroinvertebrates makes it difficult to identify trends, especially since microhabitat variability can be as high as site variability. Yet, low species richness, diversity, and abundance are representative of impaired benthic conditions (**Cooksey and Hyland 2007b**). The health of the SJR is linked to the health of benthic macroinvertebrates. A potential concern is if macroinvertebrate communities change in a large area within the river, and then affect abundances of ecologically, commercially or recreationally important species (for example, red drum, spotted sea trout, or flounder).

4.4. Threatened & Endangered Species

The species examined in this section are Federally-listed threatened and endangered species that occur in Duval, Clay, St. Johns, Putnam, Flagler and Volusia Counties in the LSJRB (**USFWS 2021a**). These animals are protected under the Endangered Species Act of 1973 (**Congress 1973**). The West Indian manatee, bald eagle, and wood stork discussed here are considered primary indicators of ecosystem health because of their direct use of the St. Johns River ecosystem. The data available for these species were relatively more robust than data on the also listed shortnose sturgeon, piping plover, Florida scrub-jay, and Eastern indigo snake (although included in past reports, the latter three have not be included in this report). In addition, other endangered or threatened species of interest to the area include the North Atlantic Right Whale and Loggerhead Sea Turtle. However, because these animals are associated with the coastal and offshore boundaries of the LSJRB, they are not discussed in this report. All these examples convey in part the diverse nature of endangered wildlife affected by human activities in the LSJRB. These species, and many more, add to the overall diversity and quality of life we enjoy and strive to protect and conserve for the future. It is important to be aware that human actions within the LSJRB affect the health of the entire ecosystem, and that the St. Johns River is a critical component of this system. Research, education and public awareness are key steps to understanding the implications of our actions towards the environment. The list of species examined here does not include all species protected under Florida State (131 species within the state) and federal laws (13 species within LSJRB) (see Appendix 4.4.1). It is likely that in the future this list will need to be periodically updated as changes occur over time or indicator species and data are identified. For additional supporting information, the reader is asked to refer to the appendices section of the report.

4.4.1. The Florida Manatee (reclassified 2016, current status: Threatened)

Photo by Chelsea Bohaty, Blue Springs State Park.

4.4.1.1. <u>Description</u>

In 1967, under a law that preceded the Endangered Species Act of 1973 the manatee was listed as an endangered species (**Udall 1967**). Manatees are also protected at the Federal level under the Marine Mammal Protection Act of 1972 (**Congress 1972**), and by the State under the Florida Manatee Sanctuary Act of 1978 (**FWC 1978**). More recently, because manatees are no longer considered to be in imminent danger of extinction, the US Fish and Wildlife Service announced that the West Indian manatee was reclassified from endangered to threatened status on March 30, 2016. This action does not affect federal protections currently enforced under the Endangered Species Act (**USFWS 2021a**).

The Florida manatee (*Trichechus manatus latirostris*) is a large aquatic mammal that inhabits the waters of the St. Johns River year round and may reach a length of 12 ft and a weight of 3,000 lbs (**Udall 1967**; **USFWS 2001**). They are generally gray to dark-brown in color; have a seal-like body tapering to a flat, paddle-shaped tail. Two small forelimbs on the upper body have three to four nails on each end. The head is wrinkled, and the face has large prehensile lips with stiff whiskers surrounding the nasal cavity flaps. They are not often observed during winter (December-February) being generally most abundant in the St. Johns River from late April through August. Because of their herbivorous nature all are found in relatively shallow waters where sunlight can penetrate and stimulate plant growth. Manatees do not form permanent pair bonds. During breeding, a single female, or cow, will be followed by a group of a dozen or more males, or bulls, forming a mating group. Manatees appear to breed at random during this time. Although breeding and birth may occur at any time during the year, there appears to be a slight spring calving peak. Manatees usually bear one calf, although twins have been recorded. Intervals between births range from three to five years (**JU 2021**). In 1989, Florida's Governor and Cabinet identified 13 "key" counties experiencing excessive watercraft-related mortality of manatees and mandated that these counties develop a Manatee Protection Plan (MPP). The following counties have state-approved MPPs: Brevard, Broward, Citrus, Collier, Dade, Duval, Indian River, Lee, Martin, Palm Beach, Sarasota, St. Lucie, and Volusia (**FWC 2014b**). In 2006, although not one of the original 13 "key" counties, Clay County also voluntarily developed a State-approved MPP. St. Johns County also voluntarily developed a manatee plan, but it is had not been approved by State or Federal agencies. Putnam County does not have a MPP, whereas Flagler County is in the process of developing one. The fourth revision to the Duval County MPP was adopted by the Jacksonville City Council on February 13th 2018 (**COJ 2021a**).

Jacksonville University has conducted some 777 aerial surveys with over 18,638 manatee sightings (1994-2020). These surveys covered the shorelines of the St. Johns River, its tributaries (Jacksonville to Black Creek), and the Atlantic Intracoastal Waterway (Nassau Sound to Palm Valley). During the winter, industrial warm water sources were also monitored for manatee presence (aerial and ground surveys). It was observed that when water temperatures decrease (December through March); the majority of manatees in the LSJRB migrate to warmer South Florida waters (**White and Pinto 2014**).

Within the St. Johns River, survey data indicate that manatees feed, rest and mate in greater numbers south of the Fuller Warren Bridge where their food supply is greatest relative to other areas in Duval County. Sightings in remaining waters have consisted mostly of manatees traveling or resting. Manatees appear to use the Intracoastal Waterway as a travel corridor during their seasonal (north/south) migrations along the east coast of Florida. Data indicate that manatees stay close to the shore, utilizing small tributaries for feeding when in these waters (**White et al. 2002**). Aerial surveys of manatees, by various organizations and individuals, in northeast Florida have occurred prior to 1994 and are listed in **Ackerman 1995**.

There are two sub-populations of manatees that use the LSJRB. The first sub-population consists of about 624 manatees from the Blue Springs area (**Hartley 2021**), of which numbers visiting the LSJRB are not known (**Ross 2021**). Most of the animals in the LSJRB, about 260+ manatees (**White and Pinto 2006b**; **White and Pinto 2006a**) are members of the greater Atlantic region sub-population, with 2,394 animals in 2019 along the entire east coast of Florida, and 3,339 along the west coast for a total of 5,733 manatees (**FWRI 2021e**). State synoptic surveys did not occur in some years (1993, 1994, 2008, 2012, 2013, 2020, and 2021) because weather conditions were not preferable. The warm winters meant that manatees did not aggregate well at warm water sources for counting. The Florida counts have grown significantly over time as the population has increased from an average of 1,530 manatees in the early 1990's to 2,376 manatees (1995-2007), 4,635 manatees (2009-2015), and more recently 6,159 manatees (2014-2019). Considerable coordination and effort by FWRI are involved, for example in 2011, 21 observers from 10 organizations counted 2,432 manatees on Florida's east coast and 2,402 on the west coast for a sum total of 4,834 (Figure 4.15). In general, few animals tend to be seen in the LSJRB because of the cold weather; although, some animals are found at artificial warm water sources. No animals were observed in the northeast Florida synoptic survey area in 2011, 2015, 2016, and 2018. However, in 2010 and 2014, two animals were observed. In 2017, the previous synoptic record count was surpassed with 3,488 animals on the east coast and 3,132 on the west coast of Florida, for a total of 6,620 manatees (on this occasion, 6 animals were observed in the northeast synoptic survey area (**FWRI 2021e**).

*"Synoptic" can be defined as a general statewide view of the number of manatees in Florida. The FWC uses these surveys to obtain a general count of manatees statewide by coordinating an interagency team that conducts the synoptic surveys from one to three times each year (weather permitting). The synoptic surveys are conducted in winter and cover all of the known wintering habitats of manatees in Florida. The survey is conducted to meet Florida state statute 370.12 (4), which requires an annual, impartial, scientific benchmark census of the manatee population. From 1991 through 2018, the counts have been conducted 33 times (**FWRI 2021e**).*

The weather conditions in 2010 were the coldest for the longest duration in Florida metrological history. Consequently, manatees were more concentrated at warm water sources throughout the state resulting in the second highest count ever recorded at that time with 2,780 animals on the east coast, and 2,296 animals on the west coast for a sum total of 5,076 animals (**FWRI 2018b**).

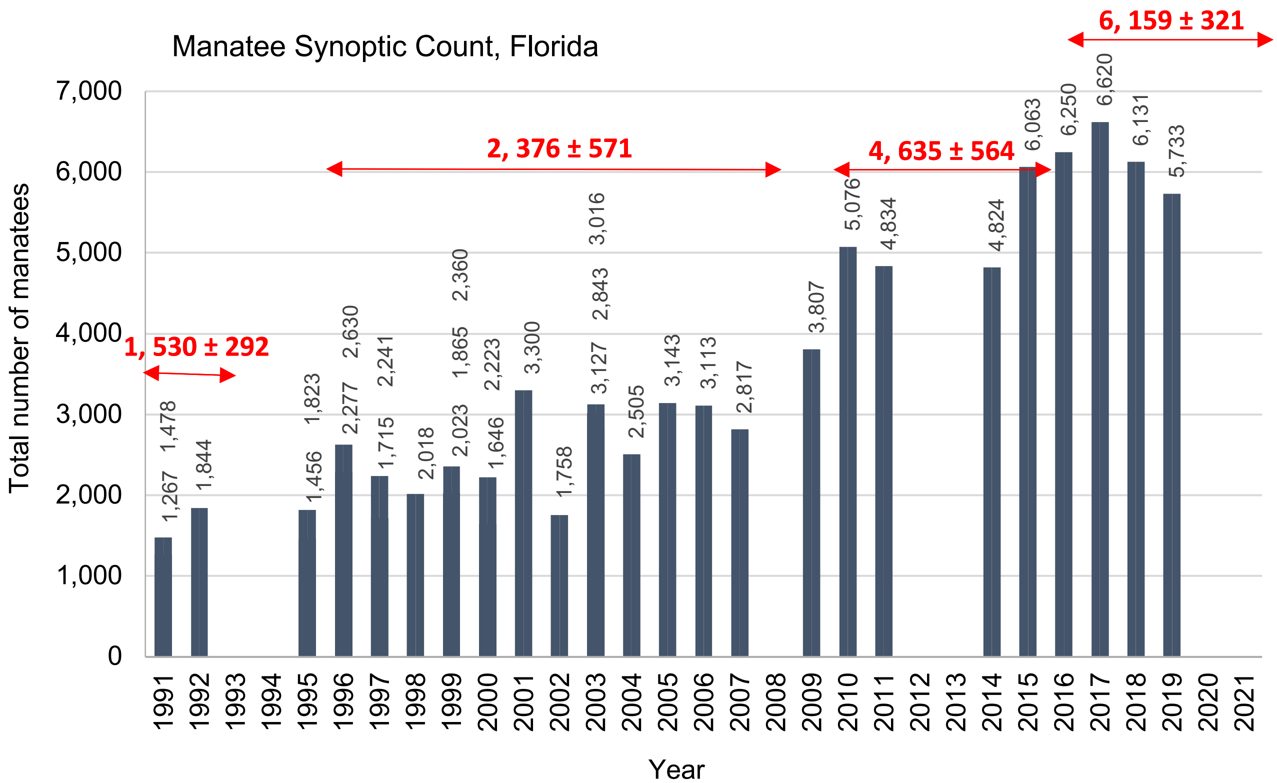


*Figure 4.15 Synoptic aerial counts of manatees in Florida 1991-2021. Vertical numbers above bars indicate totals for the east and west coast (grey); Horizontal numbers show mean and standard deviation, and arrows indicate the period averaged (red) (Source: **FWRI 2021c**).*

It should be noted that because of differences in the ability to conduct accurate aerial surveys the synoptic results cannot be used to assess population trends. For more information, see Appendix 4.4.1.A Synoptic Counts. This information is based on the results of long-term radio tracking and photo-identification studies (**Beck and Reid 1998**; **Reid et al. 1995**). **Deutsch et al. 2003** reported that the LSJR south of Jacksonville was an important area visited by 18 tagged manatees that were part of a 12-year study of 78 radio-tagged and tracked manatees from 1986 to 1998. Satellite telemetry data support the fact that most animals come into the LSJRB as a result of south Florida east coast animals migrating north/south each year (**Deutsch et al. 2000**). Scar pattern identification suggests that significant numbers of manatees are part of the Atlantic sub-population. Since 2000, a total of 7 animals: 4 recovered in Duval County (2006, 2008, 2010, and 2012); 2 from Clay County (2011, and 2013); and 1 from St. Johns County (2010), were recovered in the northeast Florida area, that were identified as animals that came from the Blue Springs sub-population (**Beck 2018**). **Hostetler et al. 2018** estimated that the average number of manatees in Florida in 2015–2016 was 8,810 (range 7,520-10,280).

4.4.1.2. Significance

The St. Johns River provides habitat for the manatee along with supporting tremendous recreational and industrial vessel usage that threatens them. From 2000 to 2019, pleasure boats have increased the most and represent about 97% of all vessels. St. Johns, Clay, and Flagler Counties experienced an increasing trend in the number of vessels. Duval and Putnam Counties experienced a decreasing trend in vessels. For information about each county, see Appendix 4.4.1.A Vessel Statistics. Watercraft caused mortalities of manatees continue to be the most significant threat to survival. Boat traffic in the river is diverse and includes port facilities for large industrial and commercial shippers, commercial fishing, sport fishing and recreational activity. Florida Department of Highway Safety and Motor Vehicles (**FDHSMV 2021**) records show that there were 34,483 registered boaters in Duval County in 2002. This number increased to 34,494 by 2007 and has since fallen from 28,519 in 2012 to 25,553 in 2019. Duval County had the most vessels (41%) followed by St. Johns and Clay (17%), then Putnam (11%) and Flagler (7%). Port statistics indicated that 4,166 vessel passages occurred to and from the Port in 2012, and that these decreased to 3,312 in 2017 but then increased again to 3,522 in 2018. This then decreased in 2019 (3,432) and 2020 (3010) due to the COVID-19 pandemic (**JAXPORT 2021**). In addition to this, in 2004, there were 100 cruise ship passages to and from the Port, and by 2007, this number rose to 156. In 2008 there was a decrease to 92 cruise ship passages, and then from 2009-2019 the number of passages averaged 155. Large commercial vessel calls and departures are projected to increase significantly in the future (**JAXPORT 2007**). Also, in order to accommodate larger ships, the JAXPORT dredged turning basins in 2008 and began to deepen the channel near the mouth of the SJR in December of 2017. Dredging can cause a change in vessel traffic patterns and increase noise in the aquatic environment that can potentially harm manatees because they cannot hear oncoming vessels (**Gerstein et al. 2006**). Dredging a deeper channel can also affect the salinity conditions in the estuary by causing the salt water wedge to move further upstream **(Talke et al. 2021b; Sucsy 2008**), which may negatively impact biological communities like tape grass beds on which manatees rely for food (**Twilley and Barko 1990**).

4.4.1.3. Data Sources & Limitations

Aerial survey data collected by Jacksonville University (Duval County 1994-2020, and Clay County 2002-2003) were used in addition to historic surveys by FWC (Putnam 1994-1995). Ground survey data came from Blue Springs State Park (1970-2020). The FWRI provided manatee mortality data from 1975-2020. Other data sources include the USGS Sirenia Project's radio and satellite tracking program, manatee photo identification catalogue, tracking work by Wildlife Trust and various books, periodicals, reports and websites.

Aerial survey counts of manatees are considered to be conservative measures of abundance. They are conducted by slow-speed flying in a Cessna high-wing aircraft or Robinson R44 helicopter at altitudes of 500-1,000 ft. (**JU 2021**) and visually counting observable manatees. The survey path was the same for each survey and followed the shorelines of the St. Johns River and tributaries, about every two weeks. Throughout the year, survey time varied according to how many manatees were observed. This is because more circling is often required to adequately count them. The quality of a survey is hampered by a number of factors including weather conditions, the dark nature of the water, the sun's glare off the water surface, the water's surface condition, and observer bias. The units of aerial surveys presented here are the average number of manatees observed and the single highest day count of manatees per survey each year. The number of surveys each year prior to 2012 averaged 19 ± 3.5 SD (range 11-26/yr). Since then, funding for aerial surveys was significantly reduced due to budget cuts, which resulted in a lower survey frequency of 2-5 surveys/yr. This includes additional assistance with surveys from the USCG Air Auxiliary Unit when possible. The reduced survey effort has significantly reduced the power to predict trends and represents a further limitation in the data.

The actual location that a watercraft-related mortality occurred can be difficult to determine because animals are transported by currents or injured animals continue to drift or swim for some time before being reported. In addition, the size of the vessel involved in a watercraft fatality is often difficult to determine with frequency and consistency.

Because the frequency and duration of elevated salinity events in the river can adversely affect the health of Submerged Aquatic Vegetation (SAV) on which manatee rely for food, rainfall and salinity were examined in conjunction with the number of manatees. Updated salinity data were provided by Bill Karlavige (Environmental Quality Division, City of Jacksonville). Water quality parameters are measured monthly at ten stations in the mainstem of the St. Johns River at the bottom (5.0 m), middle (3.0 m), and surface (0.5 m) depths. Data on rainfall came from the SJRWMD and NOAA (Appendix 4.1.7.1.E Rainfall, Hurricanes, and El Niño), and salinity data for specific SAV monitoring sites came from SJRWMD (Appendix 4.1.7.1.F Salinity) and USGS continuous sampling probes at Buckman Bridge and Racy Point. Regarding the salinity data associated with SAV sites and including grass beds information, these data were not available for 2012 to 2014 because that program (encompassing 152 sites) was suspended due to budget cuts. Sampling resumed on a more limited basis in 2015-2020 and each year more sites were added back into the SJRWMD sampling program. In 2015/2016, there were about 56 sites; in 2017 (61 sites); 2018 (81 sites) and in 2019 (112 sites). In 2020, the number of sites decreased to 44 sites because of COVID restrictions. Most of these sampling sites are in the area from the Buckman Bridge to Federal Point. Few exist from Palatka to Lake George in the St. Johns River. There were 10 new sites established around Lake George in 2019, but these were not sampled in 2020. In 2019, there were 17 sampling sites north of Buckman Bridge to Jacksonville, 13 (76%) were devoid of submerged vegetation, and in 2020, there were 16 sites sampled, and 11 (69%) did not have grass (See Appendix 4.1.7.1.A-E for the status of grass beds).

4.4.1.4. Current Status

Aerial surveys: The average numbers of manatees observed on aerial surveys in Duval County and adjacent waters decreased prior to the drought (2000-2001) and then increased again after the drought (2000-2005). In 2005, drought conditions developed again, and numbers began to decline (Figure 4.16). Since 2009, manatee numbers have begun to increase again. The longer-term trend (1994-2019) appears to be relatively stable, when excluding the variation caused by the droughts and other severe weather. Data points from 2013 to 2019 are likely to be significantly affected by reduced sampling frequency.

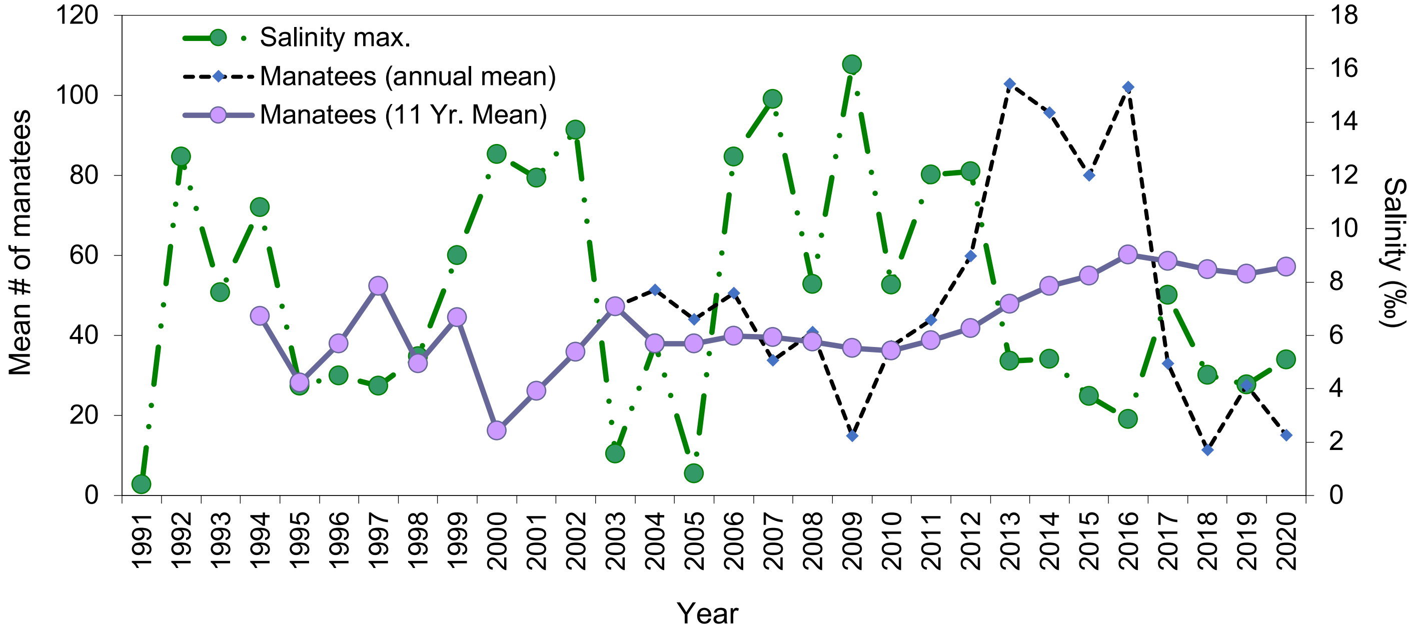


Figure 4.16 Mean numbers of manatees per survey in Duval Co., FL and adjacent waters 1994-2020.
After 2012, reduced sampling effort resulted in the annual mean (45.12 manatees S. D. ± 23.76) fluctuating more from the smoothed mean 42.25 manatees (S. D. ± 10.62, 11-year average)
Data source: Jacksonville University and City of Jacksonville (Appendix 4.4.1.A).

Single highest day counts of manatees appear to have increased to a level slightly higher than prior to the drought, but the increase is not statistically significant (2000-2005). The large dip in numbers in 1999-2000 can be attributed to the effects of the drought that caused manatees to move further south out of the Duval County survey area in search of food (Figure 4.17). A second dip in numbers (2005-2009) occurred as a result of another series of droughts. In 2010, manatee numbers began to increase again and in 2012 a high count of 177 manatees was recorded. In 2016, this was surpassed by another higher count of 192 manatees. Data points from 2013 to 2019 are likely to be significantly affected by reduced sampling frequency. In spite of this, manatee numbers dropped significantly in the area from 2016-2019, primarily because of a lack of food resources that were impacted by drought and storms that hampered recovery. In fact, 2017, 2018, and 2019 represent an anomalous period that was characterized by a severe drought in the spring and summer of 2017 which increased salinity and caused the grasses to die back. Then tremendous storm activity later in September of the same year with Hurricane Irma produced significant rainfall and tidal surge. The following year Hurricane Michael in October followed two tropical storms earlier in the year (Alberto in May and Gordon in September) all producing above average rainfall in the area. In 2019, another major Hurricane Dorian arrived in August/September affecting the area with much precipitation. The high flow conditions reduced water clarity, hampering grass bed regeneration after the drought (see Appendix 4.1.7.1.A-E for the status of grass beds and Rainfall, Hurricanes, and El Niño).

"Single highest day count" of manatees is defined as the record highest total number of manatees observed on a single aerial survey day during the year. This provides a conservative indication of the maximum number of manatees in the study area.

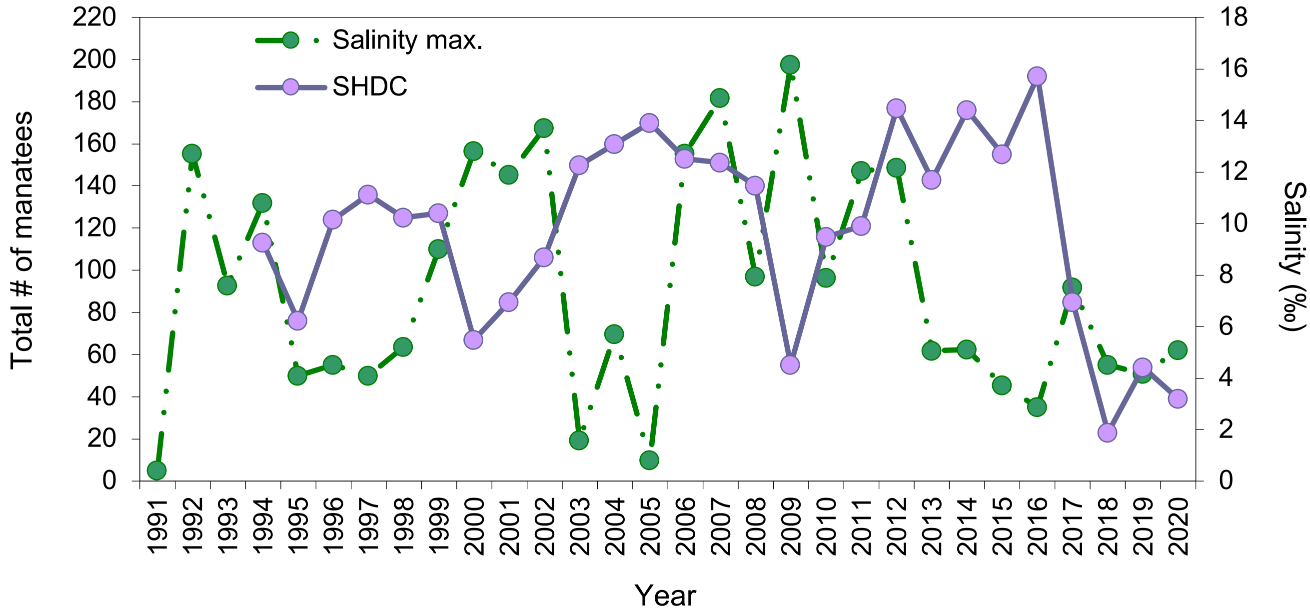


Figure 4.17 Single highest day count per year of manatees in Duval Co., FL 1994-2020.
Data source: Jacksonville University and City of Jacksonville (Appendix 4.4.1.A).

<u>Ground surveys:</u> Blue Springs is located about 40 miles south of the LSJRB within the St. Johns River system, and since this sub-population has increased over the years, we could potentially see more animals using the LSJRB in the future. The population of Blue Springs only numbered about 35 animals in 1982-83 (**Kinnaird 1983a**) and 88 animals in 1993-94 (**Ackerman 1995**). From 1990-1999, this population had an annual growth rate of about 6% (**Runge et al. 2004**). It is the fastest growing sub-population and accounts for about 5% of the total Florida manatee count (**FWC 2007**). Ground surveys indicate that the six-year average for total number of manatees seen has increased from 6% (1994-2003) to 24% (2004-2019); note also that most of these animals stay in the vicinity of Blue Springs and that calves represent about 7-9% of the total number sighted (Figure 4.18).

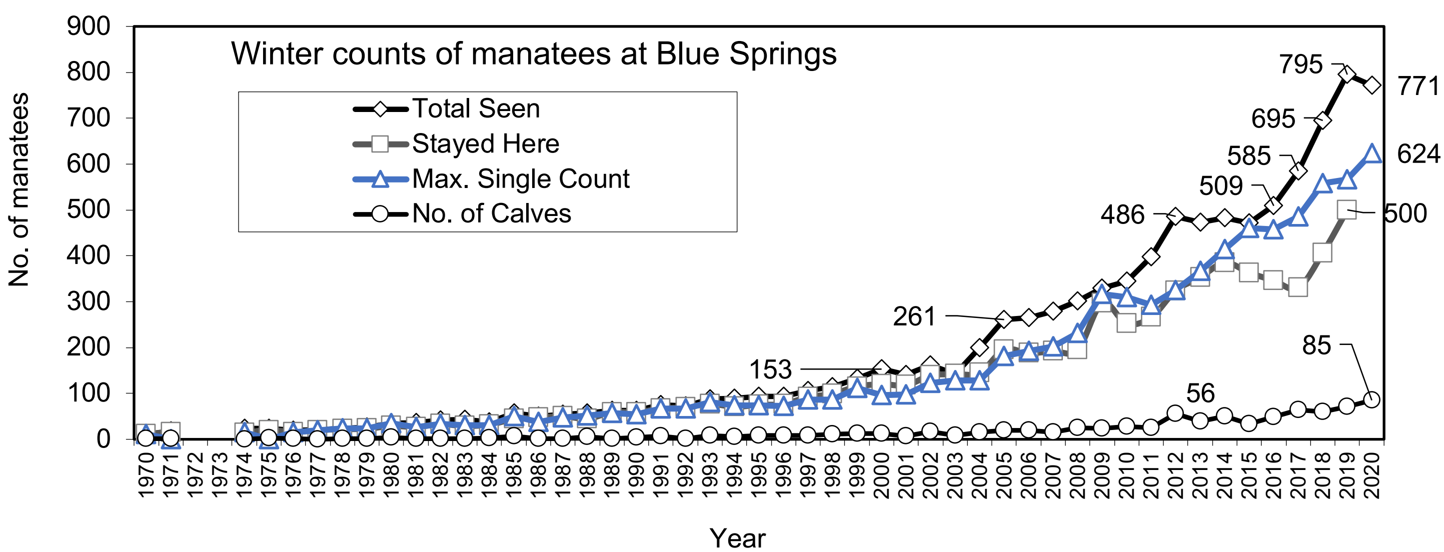


Figure 4.18 Winter counts of Florida manatees identified at the winter aggregation site in Blue Springs State Park, Volusia Co., FL 1970-2020.
*Maximum single day counts and animals that stayed at the site are also indicated (Data source: **Hartley 2021**).*

Total Mortality: There were a total of 812 manatee deaths in the LSJRB from 1980-2020 (Figure 4.19), of which a total of 205 were caused by watercraft (25% of total manatee deaths), 17 were from other human related causes, 138 were of a perinatal nature, 162 were from cold stress, 45 from other natural causes, and 230 were from undetermined causes. The total number of manatee mortalities (from all causes) increased towards the mouth of the SJR with Duval County being associated with 55% of all deaths, followed by St. Johns (17%), Putnam (14%), Clay (11%), and Flagler County with 8% (**FWRI 2021d**).

Manatee mortality categories defined by FWRI

Watercraft (Propeller, Impact, Both)	*Cold Stress*
Flood Gate/Canal Lock	*Natural, Other (Includes Red Tide)*
Human, Other	*Verified; Not Recovered*
Perinatal (Natural or Undetermined)	*Undetermined; Too decomposed*

Beginning in the winter of 2020 and into mid-2021, 761 (~10% of the population) of manatees mostly from the Indian River Lagoon area have died due to starvation. The loss of sea grasses is likely due to "super algae blooms" that screen available light from reaching the plants. These blooms are the result of excessive nutrients that continue to enter these waters for over a decade now. Plans are in the works to improve water quality but are in their early stages. In the meantime, manatee rehabilitation centers are overloaded and unfortunately, more animals are likely to perish (**Pennisi 2021**).

Watercraft Mortality: Watercraft-related mortalities in the lower basin, as a percentage of the total mortality by county, were highest in Duval (32%) followed by Putnam (18%), Clay (17%), Flagler (14%) and then St. Johns (13%). Since most deaths in the basin occurred in Duval County, watercraft deaths in Duval County were compared in five-year increments beginning 1980 through 2020 (note that the 2020-2024 category only has one year currently represented). These times were picked because they represent uniform periods either side of 1994 when the Interim Duval County MPP regulations were implemented. From 1980 to 2004, watercraft deaths of manatees in Duval County averaged 31% of total deaths, and from 2005 to 2009, watercraft deaths were 52% of total deaths. For the 5-year period from 2010 to 2014, watercraft-caused mortality decreased to 24% of total manatee mortalities in LSJRB. For the last five years from 2015 to 2019, it averaged 20% (Appendix 4.4.1.A).

In comparison, the average watercraft death rate for the state for the same period 20% (± s.d. = 2.77%). Mortalities from watercraft in LSJRB showed an upward trend since the mid-1990s, with most reported in Duval County. In the last decade, watercraft deaths of manatees have decreased slightly in LSJRB. The watercraft mortality for the LSJRB was 26% of total mortality in 2018, and the state watercraft mortality rate was 15% (lower proportionally because of high numbers of red tide deaths in south Florida. In 2016, it was 28% for LSJRB and 20% for the state (**FWRI 2021d**).

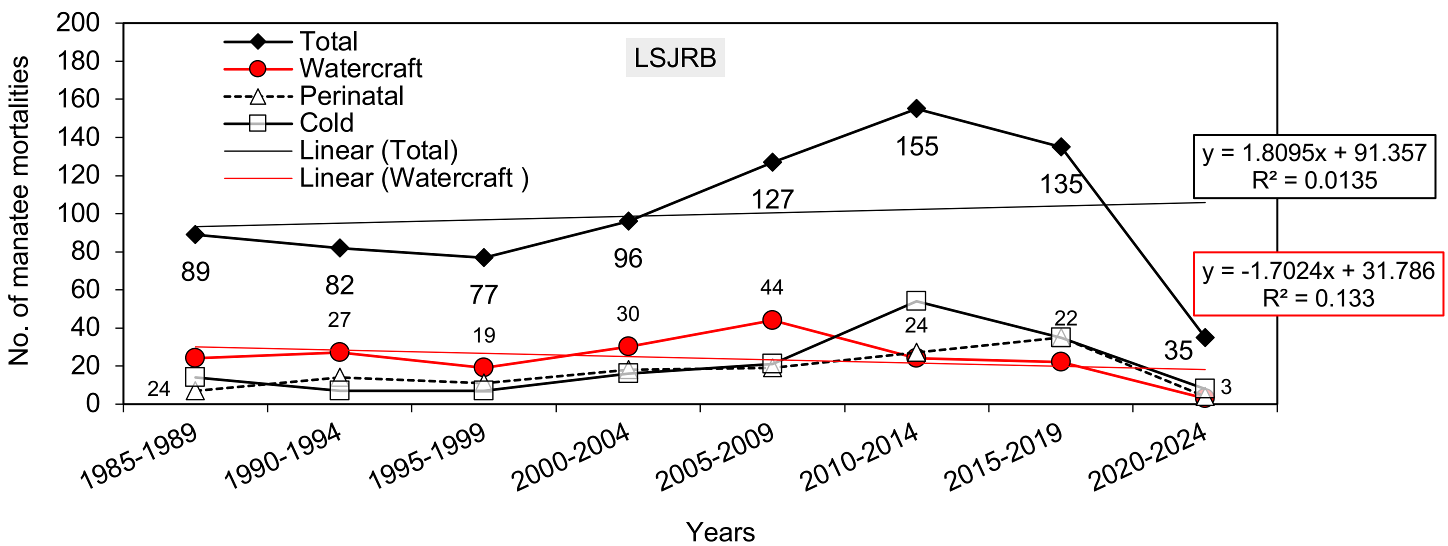


Figure 4.19 Summary of total (large numbered diamonds), watercraft (small numbered circles/red), perinatal, and cold stress manatee mortalities by county in LSJRB (five-year intervals from 1980-2019). Also, numbers do not sum to the total mortality because other natural, other human, flood gate/canal lock, unrecovered and undetermined causes are not included on the graph. Note that 2020-2024 only indicates one year of data.

In 2018, Florida experienced an unusually high level of mortality with a total of 824 manatee deaths, of these 122 were watercraft caused and some 200+ were likely from red tide effects mainly in southwest Florida (**FWRI 2021d**).

Cold stress: When manatees experience prolonged exposure to water temperatures below 68 °F (20 °C), they can develop a condition called cold-stress syndrome, which can be fatal. Effects of cold stress may be acute, when manatees succumb rapidly to hypothermia, or longer-lasting as chronic debilitation. Chronic cold-stress syndrome is a complex disease process that involves metabolic, nutritional, and immunologic factors. Symptoms may include emaciation, skin lesions (see below on the snout) or abscesses, fat depletion, dehydration, constipation and other gastrointestinal disorders, internal abscesses, and secondary infections. The manatee in the picture below was recovering well from severe cold stress (April 2018) at the Manatee Critical Care Center, Jacksonville Zoo.

Photo by G. Pinto

Cold-stress mortalities were particularly elevated throughout Florida during the period January to March 2010 (Figure 4.19). This period included the coldest 12-days ever recorded in the state of Florida with temperatures below 45 °F (7.2 °C) recorded in Naples and West Palm Beach. Central Florida experienced even colder temperatures. From January-April, 58 manatees were rescued, and 503 manatee carcasses were verified in Florida (429 in all of 2009). Mortality was highest in the central-east and southwest regions. Florida manatees rely on warm-water refuges to survive winter and extended cold periods, which are of particular concern because the long-term survival of these animals will be dependent on access to warm water springs as power plant outfalls throughout the Florida peninsula are shut down (**Laist et al. 2013**). In LSJRB there were a total of 12 cold stress deaths between January 14th and February 15th 2010 – Clay (2), Duval (1), Flagler (0), Putnam (7), and St. Johns (2), compared to a total of 6 cold stress deaths in 2011 – Clay (0), Duval (3), Flagler (0), Putnam (2), and St. Johns (1) (**FWRI 2012a**).

The State Manatee Management Plan (**FWC 2007**) requires the FWC to evaluate the effectiveness of speed zone regulations. The Plan was developed as a requirement in the process, that sought to down list manatees from endangered to threatened status. Currently, manatees are considered threatened at the federal level. Taking everything into account, the current **STATUS** of the Florida Manatee is ***Satisfactory***, and the **TREND** is ***Improving***.

4.4.1.5. Future Outlook

Manatees in the LSJRB are likely to continue to increase as more manatees move north because of population increase, decreases in manatee habitat and its quality in south Florida. Although threats still exist, manatees do not appear to be in imminent danger of extinction. As a result, the US Fish and Wildlife Service has ruled that the manatee status be upgraded to "threatened" without affecting federal protections currently enforced under the ESA (**USFWS 2021b**).

Recovery from the most recent drought cycle (2009-2012) should allow food resources to rebound and increase the carrying capacity of the environment to support more manatees. Current information regarding the status of the Florida manatee suggests that the population is growing in most areas of the southeastern US (**USFWS 2007b**). In 2013, the aerial survey budget was significantly reduced to the point that useful information about population trends is more limited. In light of that issue, the USCG Auxiliary Air Unit stepped up to offer assistance in providing flights, when possible. Just like in Lee County, Florida (**Semeyn et al. 2011**), the manatee count and distribution information in the form of maps is discriminated to local, state and federal law enforcement, maritime industry groups, the port, and the media so that efforts can be focused on raising public awareness through education. The focus on education is primarily so that manatee deaths from watercraft can be reduced. In general, there has been a spatial shift over the last fifteen years in that fewer manatees are seen in areas north of the Buckman Bridge for extended periods of time, and more tend to congregate further south. This correlates with more suitable habitat to the south versus the north. There appears to be a decreasing trend in watercraft-caused deaths for the LSJRB from 2010-2019, though if this trend is sustained or not remains unclear (**FWRI 2021d**). Although there is a decreasing trend in registered vessels in Duval and Putnam Counties, significant increases in vessel traffic in the LSJRB are projected to occur over the next decade as human population increases and commercial traffic increases. During 2020, there was an uptick in vessel purchases across the board though the general long-term trend remains unchanged (Appendix 4.4.1.A). More boats and more manatees could lead to more manatee deaths from watercraft because of an increased opportunity for encounters between the two. Dredging, in order to accommodate larger ships, significantly affects boat traffic patterns and noise in the aquatic environment (**Gerstein et al. 2006**) and has ecological effects on the environment that ultimately impact manatees and their habitat. Freshwater withdrawals, in addition to harbor deepening, will alter salinity regimes in the LSJRB. If a sufficient change in salinity regimes occurs, it is likely to cause a die-off of the grass bed food resources for the manatee. **Mulamba et al. 2019** indicate that increasing salinity effects from rising sea-level are likely to be felt throughout the whole estuary. Model results indicated a hotspot in the river where salinity was predicted to increase by about 2 ppt (near river Km 30, or in-between Acosta Bridge and the Buckman Bridge (40 km from the river mouth). **Talke et al. 2021b** noted that dredging has increased tidal amplitude most inland of the inlet (20-25 km) and that this results in increased vulnerability to storm surges and flooding (see section 2.8 Salinity).

This result would decrease carrying capacity of the environment's ability to support manatees. Some Blue Springs animals use LSJRB too, although the interchange rate is not known yet. Animals that transition through the basin are likely to be affected by the above issues. Sea level rise is another factor likely to affect the St. Johns and about which more information regarding potential impacts is needed. In addition, any repositioning of point sources can alter pollution loading to the St. Johns River and should be monitored for any potential impacts to manatees (i.e., thermal/freshwater sources), and also the grass beds on which they depend for food. Moreover, the cumulative effects of freshwater withdrawals on these and other flora and fauna should be monitored to assess the impacts of water supply policy (**NRC 2011**). On a positive note, important monitoring programs reduced or eliminated due to budget cuts in the last few years are being reestablished, although not yet to their previous levels. Fewer data impacts the ability of planners to gauge the effectiveness of programs that have the goal of improving environmental conditions in the river and may lead to additional costs in the future. Storms in 2017, 2018 and 2019 impeded the river's ability to bounce back ecologically. In the past, concerns were about insufficient rainfall and the frequency of droughts. More recently, too much rainfall and related effects, including flooding, have been more notable. In the future, we can expect more of the extremes that further limit the ability of nature to bounce back.

"Carrying Capacity" may be defined as the maximum weight of organisms and plants an environment can support at a given time and locality. The carrying capacity of an environment is not fixed and can alter when seasons, food supply, or other factors change.

4.4.2. Bald Eagle (delisted 2007)

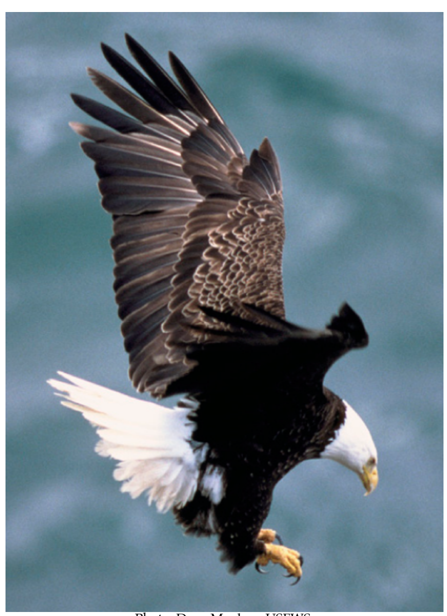


Photo: Dave Menken, USFWS.

4.4.2.1. Description

The bald eagle *(Haliaeetus leucocephalus)* is a large raptor with a wingspan of about seven feet and represents a major recovery success story. Bald eagles were listed as endangered in most of the US from 1967-1995 as a result of DDT pesticide contamination, which was determined to be responsible for causing their eggshells to be fragile and break prematurely. The use of DDT throughout the US was subsequently banned, though it is still present in the environment (see Section 5.6 Pesticides). In 1995, bald eagle status was upgraded to threatened, and numbers of nesting pairs had increased from just under 500 (1960) to over 10,000 (2007).

As a result of this tremendous recovery, bald eagles were delisted June 28, 2007 (**USFWS 2007a**, **2008a**, **2008d; AEF 2016**). The eagles are found near large bodies of open water such as the St. Johns River, tributaries, and lakes, which provide food resources like fish. Nesting and roosting occurs at the tops of the highest trees (**Scott 2003b; Jacksonville Zoo 2019**, **2020**Bald eagles are found in all of the United States, except Hawaii. Eagles from the northern United States and Canada migrate south to over winter while some southern bald eagles migrate slightly north for a few months to avoid excessive summer heat (**AEF 2021**). Wild eagles feed on fish predominantly, but also eat birds, snakes, carrion, ducks, coots, muskrats, turtles, and rabbits. Bald eagles have a life span of up to 30 years in the wild and can reach 50 years in captivity (**Scott 2003b**; **AEF 2021**; **Scott 2003b; AEF 2016; Jacksonville Zoo 2019**, **2020**). Young birds are brown with white spots. After five years of age the adults have a brown-black body, white head, and tail feathers. Bald eagles can weigh from 10-14 lbs and females tend to be larger than males. They reach sexual maturity at five years, and then find a mate that they will stay with as long as they live (**AEF 2021**).

4.4.2.2. Significance

From 2006-2010, there was an average of 59 active nests out of a total of 107 bald eagle nests surveyed. The nests were located mainly along the edges of the St. Johns River, from which the birds derive most of their food (Appendix 4.4.2.A). Most of the nests seem to be in use about 57% of the time. Active nests represented 53% (range 47-62%) of the total nests surveyed from 2006-2008. In 2010, the number of active nests increased to 70%. Data for 2009 indicated fewer nests, because of a change in survey protocol starting November 2008 (**Gipson 2014**). After a hiatus of two years, bald eagle nests were surveyed again in 2013 and numbers of active nests had not changed significantly from 2010 (**Gipson 2014**) (Figure 4.20).

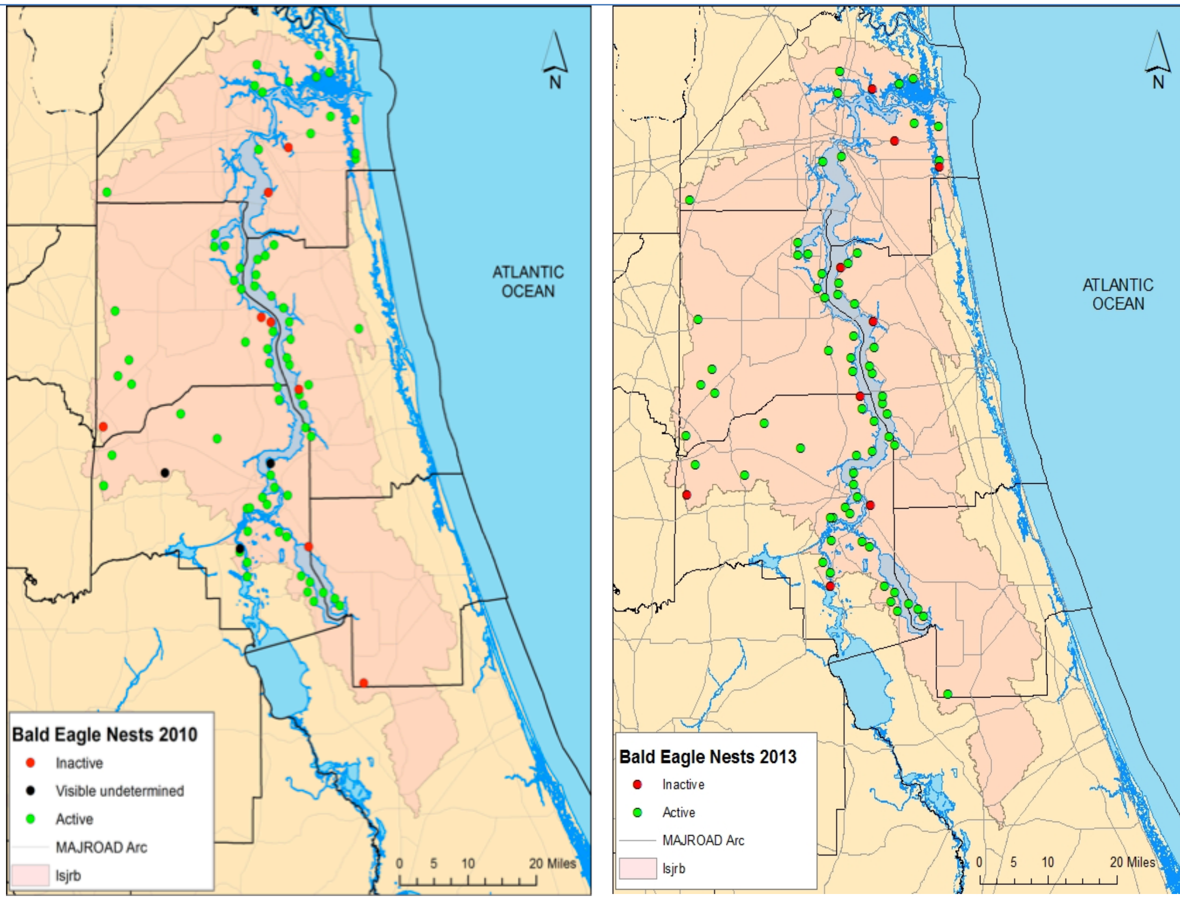


Figure 4.20 Bald eagle nesting sites in LSJRB 2010 and 2013 (Source data: ***Gipson 2014****).*

4.4.2.3. Data Sources & Limitations

Data came from a variety of sources: Audubon Society winter bird counts, FWC, Jacksonville Zoo and Gardens, USFWS and various books and web sites. No new data for the LSJRB area was available from FWCC for 2011/2013 and 2014-2020. Various groups conduct periodic surveys and the state has a five-year management plan (**FWC 2008**) to monitor the eagle's continued welfare (**FWC 2008**; **USFWS 2008a**). Known bald eagle nesting territories within the State of Florida were surveyed by FWC during the 2009 nesting season with fixed-wing or rotary-wing aircraft beginning in late November 2008 and extending through mid-April 2009. Nest locations were determined with the use of aircraft-based GPS units. Accuracy of locations is estimated to be within 0.1 miles of the true location. In 2008, the statewide bald eagle nesting territory survey protocol changed. The protocol change reduces annual statewide survey effort and increases the amount of information gained from the nests that are visited during the survey season. Nest productivity is now determined for a sub-sample of the nests that are surveyed annually. Nest activity and productivity information are critical to determining if the goals and objectives of the Bald Eagle Management Plan are being met (**FWC 2008**).

4.4.2.4. Current Status

In Alaska, there are over 35,000 bald eagles. However, in the lower 48 states of the US, there are now over 5,000 nesting pairs and 20,000 total birds. About 300-400 mated pairs nest every year in Florida and constitute approximately 86% of the entire southern population. Statewide eagle nesting surveys have been conducted since 1973 to monitor Florida's bald eagle population and identify their population trends. Now that this species is no longer listed as Threatened, the primary law protecting it has shifted from the Endangered Species Act to the Bald and Golden Eagle Act (**AEF 2014**; **USFWS 2008b**, **2008c**). In spite of the recent dip in numbers likely due to damage caused by the storms in 2017-2019 and habitat changes, Jacksonville winter bird counts by the Duval Audubon Society indicate that from 1981-2020 numbers sighted have increased significantly ($\tau = 0.776$; $p=3.52E-12$; $n=38$) since the pesticide DDT was banned in the 1960s (Figure 4.21). Taking everything into account, the current **STATUS** of the Bald Eagles is ***Satisfactory***, and the **TREND** is ***Improving***.

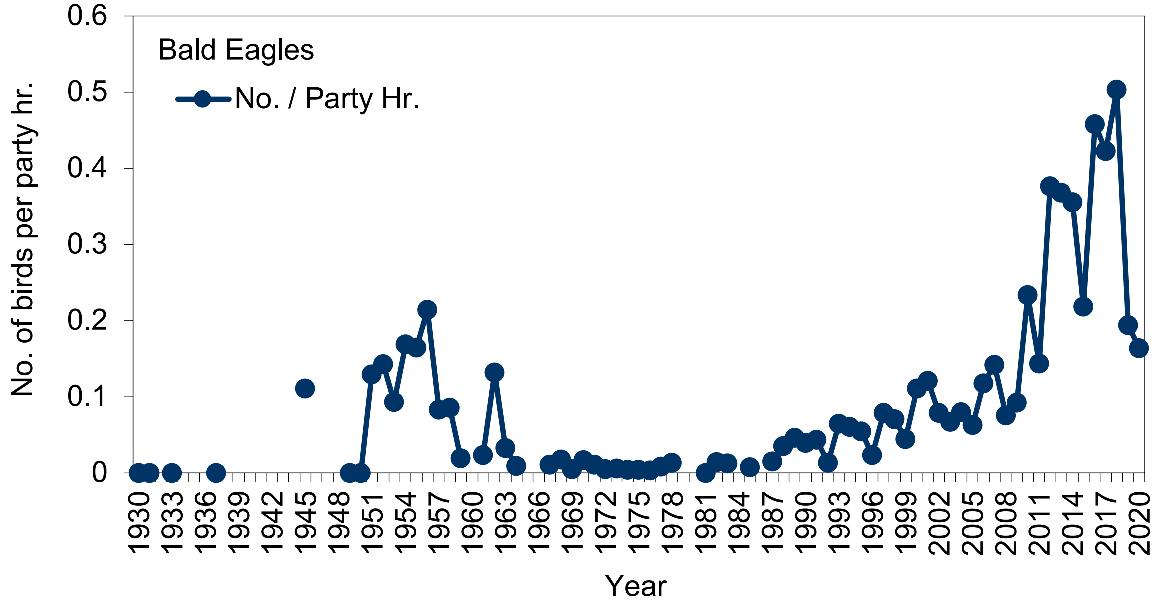


Figure 4.21 Long-term trend in the number of bald eagles counted during winter bird surveys (1929-2020) in Jacksonville, FL (Source data: ***Audubon 2021****; Appendix 4.4.2.A).*

In a recent Kendall tau correlation analysis of rainfall for the LSJRB, count data for Audubon count circle in Jacksonville was negatively correlated to rainfall, but not significant (τ = -0.068; NS). The analysis indicated increase in numbers of eagles over time with respect to party hours of effort (τ = 0.446; p=1.46E-08; n=70) and raw numbers (τ = 0.515; p=1.43E-10; n=70), respectively (Figures 4.21 and 4.22).

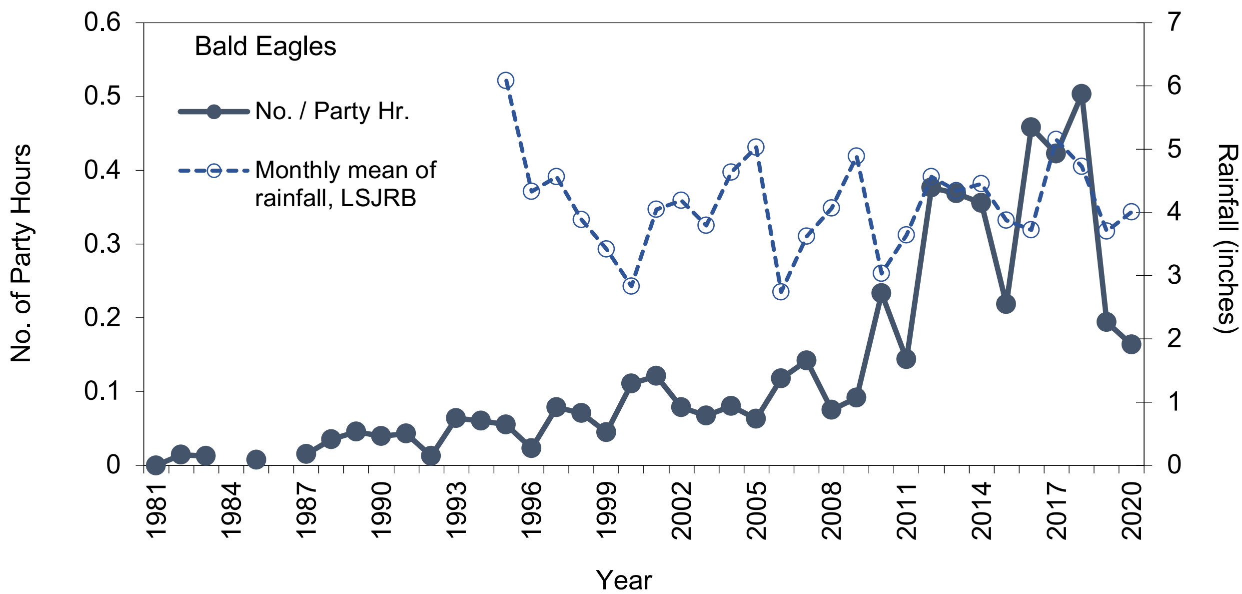


Figure 4.22 Long-term trend in the number of bald eagles counted per party hour and mean monthly rainfall (1981-2020) in Jacksonville, FL (Source data: ***Audubon 2021****;* ***SJRWMD 2021c****; Appendix 4.4.2.A).*

Eagle counts are expressed as numbers of birds per party hour, which accounts for variations due to the effort in sampling the birds. Each group of observers in the count circle for a day is considered one "party" and counts are conveyed together with the number of hours the observers recorded data (note this is not the number of hours of observation multiplied by the number of observers). Number of birds per party hour is defined as the average of the individual number per party hour values for each count circle in the region. In the case of no observations of a given species by a circle within the query region, a value of zero per party hour is averaged.

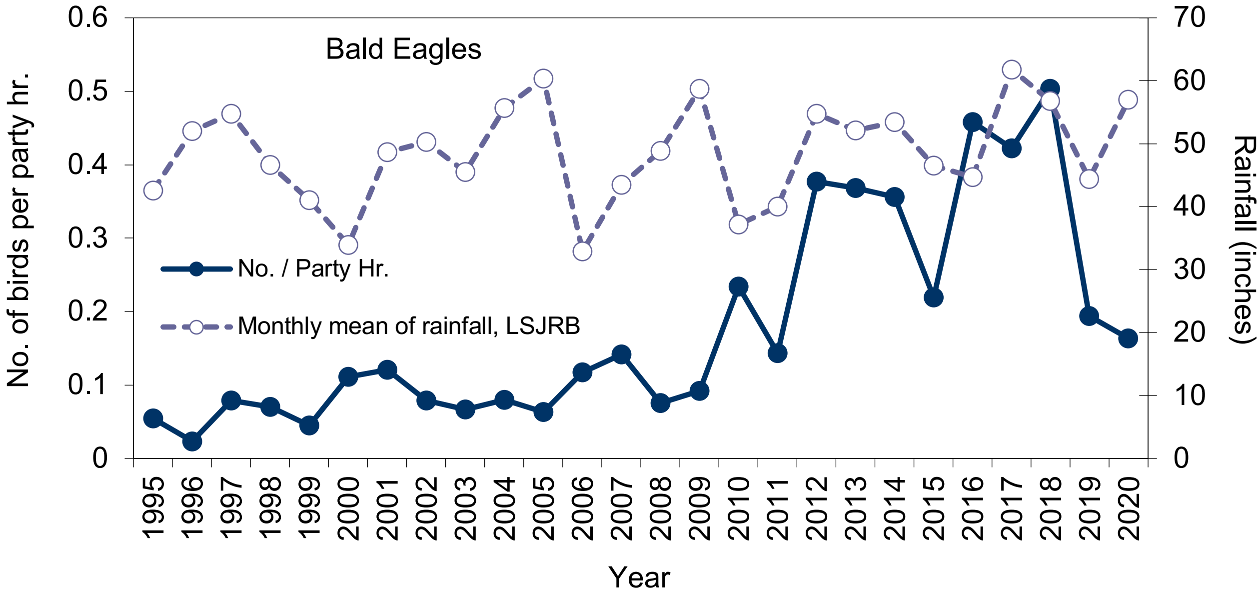


Figure 4.23 Recent trends in the number of bald eagles counted per party hour and mean monthly rainfall (1995-2020) in Jacksonville, FL (Source data: ***Audubon 2021****;* ***SJRWMD 2021c****; Appendix 4.4.2.A).*

There was a decreasing trend in rainfall 1995-2000, which represents a prolonged period of severe drought (coincides with 1997 El Niño year). Bald eagle numbers surged as the drought deepened probably because of a concentration of their prey as water levels fell. Then, rainfall increased again from 2000-2005 with averages approaching and finally exceeding the norm by 2005. During this period, the number of eagles declined somewhat, presumably because prey resources were more spread out. Also, there was an increase in severe storms (including hurricanes, which usually have a higher potential to affect the US during La Niña years) during this time period. Following 2005, another drought ensued (2005-2006), and rainfall declined at a faster rate than previously. Again, eagle numbers surged. From 2006-2009, rainfall increased toward pre-drought levels again and eagle numbers declined. Following 2009, another drought cycle began, and the eagle numbers increased abruptly. In 2010, rainfall and the number of bald eagles increased. The dip in eagle numbers in 2010/2011 may have been caused by the unusually cold weather experienced at the time. In 2012, eagle numbers remained at an all-time high with only a slight dip in 2013/2014. In 2015, there was a significant decrease in eagle numbers, but in 2016/2018, following a period of drought bald eagle numbers increased again so that the overall trend remains upward. In 2019/2020, eagle numbers decreased substantially possibly due to nest damage caused by recent storms and habitat changes that may have affected the distribution and availability of prey. High flows and water levels tend to reduce the availability of prey that are more dispersed. Decreasing salinity likely changed the distribution and abundance of fish species that eagles typically feed on (see Appendix 4.1.7.1.E Rainfall, Hurricanes, and El Niño; Appendix 4.1.7.1.F. Salinity).

4.4.2.5. Future Outlook

Although they have a good future outlook, bald eagles are still faced with threats to their survival. Environmental protection laws, private, state, and federal conservation efforts are in effect to keep monitoring and managing these birds. Even though bald eagles have been delisted from endangered to threatened, it is imperative that everyone does their part to protect and monitor them, because they are key indicators of ecosystem health. The use of DDT pesticide is now outlawed in the US Ongoing threats include harassment by people that injure and kill eagles with firearms, traps, power lines, windmills, poisons, contaminants, and habitat destruction with the latter cause being the most significant (**FWC 2008; USFWS 2008a**; **AEF 2021**).

4.4.3. Wood Stork (down listed 2014, current status: Threatened)

Photos by G. Pinto, Jacksonville Zoo Colony

4.4.3.1. Description

The wood stork *(Mycteria americana)* was listed as endangered in 1984 and is America's only native stork. The reason for the Endangered Species Act (ESA) listing was declining numbers of nesting pairs from about 20,000 (1930s) to 3,000-5,000 pairs in the 1970s (**Jacksonville Zoo 2021**). Wood storks originally recommended to be down listed (**USFWS 2007c**) were upgraded to threatened status in June 2014 (**USFWS 2018**). It is a large white bird with long legs and contrasting black feathers that occur in groups. Its head and neck are naked and black in color. Adult birds weight 4-7 lbs and stand 40-47 inches tall, with a wingspan in excess of 61 inches. Males and females appear identical. Their bill is long, dark and curved downwards (yellowish in juveniles). The legs are black with orange feet, which turn a bright pink in breeding adults.

Wood storks nest throughout the southeastern coastal plain from South Carolina to Florida and along the Gulf coast to Central and South America. Nesting occurs in marsh areas, wet prairies, ditches, and depressions, which are also used for foraging. They feed on mosquito fish, sailfin mollies, flagfish, and various sunfish. They also eat frogs, aquatic salamanders, snakes, crayfish, insects, and baby alligators. They find food by tactolocation (a process of locating food organisms by touch or vibrations). (**USFWS 2002**; **Scott 2003c**). Feather analysis of the banded chicks at Jacksonville Zoo suggests that the primary food sources being fed to the chicks is freshwater prey items not zoo food items or estuarine prey. Satellite tracking data to date supports this foraging pattern, with adults feeding primarily on an estuarine prey base prior to nesting, switching to freshwater prey base during chick rearing, and then return to an estuarine diet after chick fledging and during the rest of the year (**Jacksonville Zoo 2021**). Nesting occurs from February to May, and the timing and success is determined primarily by water levels. Pairs require up to 450 lbs of fish during nesting season. Males collect nesting material, which the female then uses to construct the nest. Females lay from 2-5 eggs (incubation approx. 30 days). To keep eggs cool, parents shade eggs with out-stretched wings and dribble water over them. Wood storks can live up to ten years but mortality is high in the first year (**USFWS 2002**; **Scott 2003c**).

4.4.3.2. Significance

Wood stork presence and numbers can be an indication of the health of an ecosystem. The wood stork is also Florida's most endangered species of wading bird that requires temporary wetlands (isolated shallow pools that dry up and concentrate fish for them to feed on). Scarcity of this specific habitat type due to human alteration of the land is one cause of nesting failures, as has been reported in the Everglades (**Scott 2003c**).

4.4.3.3. Data Sources & Limitations

Data came from Audubon Society winter bird counts from 1962-2019, USFWS surveys and *Southeast US Wood Stork Nesting Effort Database*, FWC/FWRI collaborative work in the SJRWMD area, and Donna Bear-Hull of the Jacksonville Zoo and Gardens from 2000-2019. The Audubon winter bird count area consists of a circle with a radius of ten miles surrounding Blount Island in Jacksonville, FL. The USFWS has conducted aerial surveys, which are conservative estimates of abundance and are limited in their use for developing population estimates. However, they still remain the most cost-effective method of surveying large areas. Ground surveys on individual colonies, like at the zoo, tend to be more accurate but cost more on a regional basis (**USFWS 2002**).

4.4.3.4. Current Status

An increasing trend since the 1960s was indicated by the Audubon Society winter bird count data for Jacksonville (Figure 4.24 and Appendix 4.4.3.A).

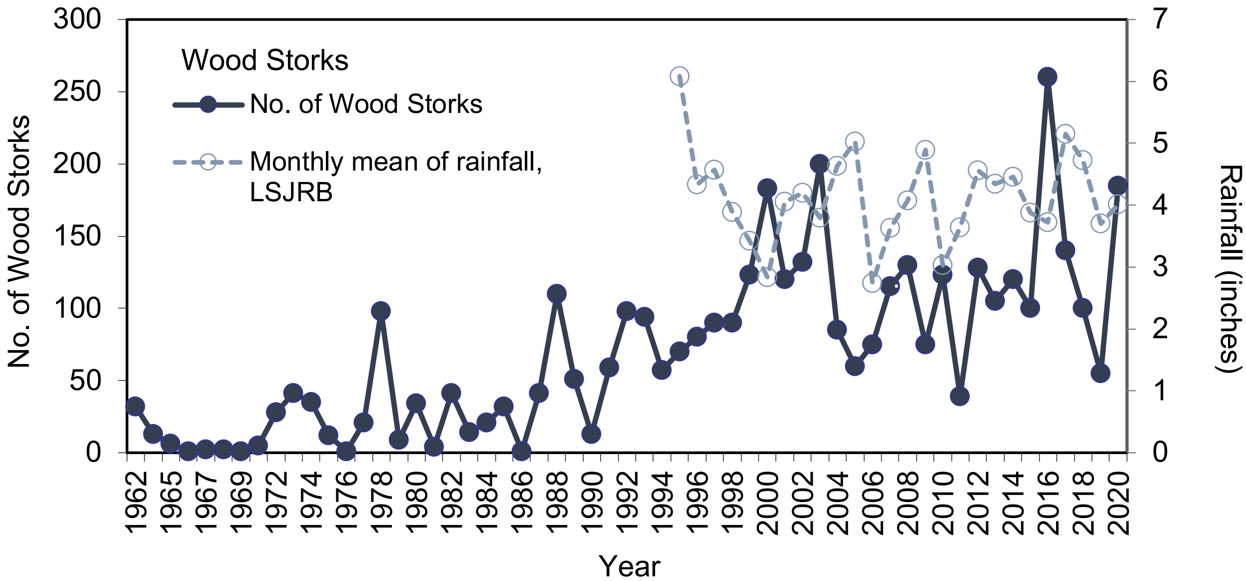


Figure 4.24 Long-term trend of the number of Wood Storks counted during winter bird surveys (1961-2020) and mean monthly rainfall in Jacksonville, FL (Source data: ***Audubon 2021****;* ***SJRWMD 2021c****; Appendix 4.4.3.A).*

Rainfall appears to affect wood stork status in several different ways. In the short term (1995-2020), rainfall for the LSJRB was negatively correlated with numbers of wood storks, but this was not significant (τ = -0.186; NS) (Figure 4.24). There was a decreasing trend in rainfall 1995-2000, which represents a prolonged period of severe drought (coincident with 1997 El Niño year). Wood storks surged in numbers as the drought deepened probably because of a concentration of prey as water levels fell. Then from 2000-2002, water levels became too low to support nesting or prey, causing a decline in numbers of wood storks (**Rodgers Jr et al. 2008a**). Rainfall increased again from 2000-2005 with averages approaching, and finally exceeding, the norm by 2005. During this period the numbers of wood storks continued to decline because of a natural lag in population and food supply. Then, numbers increased again by 2003. Although rainfall continued to increase, numbers of wood storks fell dramatically from 2003-2005. This was probably due to increased storm activity that damaged wood stork colonies, particularly in 2004 when four hurricanes skirted Florida. Also, higher water levels may have caused depressed productivity to breeding adults by dispersing available prey (**Rodgers Jr et al. 2008b**). Another drought ensued from 2005-2006 and rainfall declined at a faster rate than previously. As before, stork numbers began to increase initially. Then, from 2006-2009, rainfall continued to increase, and wood stork numbers declined. In 2010, following a prolonged cold winter, another cycle of drought began, and wood storks began to increase. Rainfall in the last few years increased close to normal levels again for the area and the wood stork population rebounded. However, in 2016 and early 2017, there was a severe drought which caused a large increase in wood storks. Then in late 2017, numbers fell sharply probably due to storm impacts to wood stork colonies from Hurricane Irma and subsequent storms in 2017/2018/2019/2020, including Hurricane Michael in October 2018, Dorian in August/September 2019, and in 2020 Hurricane Isaisa (July/August) and Eta (October /November) (see Appendix 4.1.7.1.E Rainfall, Hurricanes, and El Niño). Taking everything into account, the current **STATUS** of the Wood Storks is ***Satisfactory***, and the **TREND** is ***Improving***.

Rainfall data for LSJRB (1995-2020) was negatively correlated with Wood storks when party hours of effort were considered, but this was not significant (τ = -0.186; NS) (Figure 4.25).

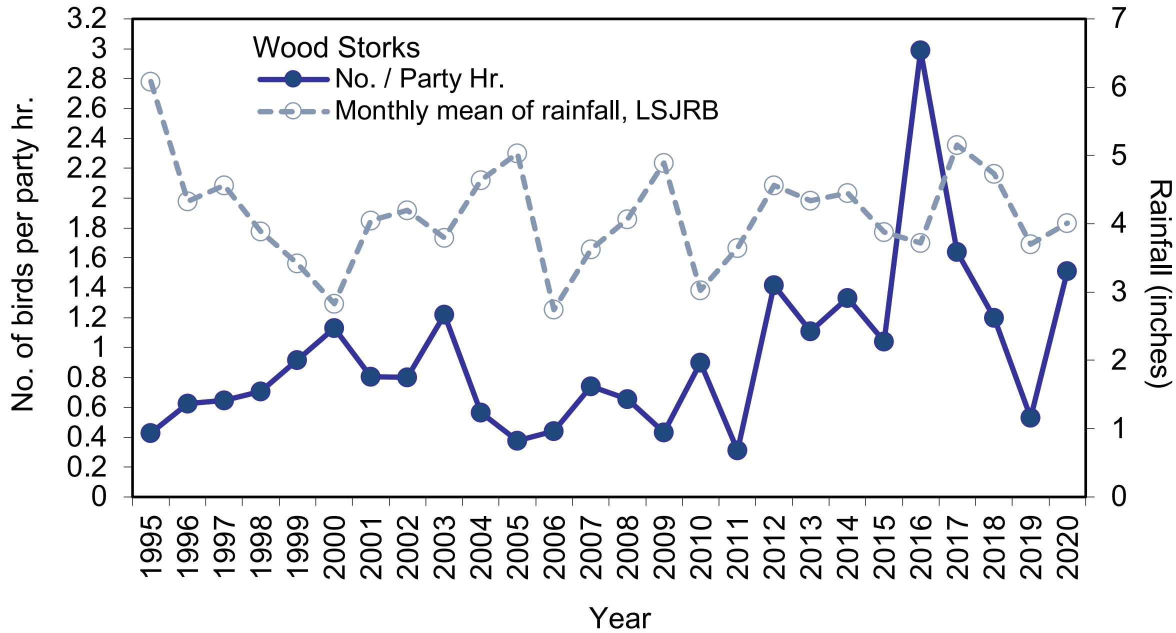


*Figure 4.25 Recent trends in the number of wood storks counted per party hour and mean monthly rainfall (1995-2020) in Jacksonville, FL (Source data: **Audubon 2021**; **SJRWMD 2021c**; Appendix 4.4.2.A).*

Brooks and Dean 2008 describe increasing wood stork colonies in northeast Florida as somewhat stable in terms of numbers of nesting pairs (Appendix 4.4.3.A). A press release by the USFWS (**Hankla 2007**) stated that the data indicate that the wood stork population as a whole is expanding its range and adapting to habitat changes and for the first time since the 1960s, that there had been more than 10,000 nesting pairs. For a map of the distribution of wood stork colonies and current breeding range in the southeastern US, see Figure 4.26.

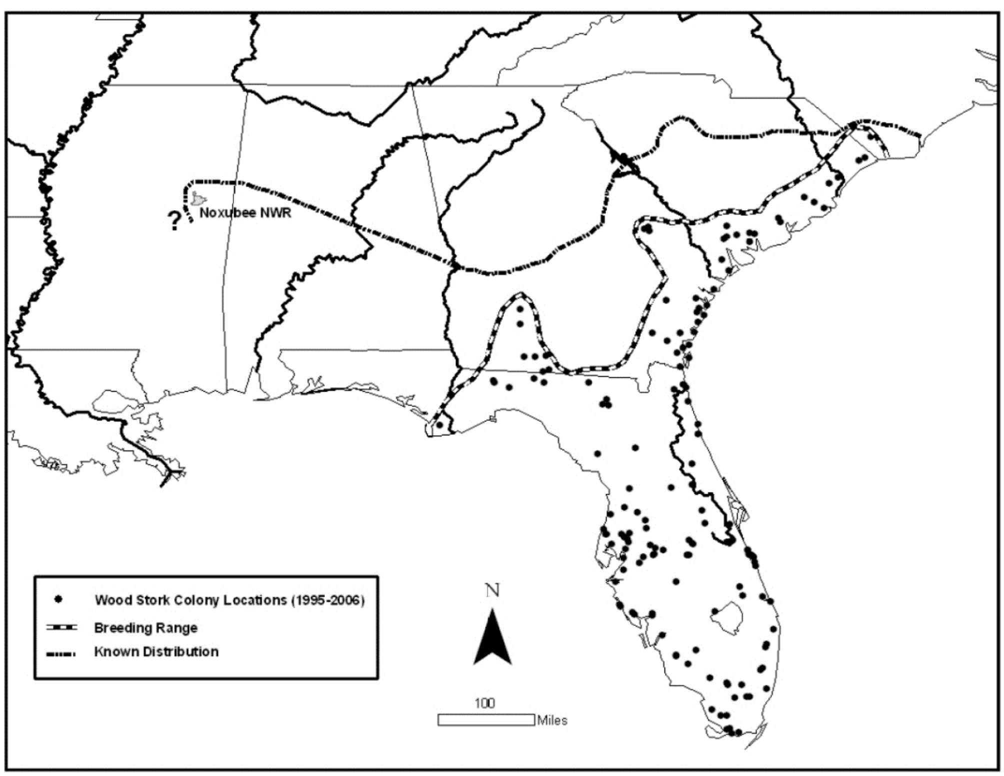


*Figure 4.26 Distribution of wood stork colonies and current breeding range in the southeastern US (**USFWS 2007c**).*

Rodgers Jr et al. 2008b made a comparison of wood stork productivity across colonies from different regions of Florida. Northern colonies in Florida exhibited greater productivity than those at more southerly latitudes. However, fledgling success was highly variable by year and colony. Local weather conditions and food resources were particularly important in determining nesting and fledgling success. Rainfall during the previous 12-24 months had a significant effect on fledging rates, as did both wetland and non-wetland habitats on fledging rate and colony size (**Rodgers Jr et al. 2010**).

In the LSJRB, there are several colonies of interest, three of these include: (1) Jacksonville Zoo and Gardens: This colony was formed in 1999 and has continued to persist strongly with growth leveling off from 2010 to 2016. In 2017, 2018, and 2019 there was a significant decline due to nest damage from storms, although numbers in 2019 show a slight rebound occurring. The Zoo group continues to have the highest number and productivity of birds in central and north Florida (**Rodgers Jr et al. 2008a**) (Figures 4.19 and 4.20; Appendix 4.4.3.B). It is considered one of the most important rookeries in Duval County (**Brooks 2021**). Donna Bear-Hull from the Jacksonville Zoo reported that the 4th year colony doubled in size from 40 breeding pairs (111 fledged chicks) in 2002 to 84 pairs (191 fledged chicks) in 2003. Since 2003, the colony's growth rate has slowed due to space limitations. Local adverse weather conditions (drought) that had an impact on the population and its food supply prevailed in 2005. As food supply was probably concentrated as water levels fell, the colony continued to grow, reaching a high of 117 pairs (267 fledged chicks) in 2006. Then in 2007, a crash occurred, and numbers of pairs declined to 47 (58 fledged chicks) following Hurricane Ernesto (August 24-September 1, 2006). In 2008, there was a rebound with the population almost doubling from the previous year to 86 pairs (181 fledged chicks) (**USFWS 2004; Bear-Hull 2021**). In 2009, the nesting and fledgling rates were similar 88 pairs, but 124 fledged chicks (**USFWS 2021c**). In 2010, the number of wood storks increased to 107 pairs and 278 fledged chicks following drought conditions. From 2011 to 2013, there was a significant decline in the numbers of fledglings to a low of 35 fledglings from 90 pairs in 2013, again after increased storm frequency. From 2010 to 2016 the population was close to carrying capacity and with stabilizing numbers of nests and reasonably consistent nest success rates (2016: 101 nests, 78% success rate; 2015: 91 nests, 81% success rate; 2014: 88 nests, 74% success rate; 2013: 90 nests, 30% success rate; 2012: 106 nests, 76% success rate). Severe drought and storms in 2017 (70 nests, 81% success rate) and 2018 (45 nests, 76% success rate) caused a significant decrease in nests and fledglings, although nest success rates were relatively unchanged. In 2019, there was a rebound with 69 nests and a 96% success rate, but in 2020, numbers decreased slightly with the impacts from further storms (**Bear-Hull 2021**).

In 2003, the zoo formed a conservation partnership with USFWS to monitor the birds/nests more closely (twice weekly). Since that time, the zoo has banded 380 chicks and 33 adults. In addition, four adults have been fitted with satellite monitoring tags. The 9 banded adults returned every year to the zoo site until 2007; some did not likely go to other rookeries. In 2021, 30 more chicks were banded. Satellite tracking data to date supports this foraging pattern, with adults feeding primarily on an estuarine prey base prior to nesting, switching to freshwater prey base during chick rearing, and then return to an estuarine diet after chick fledging and during the rest of the year (**Jacksonville Zoo 2021**).

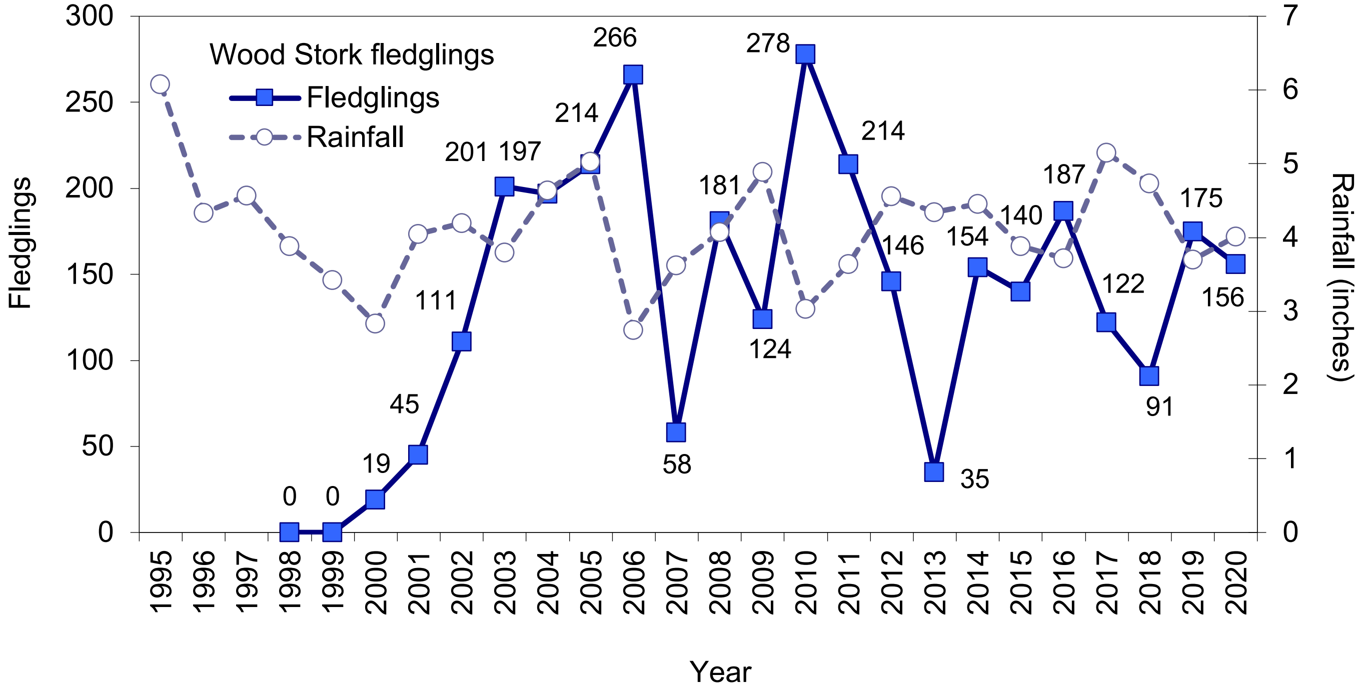


Figure 4.27 Number of wood stork nests at Jacksonville Zoo (2003-2020) (Source data: ***USFWS 2021c****;* ***SJRWMD 2021c****;* ***Bear-Hull 2021****).*

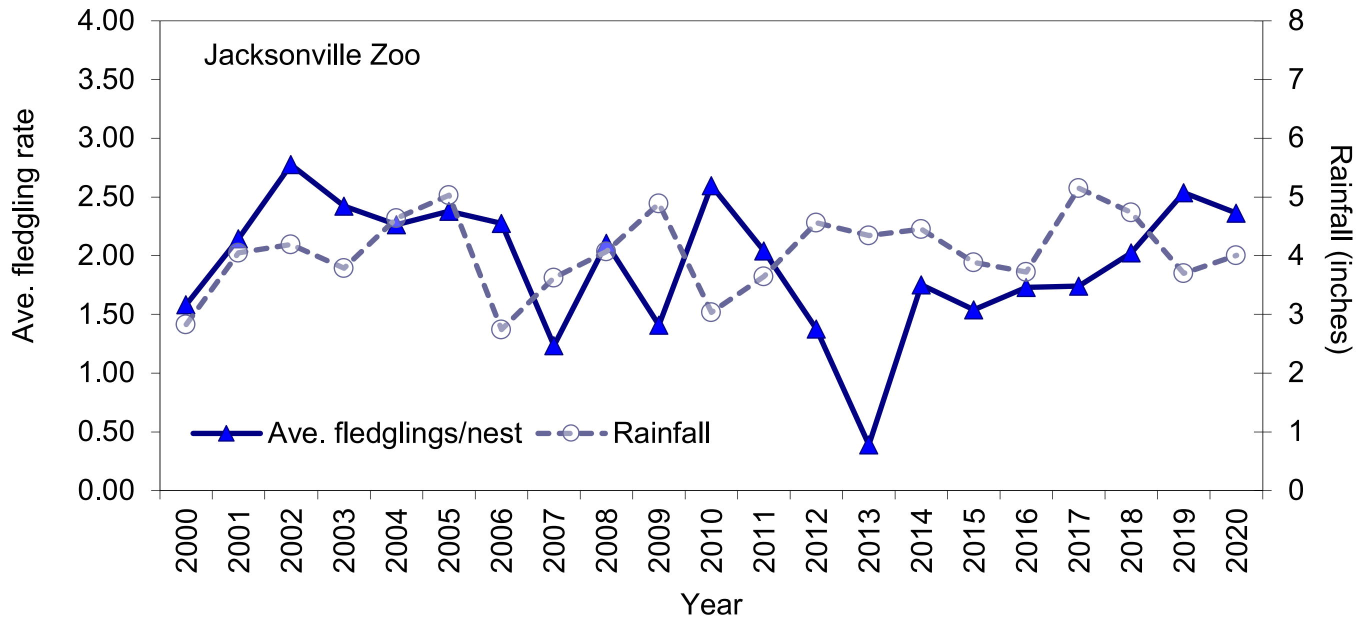


Figure 4.28 Wood stork productivity chicks/nest/year at Jacksonville Zoo (2003-2020) and mean monthly rainfall (Source data: ***USFWS 2005, 2007c; Rodgers Jr. 2011; Bear-Hull 2021****;* ***SJRWMD 2021c****;* ***USFWS 2021c****).*

(2) Dee Dot Colony: In 2005, the USFWS reported that there were over a hundred nests in this cypress swamp impounded lake in Duval County. However, the fledgling rate was low (1.51 chicks/nest in 2003, and 1.42 chicks/nest in 2004). Fledgling rates greater than two chicks/nest/year are considered acceptable productivity (**USFWS 2005**). Furthermore, the number of nests decreased from 118 in 2003 to 11 in 2007. This decline was probably due to nesting failure in 2003 caused by winds greater than about 20 mph and rain in excess of 1.5 inches/hr) (**Rodgers Jr et al. 2008b**; **Rodgers Jr et al. 2008a**). Fledgling rate improved from an average of 1.75 chicks/nest/year (2003-2005) to 2.11 chicks/nest/year in 2006 (**USFWS 2007c**). The rate then declined to 1.45 (2007) and rose back to 2.07 (2008) (**Rodgers Jr et al. 2008b**; **Rodgers Jr et al. 2008a**). Rainfall continued an upward trend; although the colony was active (determined by aerial survey), data on wood stork numbers were unavailable for the years 2010, 2012, and 2013 (Figures 4.29 and 4.30). In 2014, the colony consisted of 170 active wood stork nests, determined from aerial photographs, and in 2015, there were in excess of 130 nests. In 2016, 100 nest were

reported with 28% successful and 81 chicks fledged (2.85 chicks/nest). Increased storm activity in 2017 likely led to a significant decrease in nests, totaling 43, and a low fledgling rate of 1.03 chicks/nest. Nests increased to 121 (2018) with 97% successful and 72 nests in 2019, with 67% successful. (**Bear-Hull 2021**; **USFWS 2021c**).

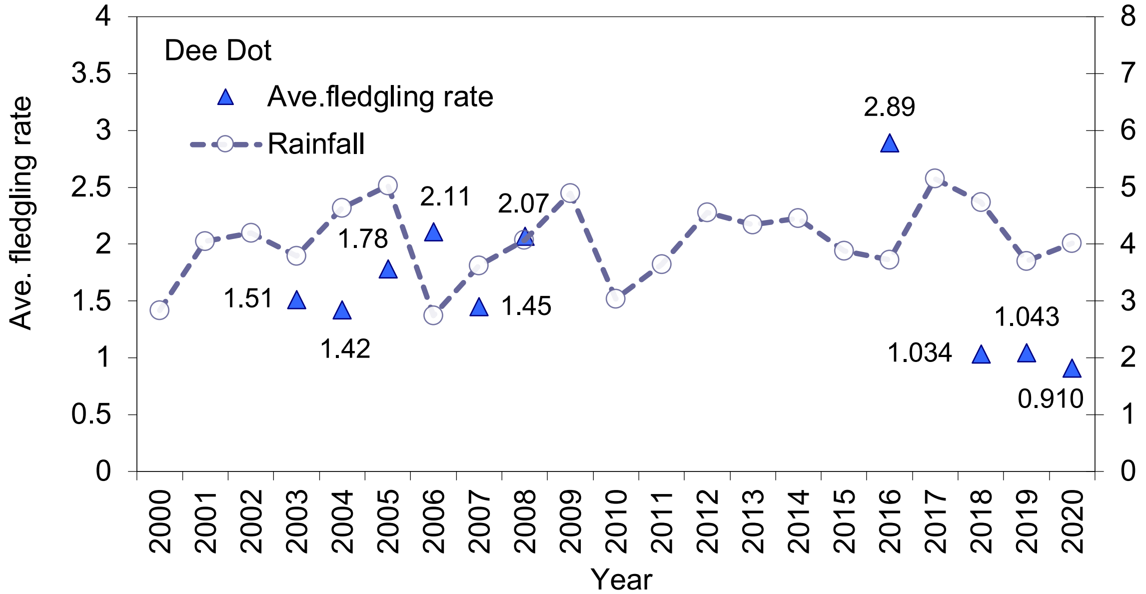


*Figure 4.29 Wood stork productivity (chicks/nest/year) at Dee Dot (2003-2008, 2016, 2018-2020) and mean monthly rainfall (2000-2020) (Source data: **USFWS 2005, 2007c; Rodgers Jr et al. 2008b; SJRWMD 2021c; USFWS 2021c; Bear-Hull 2021**).*

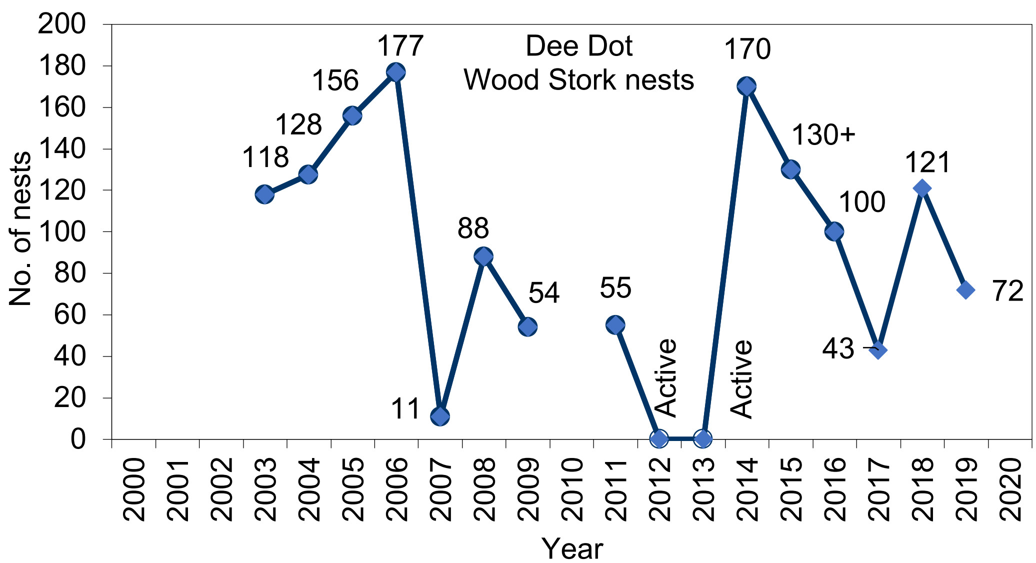


*Figure 4.30 Number of wood stork nests at Dee Dot (2003-2019) Note: there were no data for 2010, 2012, 2013 and 2020 (Source data: **Rodgers Jr et al. 2008a; Rodgers Jr et al. 2008b; USFWS 2021c; Bear-Hull 2021**).*

(3) Pumpkin Hill Creek Preserve State Park: This colony in Duval County had 42 nests in 2005 and 2008 (down from 68 in 2003) and fledgling rate averaged 1.44 chicks/nest/year in those years (**USFWS 2005**). Lack of rainfall during the breeding season (March to August) resulted in no water below the trees in 2004 that contributed to nest failures. Flooding following post-August 2004 hurricane season resulted in a return of breeding storks in 2005 (**Rodgers Jr et al. 2008a**). In 2009, the colony was described as being active, but no data were available (**Brooks 2021; USFWS 2021c**). The site was inactive during 2010 to 2016, and no data were available for 2017 to 2020 (Figures 4.31 and 4.32).

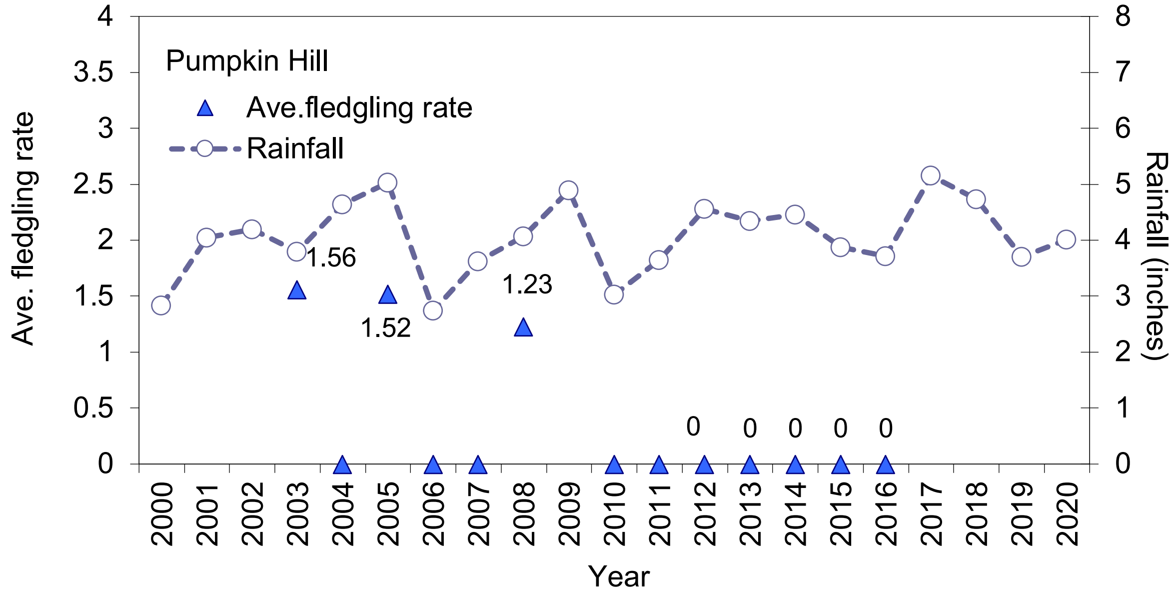


Figure 4.31 Wood stork productivity (chicks/nest/year) at Pumpkin Hill (2003-2016) and mean monthly rainfall. There are two colonies at this site, which is characterized by cypress-dominated domes. In 2004, the period 2006 to 2007, and from 2010-2016 no wood stork activity has been documented at this site (no data in 2017 to 2020). In 2009, the colony was described as being active, but no data was available (Source data: ***Rodgers Jr et al. 2008a; Rodgers Jr et al. 2008b; SJRWMD 2016b**, **2021c; USFWS 2021c**).*

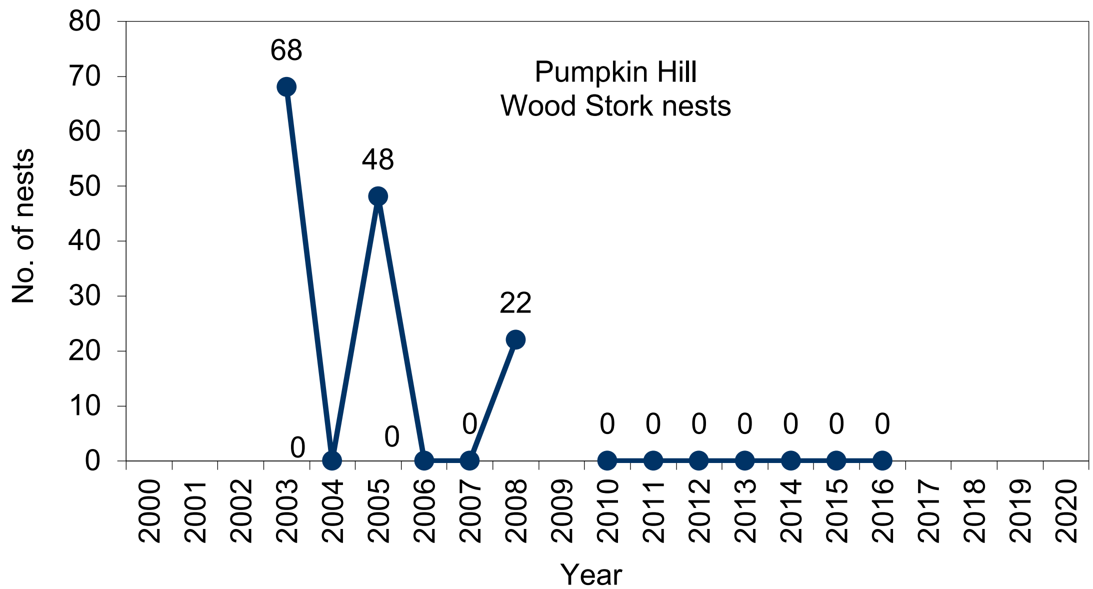


Figure 4.32 Number of wood stork nests at Pumpkin Hill (2003-2016). In 2004, the period 2006 to 2007, and from 2010-2016 no wood stork activity has been documented at this site (no data in 2017 to 2020). In 2009, the colony was described as being active, but no data was available. (Source data: ***Rodgers Jr et al. 2008a; Rodgers Jr et al. 2008b; USFWS 2016**, **2021c**).*

4.4.3.5. Future Outlook

Historically, the wood stork breeding populations were located in the Everglades, but now their range has almost doubled in extent and moved further north. The birds continue to be protected under the Migratory Bird Treaty Act and state laws. Although they are not as dependent on the Everglades wetlands, wetlands in general continue to need protection. Threats continue to exist such as contamination by pesticides, harmful algae blooms, electrocution from power lines and human disturbance such as road kills. Adverse weather events like severe droughts, thunderstorms, or hurricanes also threaten the wood stork colonies. The USFWS Wood Stork Habitat Management Guidelines help to address these issues. Continued monitoring is essential for this expanding and changing population (**USFWS 2007c**). The US Fish and Wildlife Service upgraded the status for wood storks from endangered to threatened because of the success of conservation efforts over the last 30 years (**USFWS 2016**).

4.4.4. Shortnose Sturgeon (Endangered)

Source: USFWS

4.4.4.1. Description

The shortnose sturgeon *(Acipenser brevirostrum)* is a native species historically associated with rivers along the east coast of US from Canada, south to Florida. The fish tend to be found in larger populations in more northerly rivers. The Shortnose sturgeon was listed as endangered in 1967. It is a semi-anadromous fish that swims upstream to spawn in freshwater before returning to the lower estuary, but not the sea. The species is particularly imperiled because of habitat destruction and alterations that prevent access to historical spawning grounds. The St. Johns River is dammed in the headwaters, heavily industrialized and channelized near the sea, and affected by urbanization, suburban development, agriculture, and silviculture throughout the entire basin. Initial research conducted by the National Marine Fisheries Service in the 1980s and 1990s culminated in the Shortnose Sturgeon Recovery and Management Plan of 1998 (**NMFS 1998; FWRI 2021f**; **NOAA 2021b**).

"Anadromous" fish live in the ocean, but return to freshwater to spawn.

4.4.4.2. Significance

There are no legal fisheries or by-catch allowances for shortnose sturgeon in US waters. Principal threats to the survival of this species include blockage of migration pathways at dams, habitat loss, channel dredging, and pollution. Southern populations are particularly at risk due to water withdrawal from rivers and ground waters and from eutrophication (excessive nutrients) that directly degrades river water quality causing loss of habitat. Direct mortality is known to occur from getting stuck on cooling water intake screens, dredging, and incidental capture in other fisheries (**NMFS 1998**).

4.4.4.3. Data Sources & Limitations

Information on shortnose sturgeon in literature is limited to a few captured specimens. Information sources included books, reports and web sites. Shortnose sturgeons have been encountered in the St. Johns River since 1949 in Big Lake George and Crescent Lake (**Scott 2003a**). Five shortnose sturgeons were collected in the St. Johns River during the late 1970s (**Dadswell et al. 1984**) and, in 1981, three sturgeons were collected and released by the FWC. All these captures occurred far south of LSJRB in an area that is heavily influenced by artesian springs with high mineral content. None of the collections was recorded from the estuarine portion of the system (**NMFS 1998**). From 1949-1999, only 11 specimens had been positively identified from this system. Eight of these captures occurred between 1977 and 1981. In August 2000, a cast net captured a shortnose sturgeon near Racy Point just north of Palatka. The fish carried a tag that had been attached in March 1996 by Georgia Department of Natural Resources near St. Simons Island, Georgia. During 2002/2003 an intensive sampling effort by researchers from the FWRI captured one 1.5 kg (3.3 lbs) specimen south of Federal Point, again near Palatka. As a result, FWRI considers it unlikely that any sizable population of shortnose sturgeon currently exists in the St. Johns River. In addition, the rock or gravel substrate required for successful reproduction is scarce in the St. Johns River and its tributaries. Absence of adults and marginal habitat indicate that shortnose sturgeons have not actively spawned in the system and that infrequent captures are transients from other river systems (**FWRI 2021f**; **NOAA 2021b**).

4.4.4.4. Current Status

The species is likely to be declining or almost absent in the LSJRB (**FWRI 2021f**). Population estimates are not available for the following river systems: Penobscot, Chesapeake Bay, Cape Fear, Winyah Bay, Santee, Cooper, Ashepoo Combahee Edisto Basin, Savannah, Satilla, St. Marys, and St. Johns River (Florida). Shortnose sturgeon stocks appear to be stable and even increasing in a few large rivers in the north but remain seriously depressed in others, particularly southern populations (**Friedland and Kynard 2004**).

4.4.4.5. Future Outlook

The Shortnose Sturgeon Recovery and Management Plan (**NMFS 1998**) identifies recovery actions to help reestablish adequate population levels for de-listing. Captive mature adults and young are being held at Federal fish hatcheries operated by the USFWS for breeding and conservation stocking.

4.5. Non-native Aquatic Species

4.5.1. Description

The invasion and spread of non-native, or "exotic," species is currently one of the most potent, urgent, and far-reaching threats to the integrity of aquatic ecosystems around the world (**NRC 1995**; **NRC 1996**; **NRC 2002**; **Ruckelshaus and Hays 1997**). Non-native species can simply be defined as "any species or other biological material that enters an ecosystem beyond its historic, native range" (**Keppner 1995**).

Protection from and management of aquatic species occurs at the federal and state levels. At the federal level, impairment by invasive species is not recognized under the Clean Water Act (**ELI 2008**). USACE in Jacksonville leads invasive species management with the Aquatic Plant Control Operations Support Center and the Removals of Aquatic Growth Program. The US Department of Agriculture Animal and Plant Health Inspection Services is charged with protection from invasive species (**ELI 2008**).

In Florida, management of invasive species is coordinated by Florida Fish and Wildlife Commission's Aquatic Plant Management Program. In 1994, Florida Department of Environment (DEP) included a TMDL water body impairment category of "WEED-exotic and nuisance aquatic plants density impairing water body" (**ELI 2008**).

However, DEP has yet to develop a TMDL for this category. FWC regulates import of vertebrate and invertebrate aquatic species, Florida Department of Agriculture and Consumer Services (FDACS) contributes to prevention of invasive species with importation regulation. Water management districts also contribute with control and restoration programs (**ELI 2008**). Non-profit organizations, such as the First Coast Invasive Working Group, organize invasive species removal events and education outreach.

4.5.2. Significance

The transport and establishment of non-native aquatic species in the St. Johns River watershed is significant due to a number of ecosystem, human health, social, and economic concerns.

4.5.2.1. Ecosystem Concerns

"Generalizations in ecology are always somewhat risky, but one must be offered at this point. The introduction of exotic (foreign) plants and animals is usually a bad thing if the exotic survives; the damage ranges from the loss of a few native competing species to the total collapse of entire communities" (**Ehrenfeld 1970**). The alarming increase in the number of documented introductions of non-native organisms is of pressing ecological concern (**Carlton and Geller 1993**). This concern is supported by the evidence that non-native species, within just years of introduction, are capable of breaking down the tight relationships between resident biota (**Valiela 1995**). Once introduced, a subset of these exotic species may encounter few (if any) natural pathogens, predators, or competitors in their new environment, and then may become invasive.

For example, the submerged aquatic plant *Hydrilla verticillata* is an invasive non-native species identified as the #1 aquatic weed in Florida. Native to Asia, hydrilla was likely introduced to Florida in the 1950s (**Simberloff et al. 1997**) and has spread through the Lower St. Johns River Basin since at least 1967 (**USGS 2015**). Even the smallest fragment of hydrilla can rapidly grow and reproduce into dense canopies, which are poor habitat for fish and other wildlife. Hydrilla is a superb competitor with native species by monopolizing resources and growing throughout months of lower light (**Gordon 1998**). Huge masses of hydrilla slow water flow, obstruct waterways, reduce native biodiversity, and create stagnant areas ideal for the breeding of mosquitoes (**McCann et al. 1996**).

Eutrophic conditions due to excessive nitrate conditions can contribute to proliferation of *H. verticillata* in historically oligotrophic waters (**Kennedy et al. 2009**). In an aquaria experiment with low and high nitrate treatments (0.2 and 1.0 mg nitrate per L, respectively), *H. verticillata* more than doubled its weight in the high nitrate treatment (547 g dry weight) as compared to the low nitrate treatment (199 g dry weight). By comparison, the native species *Sagittaria kurziana* and *Vallisneria americana* did not have a significant difference in weight despite the addition of nitrates. This study suggests that *H. verticillata* will outgrow native aquatic plants as nitrates continue to increase (**Kennedy et al. 2009**).

A number of non-native herbivorous fish are altering native ecosystems in the Lower St. Johns River. Many of these fish are common in the aquarium trade and include the Eurasian goldfish (*Carassius auratus*; which commonly becomes brown in the wild), Mozambique tilapia *(Oreochromis mossambicus)*, African blue tilapia *(Oreochromis aureus)*, South American brown hoplo *(Hoplosternum littorale)*, and a number of unidentified African cichlids *(Cichlidae spp.)* (**Brodie 2008**; **USGS 2015**). Additionally, several species of South American algae-eating catfish commonly known in the aquarium trade as "plecos," including the suckermouth catfish *(Hypostomus sp.)* and vermiculated sailfin catfish *(Pterygoplichthys disjunctivus)* appear to be established in the Lower St. Johns River (**USGS 2015**). As most aquarium enthusiasts know, "plecos" are extremely efficient algae eaters, and, when released into the wild, can have profound impacts on the native community of aquatic plants and animals. Recently, the vermiculated sailfin catfish has been eradicated from the Rainbow River following removal of 28 individuals by hand and spear, demonstrating that early removal of invasive species is possible (**Hill and Sowards 2015**).

Urbanization can contribute to the altering of flow regimes and water quality in the LSJRB (**Chadwick et al. 2012**) that may enable invasive organisms to survive. As compared to rural streams where the flow is typically intermittent, urban streams may have perennial flow due to irrigation, leaky sewage tanks and perhaps stormwater that was not diverted to retention ponds. The invasive clam *Corbicula fluminea* contributes significant biomass in two urban perennial streams (**Chadwick et al. 2012**). *Rangia cuneata* was also common on silt-sand substrates near Sixmile Creek and northward in the main river channel to near Cedar River (**Mason Jr 1998**).

4.5.2.2. Human Health Concerns

Non-native aquatic species can negatively affect human health. Some non-native microorganisms, such as blue-green algae and dinoflagellates, produce toxins that cause varying degrees of irritation and illness in people (**Hallegraeff et al. 1990**; **Hallegraeff and Bolch 1991**; **Stewart et al. 2006**). During the summer of 2005, large rafts of toxic algal scum from Lake George to the mouth of the St. Johns River in Mayport, Florida, brought headline attention to toxic bloom-forming algae. The organisms responsible for this bloom were two toxin-producing cyanobacteria (blue-green algae) species: the cosmopolitan *Microcystis aeruginosa* and the non-native *Cylindrospermopsis raciborskii* (**Burns Jr 2008a**). *C. raciborskii* has been recorded throughout tropical waters globally, but appears to be expanding into temperate zones as well throughout the US and the world (**Kling 2004**; **Jones and Sauter 2005**). *Cylindrospermopsis* may have been present in Florida since the 1970s; however, its presence in the St. Johns River Basin was not noted prior to 1994 (**Chapman and Schelske 1997**; **Phlips et al. 2002**; **SJRWMD 2005**). Genetic studies reveal strong genetic similarities between populations in Florida and Brazil, suggesting the two populations continually mix or came from the same source relatively recently (**Dyble et al. 2002**).

Cylindrospermopsis now appears to bloom annually each summer in the St. Johns River with occasionally very high concentrations in excess of 30,000 cells/mL (**Phlips et al. 2002**). During the intense bloom of 2005, the Florida Department of Health released a human health alert recommending that people avoid contact with waters of the St. Johns River, because the toxins can cause "irritation of the skin, eyes, nose and throat and inflammation in the respiratory tract" (**FDOH 2005**). This public health concern will likely continue to menace the Lower St. Johns River Basin in the foreseeable future, particularly when the water becomes warm, still, and nutrient-rich: conditions favorable to the formation of algal blooms.

4.5.2.3. Social Concerns

In general, many non-native species reproduce so successfully in their environment, that they create unsightly masses that negatively impact recreation and tourism. Such unsightly masses, as those created by water hyacinth *(Eichhornia crassipes)* or hydrilla *(Hydrilla verticillata)*, also shift the way we view and appreciate the aesthetic, intrinsic qualities of our aquatic ecosystems. The Cuban treefrog is also considered a nuisance due to their ability to clog pipes and invade ponds and residences (**Stepzinkski 2019**).

4.5.2.4. Economic Concerns

Excessive fouling by successful non-native species can lead to economic losses to industries. In 1986, the South American charrua mussel (*Mytella charruana)* caused extensive fouling at Jacksonville Electric Authority's Northside Generating Station on Blount Island, Jacksonville, Florida (**Lee 2012a**). The charrua mussel probably hitchhiked to the St. Johns River in the ballast water of a ship from South America and continues to persist in the area as evidenced by collections in Mayport, Marineland, and the Arlington area of Jacksonville as recently as 2008 (**Frank and Lee 2008**). Other non-native fouling organisms identified in the St. Johns River include the Asian clam *(Corbicula fluminea)*, Indo-Pacific green mussel *(Perna viridis)*, and Indo-Pacific striped barnacle *(Balanus amphitrite)*. Cleaning these fouling organisms from docks, bridges, hulls of boats and ships, and industrial water intake/discharge pipes is time-consuming and extremely costly.

Just as importantly, yet often overlooked, non-native species can be serious nuisances on a small scale. They foul recreational boats, docks, sunken ships, and sites of historical and cultural value. Clean-up and control of aquatic pests, such as the floating plant water hyacinth *(Eichhornia crassipes)*, can have high economic costs to citizens, not only in taxpayer dollars, but in out-of-pocket money as well.

4.5.3. Data Sources

Numerous online databases containing non-native species reports were queried. The most comprehensive listing of species is maintained in the Nonindigenous Aquatic Species (NAS) database of the United States Geological Service. Resources to investigate distributions of non-native plants include EDDMAPS, USDA, and the Atlas of Florida Vascular Plants. Additional records and information were obtained from agency reports, books, published port surveys, and personal communication data.

4.5.4. Limitations

We expect that many more non-native species are found within the LSJRB, but have not been recognized or recorded, either because they are *naturalized*, *cryptogenic*, or lack of the taxonomic expertise to identify foreign species, subspecies, or hybrids.

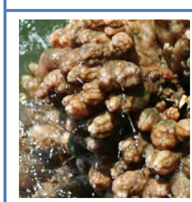
*A naturalized species is any non-native species that has adapted and grows or multiplies as if native (**Horak 1995**).*

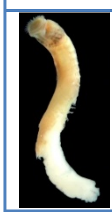
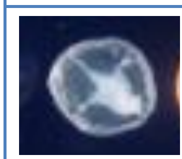
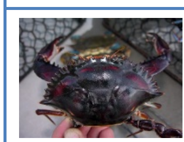
*A cryptogenic species is an organism whose status as introduced or native is not known (**Carlton 1987**).*

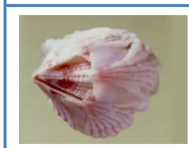
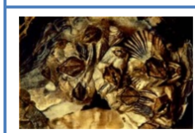
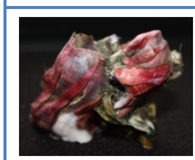
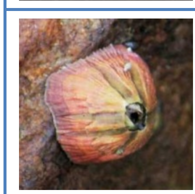
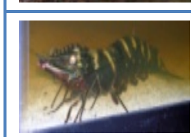
4.5.5. Current Status

The current **STATUS** is rated as ***Unsatisfactory***. Approximately 92 non-native aquatic species are documented and believed to be established in the LSJRB (Table 4.11). Non-native species recorded in the Lower Basin include floating or submerged aquatic plants, molluscs, fish, crustaceans, amphibians, jellyfish, mammals, reptiles, tunicates, bryozoans, and blue-green algae (Table 4.11). Freshwater species represent >65% of the species introduced into the LSJRB. Non-native aquatic species originate from the Central and South America, the Caribbean, Asia, and Africa (Table 4.11).

Table 4.11 Non-native aquatic species recorded in the Lower St. Johns River Basin.

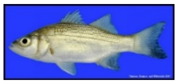
LIFEFORM	COMMON NAME	SCIENTIFIC NAME	HABITAT REALM	DATE	ORIGIN	PROBABLE VECTORS	COUNTY: FIRST REPORTED	REF
AMPHIBIANS								
	Cane toad *Photo: USGS NAS*	*Rhinella marina*	Freshwater, Brackish	1987	South and Central America	Humans, range expansion from South Florida populations	Clay, 1987	**USGS 2015**
	Cuban treefrog *Photo: USGS NAS*	*Osteopilus septentrionalis*	Terrestrial, Freshwater (springs, lakes, ponds)	1991	Caribbean	Dispersing northward from S. Florida populations, floating vegetation/debris, humans, vehicles, bulk freight/cargo, plant or parts of plants	Clay, 1991; Duval, 2002; Flagler, 2004; St. Johns, 2006; Volusia, 2012	**CISEH 2014; USGS 2015**
TUNICATES								
	Pleated (or rough) sea squirt *Photo: SERTC/SC DNR*	*Styela plicata*	Marine	1940	Indo-Pacific? This species is now found in tropical and warm-temperate oceans around the world.	Ship/boat hull fouling, ship ballast water/sediment, importation of mollusk cultures	Offshore Jacksonville, 1940	**De Barros et al. 2009; GBIF 2012d**

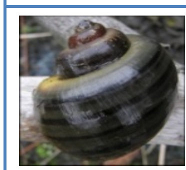
ECTOPROCTS - BRYOZOANS								
	Brown bryozoan	*Bugula neritina*	Marine, Brackish	mid-1900s.	Native range is unknown - probably Mediterranean Sea (1758 record).	Ship/boat hull fouling		**Eldredge and Smith 2001; NEMESIS 2014**
		Celleporaria pilaefera	Marine	2001	Indo-Pacific	Ship/boat hull fouling, aquaculture	Duval (SJR), 2001	**McCann et al. 2007; NEMESIS 2014**
		Arbopercula bengalensis	Marine	2001	India and tropical, subtropical coast of China		Duval (SJR), 2001	**McCann et al. 2007; NEMESIS 2014**
		Hippoporina indica	Marine	2001	Western Pacific	Ship/boat hull fouling	Duval (SJR), 2001	**McCann et al. 2007; NEMESIS 2014**
		Sinoflustra annae	Marine	2001	Indo-Pacific		Duval (SJR), 2001	**McCann et al. 2007; NEMESIS 2014**
POLYCHAETE								
		*Ficopomatus uschakovi*	Marine	2002	Indo-Pacific	Ship/boat hull fouling, ballast water	Duval (SJR), 2002	**NEMESIS 2014**
		*Hydroides diramphus*	Marine	2002	Western Atlantic and/or Indo-Pacific	Ship/boat hull fouling, ballast water	Duval (Mayport), 2002	**NEMESIS 2014**
JELLYFISH								
	Freshwater jellyfish *Photo: USGS NAS*	*Craspedacusta sowerbyi*	Freshwater (ponds, lake)	1980	Asia	Aquaculture stock, other live animal, plant or parts of plants	Duval, 1999; Putnam, 1980	**USGS 2015**
CRUSTACEANS								
	Bocourt swimming crab *Photo: Big Bend Brian*	*Callinectes bocourti*	Marine, Brackish	2002	Caribbean and South America	From the Caribbean via major eddies in Gulf Stream or southern storm events	Duval, 2002; Flagler, 2014	**CISEH 2015; USGS 2015**
	Indo-Pacific swimming crab *Photo: SC DNR*	*Charybdis hellerii*	Marine-offshore	2003	Indo-Pacific	Ship ballast water/sediment, or drift of juveniles from Cuba	Duval, 2003	**USGS 2015**

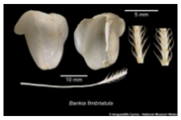
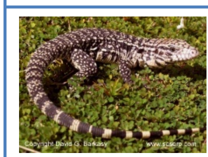
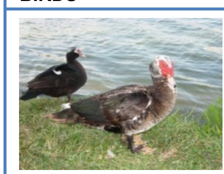
	Green porcelain crab *Photo: D. Knott*	*Petrolisthes armatus*	Marine, Brackish	Unknown	Caribbean and South America	Natural range expansion, ship ballast water/sediment, importation of mollusk cultures		**Power et al. 2006**
	Slender mud tube-builder amphipod *Photo: VIMS*	*Corophium lacustre*	Freshwater, Brackish	1998	Europe and Africa	Ship ballast water/sediment from Europe	St. Johns River, 1998	**Power et al. 2006; GBIF 2012b**
	Skeleton shrimp *Photo: D. Knott*	*Caprella scaura*	Marine	2001	Indian Ocean	Ship/boat hull fouling, ship ballast water/sediment	St. Johns River, 2001	**Foster et al. 2004; GBIF 2012a**
	Asian tiger shrimp *Photo: David Scott SERTC*	*Penaeus monodon*	Marine	2008	East Africa, South Asia, Southeast Asia, the Philippines, and Australia	Accidental release	Duval, 2008; Putnam, 2013	**USGS 2015**
	Wharf roach *Photo: Ruppert and Fox (1998)*	*Ligia exotica*	Marine	Unknown	Northeast Atlantic and Mediterranean Basin	Bulk freight/cargo, ship ballast water/sediment, shipping material from Europe		**Power et al. 2006**
	Striped barnacle *Photo: A. Cohen*	*Balanus amphitrite*	Marine	Unknown	Indo-Pacific	Ship/boat hull fouling		**Power et al. 2006**
	Triangular barnacle *Photo: D. Elford*	*Balanus trigonus*	Marine	Unknown	Indo-Pacific	Ship/boat hull fouling		**GSMFC 2010**
	Barnacle *Photo: C. Baike*	*Balanus reticulatus*	Marine	Unknown	Indo-Pacific	Ship/boat hull fouling		**GSMFC 2010**
	Titan acorn barnacle *Photo: H. McCarthy*	*Megabalanus coccopoma*	Marine	2004	Pacific Ocean	Ship/boat hull fouling	Duval, 2004; Mayport, 2008	**Frank and Lee 2008**
	Mediterranean acorn barnacle *Photo: H. McCarthy*	*Megabalanus antillensis* (*also known as M. tintinnabulum)*	Marine	Unknown	Europe (Mediterranean Sea)	Ship/boat hull fouling		**Masterson 2007; McCarthy 2011**
	Asian tiger shrimp *Photo: M. Watkins, FWRI-Jacksonville*	*Penaeus monodon*	Marine, Brackish	2008	Australasia	Aquaculture stock	Duval, 2008; St. Johns, 2011; Volusia, 2010	**CISEH 2014; USGS 2015**

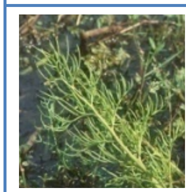
FISH

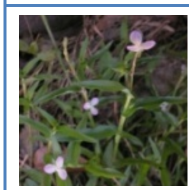
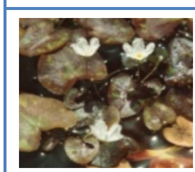
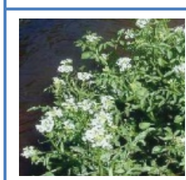
	Lionfish *Photo: A. Baeza*	*Primarily Pterois volitans (red lionfish) with a small number of Pterois miles (devil firefish)*	Marine-offshore	2001	Indo-Pacific	Humans: aquarium releases or escapes	Offshore Jacksonville, 2001	**USGS 2015**
	Goldfish *Photo: USGS NAS*	*Carassius auratus*	Freshwater	1974	Eurasia	Intentional release, ornamental purposes, stocking, aquarium trade, escape from confinement, landscape/fauna "improvement"	Clay, 1991; Putnam, 1974	**USGS 2015, 2018**
	Grass carp *Photo: USGS NAS*	*Ctenopharyngodon idella*	Freshwater	2007	Eastern Asia	Intentional, biological control of vegetation	Clay, 2007; Duval, 2015	**USGS 2015**
	Unidentified cichlids *Photo: USGS NAS*	*Cichlidae spp.*	Freshwater	2001-2006	Africa	Humans		**GSMFC 2010**; **Brodie 2008**; **USGS 2015**
	Blue tilapia *Photo: USGS NAS*	*Oreochromis aureus*	Freshwater (pond, lake)	1986	Europe and Africa	Humans: intentional fish stocking	Clay, 1991; Duval, 1984; Putnam, 1984; St. Johns 1986	**GSMFC 2010**; **Brodie 2008**; **USGS 2015**
	Mozambique tilapia *Photo: USGS NAS*	*Oreochromis mossambicus*	Freshwater Brackish	2001-2006	Africa	Humans: stocked, intentionally released, escapes from fish farms, aquarium releases		**GSMFC 2010**; **Brodie 2008**; **USGS 2015**
	Nile Tilapia	*Oreochromis niloticus*	Freshwater	2020	Tropical and subtropical Africa, Middle East	Escapes from fish farms,	Duval, 2020	**USGS 2021**
	Unidentified tilapia *Photo: USGS NAS*	*Tilapia spp.*	Freshwater (pond)	2001-2006	Africa	Humans		**GSMFC 2010**; **Brodie 2008**
	Unidentified Pacu *Photo: USGS NAS*	*Colossoma or Piaractus sp.*	Freshwater, Brackish (tributary, creek)	1989	South America	Aquaculture stock (fish farm escapes or releases), humans (aquarium releases)	Duval, 1989	**USGS 2015**
	Brown Hoplo *Photo: USGS NAS*	*Hoplosternum littorale*	Freshwater	2005	South America	Humans	Duval, 2005; Flagler, 2008; Putnam 2008	**CISEH 2015; USGS 2015**
	Wiper (Hybrid Striped Bass) (Whiterock = *female striped bass x male white bass,* Sunshine Bass = *male striped bass x female white bass*) *Photo: T. Pettengill*	*Morone chrysops x saxatilis (Artificial hybrid between the white bass and the striped bass)*	Freshwater (pond, lake), Brackish, Marine	1992	Artificial Hybrid	Humans: intentional fish stocking	Duval and Clay, 1992	**USGS 2015**

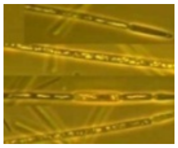
	White bass *Photo: Thomas, Bonner, and Whiteside*	*Morone chrysops*	Freshwater, Marine	1980	Northern and Central USA	Intentional introduction	Putnam, 1980	**USGS 2015**
	Unidentified armored catfish *Photo: USGS NAS*	*Loricariidae spp.*	Freshwater	2001-2006.	South and Central America	Aquaculture stock (fish farm escapes or releases), humans (aquarium releases)		**FWRI 2006**; **Brodie 2008**
	Suckermouth catfish *Photo: L. Smith*	*Hypostomus sp.*	Freshwater	1974, 2003	South and Central America	Aquaculture stock (fish farm escapes or releases), humans (aquarium releases)		**USGS 2015**
	Southern sailfin catfish *Photo: K.S. Cummings*	*Pterygoplichthys anisitsi*	Freshwater (river)	2007	South America	Humans: likely aquarium release	St. Johns, 2007	**USGS 2015**
	Vermiculated sailfin catfish *Photo: USGS NAS*	*Pterygoplichthys disjunctivus*	Freshwater (river)	2003	South America	Aquaculture stock (fish farm escapes or releases), humans (aquarium releases)	Duval, 2015; Putnam, 2003	**USGS 2015, 2018**
	Threadfin shad *Photo: USGS NAS*	*Dorosoma petenense*	Freshwater (river)	1941	Belize	Intentional stock	Clay, 1948; Duval, 1949; Putnam, 1941; St. Johns, 1985	**USGS 2015**
	Orinoco sailfin catfish	*Pterygoplichthys multiradiatus*	Freshwater	2009	Tropical America	Aquaculture stock (fish farm escapes and/or releases)	Duval, 2013; Putnam, 2009	**USGS 2015**
	Walking catfish *Photo: USGS NAS*	*Clarias batrachus*	Freshwater	2015	Southeastern Asia	Aquaculture stock (fish farm escapes and/or releases)	Clay, 2015	**USGS 2015**
	Flathead catfish *Photo: Garold Sneegas*	*Pylodictis olivaris*	Freshwater	1979	Northern and Central USA	Introductions	Duval, 1979	**USGS 2015**
	Redtail catfish *Photo: Monika Betley commons.wikimedia. or*	*Phractocephalus hemioliopterus*	Freshwater, Brackish	2007	Tropical America	Humans (aquarium releases)	Clay, 2014	**News4JAX 2015; USGS 2015**
MAMMALS								
	Nutria *Photo: USGS NAS*	*Myocaster coypus*	Freshwater (retention pond, drainage ditch), Terrestrial	1957	South America	Humans: escaped or released from captivity	Clay, 2017; Duval, 1963; Putnam, 1957	**CISEH 2014; USGS 2015**, 2018
	Capybara *Photo: USGS NAS*	*Hydrochoerus hydrochaeris*	Freshwater	2015	South America	Pet escapee	Clay, 2015	**USGS 2016**

MOLLUSCS								
	Asian clam *Photo: USGS NAS*	*Corbicula fluminea*	Freshwater (stream, lake)	1990	Asia and Africa	Humans, live seafood, bait, aquaculture stock, water	Clay, 2006; Duval, 2003; Flagler, 2008; Putname, 2008; Volusia 1990	**Frank and Lee 2008; Lee 2008; CISEH 2014; USGS 2015**
	Charua mussel *Photo: H. McCarthy*	*Mytella charruana*	Marine	1986	South America	Ship ballast water/sediment	Duval, 1986; Flagler, 2006	**Boudreaux and Walters 2006; Power et al. 2006; Frank and Lee 2008; Spinuzzi et al. 2012; CISEH 2014; USGS 2015**
	Green mussel *Photo: H. McCarthy*	*Perna viridis*	Marine, Brackish (river)	2002	Indo-Pacific	Ship ballast water/sediment, ship/boat hull fouling, humans	Duval, 2003; St. Johns, 2009; Volusia, 2002	**Power et al. 2006; Frank and Lee 2008; Spinuzzi et al. 2012**; **CISEH 2015**
	Paper pondshell *Photo: B. Frank*	*Utterbackia imbecillis*	Freshwater (lake)	1990	North America: Native in Mississippi River and Great Lakes	Other live animal, plant or parts of plants, ship/boat	Duval, 1990	**Frank and Lee 2008; Lee 2008**
	Red-rim melania *Photo: B. Frank*	*Melanoides tuberculata*	Freshwater (river)	1976	Asia and Africa	Other live animal, plant or parts of plants, ship/boat	Duval, 1976; Volusia, 2005	**Frank and Lee 2008; Lee 2008; CISEH 2014; USGS 2015**
	Fawn melania *Photo: B. Frank*	*Melanoides* cf. *turricula*	Freshwater	2006	North America: Native in western US and Canada	Other live animal, plant or parts of plants, ship/boat	Duval, 2006; St. Johns, 2006	**Frank and Lee 2008**
	Spiketop applesnail *Photo: B. Frank*	*Pomacea diffusa*	Freshwater (pond, drainage ditch)	2006	South America	Humans: probable aquarium releases	Duval, 2006; Clay, 2011	**Rawlings et al. 2007; Frank 2008; CISEH 2014**
	Channeled applesnail *Photo: Georgia DNR*	*Pomacea canaliculata*	Freshwater (retention pond)	2005	South America	Humans: probable aquarium releases	Duval, 2005; St. Johns, 2005	**Rawlings et al. 2007; Frank 2008; CISEH 2014; USGS 2015**
	Island applesnail *Photo: B. Frank*	*Pomacea (maculatum) insularum*	Freshwater (lake, creek, drainage ditch, river)	2005	South America	Humans: probable aquarium releases	Duval, 2005; St. Johns, 2005; Volusia, 2005	**Rawlings et al. 2007; Frank 2008; CISEH 2014; USGS 2015**
	Mouse-ear marshsnail *Photo: B. Frank*	*Myosotella myosotis*	Marine	Unknown	Europe	Bulk freight/cargo, ship ballast water/sediment,		**Frank and Lee 2008**

	Striped falselimpet *Photo: B. Frank*	*Siphonaria pectinata*	Marine (Mayport), Brackish (Sisters Creek)	2008	Europe and Africa (Mediterranean Sea)	Bulk freight/cargo, ship ballast water/sediment, ship/boat hull fouling, humans	Duval 2008; Mayport 2011	**Frank and Lee 2008; McCarthy 2008**
	Fimbriate shipworm *Photo: A. Cymru (Nat'l Museum of Wales)*	*Bankia fimbriatula*	Marine	Unknown	Pacific?	Ship/boat hull fouling, humans		**Frank and Lee 2008**
	Striate piddock shipworm *Photo: J. Wooster*	*Martesia striata*	Marine	Unknown	Indo-Pacific?	Ship/boat hull fouling, humans		**Frank and Lee 2008**
	Gulf Wedge Clam *Photo: B. Frank*	*Rangia cuneata*	Brackish	Present in Atlantic east coast Pleistocene deposits; First live Atlantic record in 1946.	Prior to 1946, native range was considered Gulf Coast of northern FL to TX.	Possible vectors: transplanted seed oysters, oyster shipments, ballast water		**Carlton 1992; Foltz et al. 1995; Verween et al. 2006; Carlton 2012; GBIF 2012c; Lee 2012b; NEMESIS 2014**
REPTILES								
	Red-eared slider *Photo: USGS NAS*	*Trachemys scripta elegans*	Freshwater (drainage ditch), Brackish	1991	North America: US midwestern states to northeastern Mexico	Humans: pet releases and escapes	Duval, 1991; Clay, 2012; Volusia, 2000; St. Johns, 2019	**CISEH 2014; USGS 2015, 2019b**
	Razorback Musk Turtle *Photo: R.C. Thomson*	*Sternotherus carinatus*	Freshwater (drainage ditch) Brackish	1958	Native to 6 states: statewide in LA, southern MS, southern AR, southeastern OK, eastern TX, small portion of southwestern AL	Humans: pet releases and escapes	Putnam, 1958	**Lindeman 2008; Krysko et al. 2011; USGS 2015**
	Black and White Tegu	*Tupinambis merianae*		2012			Duval, 2013; Volusia, 2012	**CISEH 2015; JHS 2014**
	Peters's rock agama	*Agama picticauda*		2007	East Africa	Pet release	Duval, 2007; Flagler, 2017; St. Johns, 2017	**EDDMapS 2021**
BIRDS								
	Muscovy duck *Photo: FWC*	*Cairina moschata*	Freshwater	1967	Central and South America	Humans: pet releases and escapes	Clay, 1986; Duval, 1991; Flagler, 1991; Putnam, 1991; St. Johns, 1991; Volusia, 1991	**CISEH 2014; FWC 2014c**
AQUATIC PLANTS								
	Alligator-weed *Photo: USGS NAS*	*Alternanthera philoxeroides*	Freshwater	1983	South America	Ship ballast water/sediment	Duval, 1984; Clay, 1983; Flagler, 1984; Putnam, 1984; St. Johns, 1984; Volusia, 1984	**McCann et al. 1996; USDA 2013; CISEH 2014; USGS 2015**

	Para grass *Photo: F. & K. Starr*	*Urochloa (Brachiaria) mutica*	Freshwater	2003	Africa	Humans: intentional release for agriculture	Flagler, 2009; Putnam, 2003; Volusia, 2004	**CISEH 2015; McCann et al. 1996; FCCDR 2008; USGS 2015**
	Water spangles *Photo: IFAS Univ. of Florida*	*Salvinia minima*	Freshwater (lakes, ponds)	1940	South and Central America	Ship ballast water/sediment, humans, aquarium trade	Clay, 1982; Duval, 1949; Flagler, 1940; Putnam 1940; St. Johns, 1982; Volusia, 1930	**McCann et al. 1996; CISEH 2014; USGS 2015**
	Hydrilla *Photo: USGS NAS*	*Hydrilla verticillata*	Freshwater (lake, creek, river)	1967	Asia	Debris associated with human activities, ship/boat, aquarium trade, garden waste disposal	Duval, 1982; Clay, 1967; Flagler, 2010; Putnam, 1967; St. Johns, 1967; Volusia, 2007	**McCann et al. 1996; CISEH 2014; USGS 2015**
	Water-hyacinth *Photo: USGS NAS*	*Eichhornia crassipes*	Freshwater (pond, lake, ditch, canal, river)	1890	South America	Humans, aquarium trade, garden escape	Duval, 1982; Clay 1900; Flagler, 1982; Putnam, 1890; St. Johns, 1900; Volusia 1963	**McCann et al. 1996; CISEH 2014; USGS 2015**
	Water-lettuce *Photo: USGS NAS*	*Pistia stratiotes*	Freshwater	1766	South America	Ship ballast water/sediment	Duval, Clay, 1982; Flagler, 2003; Putnam, 1982; St. Johns, 1982; Volusia, 1766	**CISEH 2015; McCann et al. 1996; FCCDR 2008; USGS 2015**
	Brazilian waterweed *Photo: USGS NAS*	*Egeria densa*	Freshwater	1969	South America	Humans: accidental aquarium releases, intentional release for control of mosquito larvae	Duval, 1995; Putnam, 1969; St. Johns, 1983; Volusia, 1990	**McCann et al. 1996; FCCDR 2008; CISEH 2014; USGS 2015**
	Watersprite *Photo: A. Murray*	*Ceratopteris thalictroides*	Freshwater	1984	Australasia	Humans	Duval, 2010; Clay, 2002; Flagler, 1990; Putnam, 1990; St. Johns, 1984; Volusia; 2014	**CISEH 2015; McCann et al. 1996; FCCDR 2008; USGS 2015**
	Wild taro *Photo: K. Dressler*	*Colocasia esculenta*	Freshwater (ditch, stream, lakeside, floodplain swamp, baygall)	1971	Africa	Humans	Duval, 2006; Clay, 1985; Flagler, 2003; Putnam, 1971; St. Johns, 1999; Volusia, 1995	**McCann et al. 1996; CISEH 2014; USGS 2015**
	Miramar weed	*Hygrophila polysperma*	Freshwater	2006	East Indies, India, Malaysia, Taiwan	Aquarium trade	Duval, 2006	**FLEPPC 2016; USGS 2016**
	Umbrella flatsedge	*Cyperus involucratus*	Freshwater	1984	Africa	Escaped cultivation	Duval, 2010; Clay, 1984	**CISEH 2016; Langeland et al. 2008; USGS 2016**

	Cuban bulrush	*Cyperus blepharoleptos*	Freshwater	1982	South America and West Indies	Ship ballast, migratory birds	Clay, 2002; Duval, 2004; Flagler, 1982; Putnam 1988; St. Johns, 1982; Volusia, 1984;	**CISEH 2016; USGS 2018**
	Papyrus	*Cyperus papyrus*	Freshwater, Brackish	2011			Putnam, 2011	**USGS 2018**
	Eurasian watermilfoil *Photo: USGS NAS*	*Myriophyllum spicatum*	Freshwater	1999	Europe, Asia, and Northern Africa	Aquarium, aquatic nursery trade, intentional	Flagler, 1999	**USGS 2018**
	Tropical American watergrass	*Luziola subintegra*	Freshwater	2016			St. Johns, 2018	**USGS 2019b**
	Large-flower primrose-willow	*Ludwigia grandiflora*	Freshwater	1998	South America	Humans	Clay, 1998 Duval, 2020; St. Johns, 2020	**USGS 2021**
	West Indian marsh grass	*Hymenachne amplexicaulis*	Freshwater	2008	West Indies, tropical Central and South America	Humans and/or migratory birds, forage	St. Johns, 2008	**Diaz et al. 2015** **USGS 2016**
	Uruguay water-primrose *Photo: Washington State Noxious Weed Control Board*	*Ludwigia uruguayensis*	Freshwater	1998	South America	Humans	Clay, 1998	**McCann et al. 1996; CISEH 2014; USGS 2015**
	Marsh dewflower *Photo: L. Lee*	*Murdannia keisak*	Freshwater	1960	Asia	Humans	Duval, 1960	**CISEH 2014; USGS 2015**
	Parrot-feather *Photo: USGS NAS*	*Myriophyllum aquaticum*	Freshwater (slough)	1940	South America	Humans	Clay, 1940; Duval, 1983; St. Johns, 1983; Flagler, 1940; Putnam, 1983; Volusia, 1986	**McCann et al. 1996; FCCDR 2008; CISEH 2014; USGS 2015**
	Brittle naiad *Photo: USGS NAS*	*Najas minor*	Freshwater (lake)	1983	Eurasia	Humans	Putnam, 1983	**McCann et al. 1996; FCCDR 2008; CISEH 2014; USGS 2015**
	Crested floating-heart *Photo: C. Jacono*	*Nymphoides cristata*	Freshwater	2010	Asia	Humans	St. Johns, 2010	**CISEH 2015; FCCDR 2008; USGS 2015**
	Water-cress *Photo: WI DNR*	*Nasturtium officinale*	Freshwater	1995	Eurasia	Humans	Duval, 1995; Clay, 1995; Putnam, 1995; St. Johns	**McCann et al. 1996; FCCDR 2008; CISEH 2014; USGS 2015**

	Torpedo grass *Photo: V. Ramey*	*Panicum repens*	Freshwater (adjacent to waterways)	2002	Europe	Humans	Duval, 2004; Clay, 2005; Flagler, 2003; Putnam, 2002; St. Johns, 2003; Volusia, 2003	**McCann et al. 1996; FCCDR 2008; CISEH 2014; USGS 2015**
BLUE-GREEN ALGAE								
	Blue-green alga *Photo: Umwelt Bundes Amt*	*Cylindrospermopsis raciborskii*	Freshwater	1950s First ID in the US; 1995 First ID in Florida	South America (high degree of genetic similarity with specimens from Brazil)	Humans, other live animal (digestion/ excretion), aquarium trade, ship ballast water/sediment, ship/boat, water (interconnected waterways)		**Dyble et al. 2002**

4.5.6. *Trend*

The cumulative number of non-native aquatic species introduced into the LSJRB has been increasing at an exponential rate since records were kept prior to 1900 (Figure 4.33). This trend is the reason that the category is assigned a CONDITIONS WORSENING status – indicating that non-native species are contributing to a declining status in the health of the St. Johns River Lower Basin. For this reason, the current **STATUS** has been assigned as ***Unsatisfactory***.

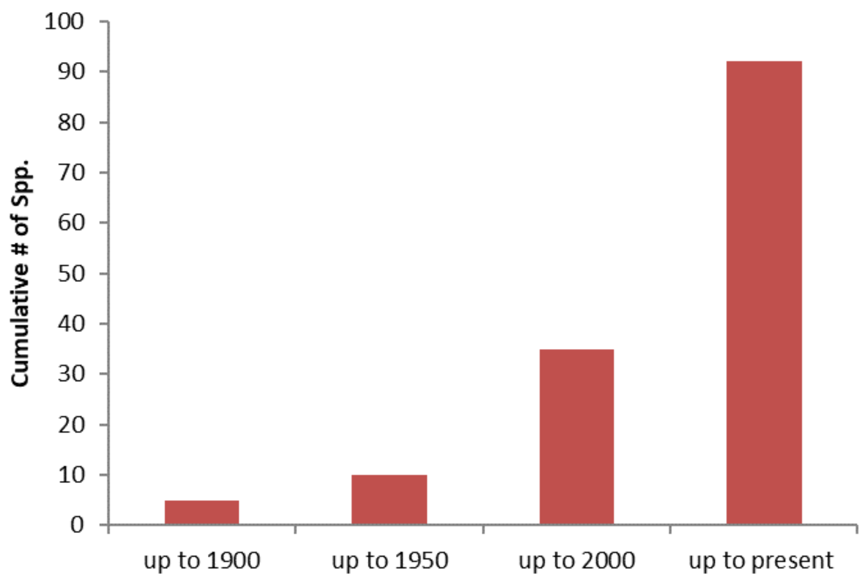


Figure 4.33 Cumulative number of non-native aquatic species introduced into the Lower St. Johns River Basin, Florida since the turn of the 20th century.

Non-native plants and animals arrive in the St. Johns River watershed by various means. Common vectors of transport have been humans, ship ballast consisting of water and/or sediment, ship/boat hull fouling, and mariculture/aquaculture activities. For example, JAXPORT imported >18,000 50-pound bushels of oysters (**JAXPORT 2017**), which have the potential to carry non-native organisms. One of the most widespread ways that non-native species arrive in Florida is when people accidentally or intentionally release exotic aquarium plants or pets into the wild. Exotic pet releases are especially problematic in Florida where the likelihood of the animal's survival and subsequent invasion into the natural habitats can be quite high (**Episcopio-Sturgeon and Pienaar 2019**). Such releases not only violate state and federal laws but can have devastating impacts on native ecosystems and native biodiversity. **Episcopio-Sturgeon and Pienaar 2019** investigated perceptions of Florida stakeholders in the pet trade towards managing invasion risks. Their study highlighted the challenges in monitoring the pet trade industry for invasive species and mistrust between stakeholders and regulators. In addition, there was concern that banning species may promote an illegal pet trade industry. Self-regulation by the pet trade industry was suggested as one solution, but also has associated risks (**Episcopio-Sturgeon and Pienaar 2019**).

4.5.7. Future Outlook

IRREVERSIBLE IMPACTS. Once a non-native species becomes naturalized in a new ecosystem, the environmental and economic costs of eradication are usually prohibitive (**Elton 1958**). Thus, once an invasive species gets here, it is here to stay, and the associated management costs will be passed on to future generations. Since the early 1900s, taxpayer dollars have been paying for ongoing efforts to control the spread of invasive non-native aquatic species in the St. Johns River. Currently, the FWC has imposed a temporary executive order to limit importation of species considered to be listed as Prohibitive Species (full list provided by **FWC 2019**) and is working to develop rules for nonnative species import permit applications and potential additions to the Prohibited Species list (**Segelson 2018**).

HIGH RISK. There is a high probability that future invasions of non-native aquatic species will continue to occur in the LSJRB. Human population growth in northeast Florida is projected to more than double by 2060 (**Zwick and Carr 2006**). Significant vectors for transporting non-native organisms are imported products and ship ballast, and these vectors are expected to contribute to the likelihood for additional and potentially more frequent introductions.

The Port of Jacksonville reported 1.28 million containers moved through the port with declines in the number of calls due to the pandemic (Figure 4.26, **JAXPORT 2021**). On average, 71.8 ± 5.9 cruise vessel calls per year were recorded between 2010-2020 (**JAXPORT 2021**). Approximately $100 million in federal funds have been awarded to JAXPORT to deepen the first 11 miles of the St. Johns River to 47 feet, along with FDOT award of $35.3 million which results in $72.5 million in state funding (**JAXPORT 2019b**). Deepening of the first eight miles will conclude by May 2021 the remaining three miles to Blount Island is in the design phase (**JAXPORT 2019b**).

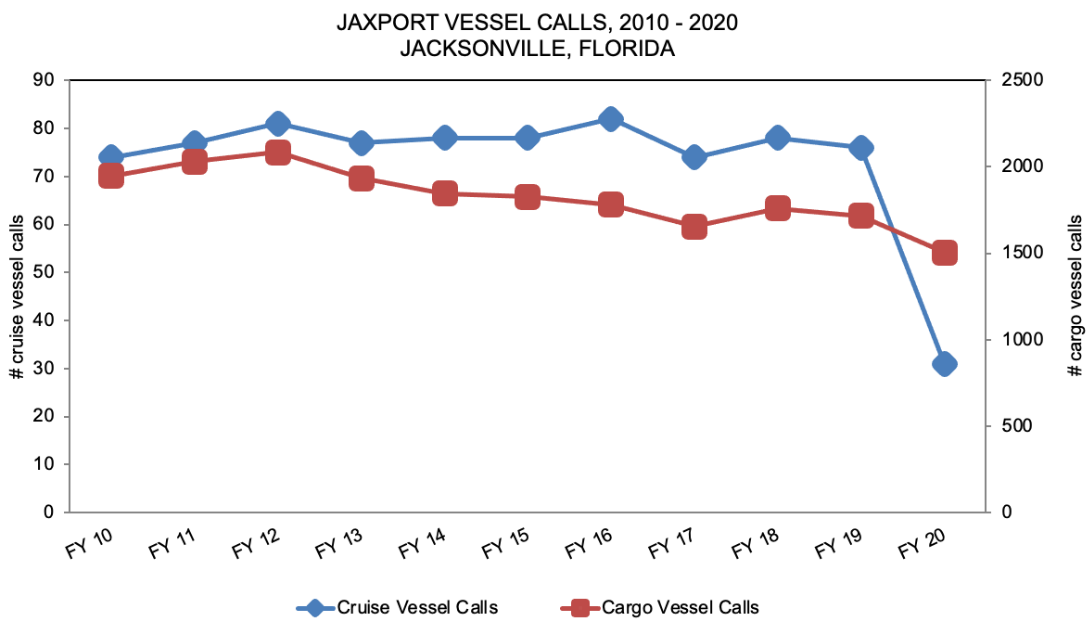


*Figure 4.34 Number of containers and cargo ships calling on Port of Jacksonville, FL (JAXPORT) terminals between fiscal year 2010 and 2020. Fiscal year (FY) begins Oct 1 (**JAXPORT 2019a**).*

Additional invasions into the Lower St. Johns River Basin are expected from adjacent or interconnected waterbodies. For example, 19 non-native aquatic species not found in the LSJRB have been recorded in the Upper St. Johns River Drainage Basin (**USGS 2015**). These species may disperse into the LSJRB. In addition, 85% of living non-native plants that are received into the US come from the Port of Miami (**ELI 2008**).

Rising global temperatures may also contribute to a northward expansion in the range of non-native species from Central and south Florida. For example, the old world climbing fern and Cuban treefrog were recorded in St. Johns and Duval counties in 2016, species spreading from southern Florida (**CISEH 2014**). There is concern that the Cuban treefrog can spread as tadpoles in fresh and brackish water with ~80% survival at 12 ppt and were able to survive 14 ppt for up to 24 hours (**Johnson and McGarrity 2013**). The habitat for the most northern record of Cuban treefrog tadpoles was described as ponds created after Hurricane Matthew (**CISEH 2016**). Hurricane Irma also helped spread the species (**Stepzinkski 2019**). The frog is known to eat native species green and squirrel treefrogs, lizards, and snakes (**Stepzinkski 2019**). **Gilg et al. 2014** studied dispersal of the green mussel near the Matanzas, St. Augustine, and Ponce de Leon Inlet. Mussel spat density was positively correlated with temperature and likely to be correlated with phytoplankton availability. Larvae settled within 10 km of source population located in the Intracoastal Waterway. The authors suggest that populations at the mouth of the SJR may be connected to the more southern populations due to transport along the coast, but that persistence is due to localized recruitment (**Gilg et al. 2010**).

Non-native fish species are often caught by local recreational fishers and researchers. A predatory redtail catfish was caught in Clay County from a local pond (**News4JAX 2015**). The aquarium fish was likely released and can reach 80 kg in weight (**News4JAX 2015; USGS 2015**). A foot-long Asian tiger shrimp was netted in July 2015 (**FCN 2015**). In addition, significant numbers of tilapia and sailfin catfish were collected within 10 km of the mouth of Rice Creek (**Gross and Burgess 2015**). Other species raising concern is the Muscovy duck that can transmit disease to and can interbreed with Florida's native waterfowl (**FWC 2014c**). In addition, the Black and white tegu has been observed in Avondale and have the potential to enter gopher tortoise holes for mice and tortoise eggs (**JHS 2014; CISEH 2015**) and the Peters's rock agama (**Brasileiro 2020**).

Given the devastating impacts of lionfish on coastal communities, Florida Fish and Wildlife Conservation Commission have waived the recreational license requirement if using designated spearing devices and have also waived bag limits harvesting lionfish (**FWC 2014a**). To date, lionfish have only been recorded off shore of northeast Florida and not in the SJR; however, they are known to eat commercially and recreationally important species of black sea bass and vermilion snapper (**Stepzinkski 2019**). **Johnson and Swenarton 2016** developed a length-based age-structured model for lionfish from >2,000 individuals caught by spear fishermen off the coast of northeastern Florida in 2013-2015. The authors reported that larger lionfish are culled or are moving to deeper waters. Recruitment events are occur during early summer, and growth rates are much greater than recorded from their native ranges (**Johnson and Swenarton 2016**). In the 2020 Lionfish Challenge sponsored by FWC, 21,569 lionfish were removed from Florida waters (**News4JAX 2020**).

Another point of concern is the lack of knowledge regarding invasive species in Florida. A recent survey by UF/IFAS Center for Public Issues Education in Agricultural and Natural Resources (PIE Center) indicated 62% of 515 Florida residents to be slightly or not knowledgeable of invasive species in general, 63% were slightly or not knowledgeable of the types of invasive species in Florida, and 66% were slightly or not knowledgeable of how to prevent invasions from entering Florida (**Dodds et al. 2014**). Yet, 79% of respondents were likely to pay attention to a story covering invasive species, with >70% preferring to learn about invasive species from the television, websites, videos, fact sheets, and newspapers. To test the effectiveness of educational programs on invasive species and plant biosecurity, **Pinkerton et al. 2019** provided in-class presentation and hand-on activity to Florida high schools students. Survey responses indicated that students improved their understanding and that plant biosecurity generated quite a bit of discussion (**Pinkerton et al. 2019**). These surveys highlights the importance of educational outreach and the interest of the public in learning about invasive species (**Dodds et al. 2014; Pinkerton et al. 2019**).

In addition to the general public, stakeholders in the pet trade industry may also have a lack of understanding and awareness of the potential of invasion and societal and environmental costs of exotic pets (**Episcopio-Sturgeon and Pienaar 2019**). Furthermore, mistrust between members in the Florida pet trade industry with the government and media can make self-regulation and management of invasion risk difficult. Pet trade stakeholders suggest that 'point-of-sale information and improved outreach' should be explored to better educate pet owners of the invasion risks of exotic pets (**Episcopio-Sturgeon and Pienaar 2019**).