Interactions between river stage and wetland vegetation detected by LANDSAT images in the Apalachicola delta, Florida

Daniele la Cecilia^{1,2}, Marco Toffolon², Curtis E. Woodcock¹, Sergio Fagherazzi¹

¹ Department of Earth and Environment, Boston University, Boston MA

² Department of Civil, Environmental and Mechanical Engineering, University of Trento,
Trento, Italy

Abstract

The distribution of swamp floodplain vegetation and its evolution in the lower non-tidal reaches of the Apalachicola River, Florida USA, is mapped using Landsat Thematic Mapper and Enhanced Thematic Mapper Plus (TM/ETM+) images captured over a period of 29 years. A newly developed seasonality index (SI), the ratio of the NDVI in winter months to the summer months, shows that the hardwood swamp, dominated by bald cypress and water tupelo, is slowly replaced by bottomland hardwood forest. This forest shift is driven by lower water levels in the Apalachicola River in the last 30 years, and predominantly occurs in the transitional area between low floodplains and high river banks. A negative correlation between maximum summer NDVI and water levels in winter suggests the growth of more vigorous vegetation in the vicinity of sloughs during years with low river flow. A negative correlation with SI further indicates that these vegetation patches are possibly replaced by species typical of drier floodplain conditions.

1 Introduction

Floodplain swamps offer multiple beneficial ecological services. Not only they mitigate flood risk and improve water quality, but also provide habitat for endangered species (Keddy, 2010). In recent times, these delicate environments are threatened by anthropogenic stresses and climate change. Changes in river hydrology can modify the flooding regime of the floodplain endangering its ecosystems.

Hydrology is identified as the major driving variable defining and influencing the ecosystem structure and functions in floodplains (Mertes et al., 1995; Amoros and Bornette 2002). Floodplain ecosystems are the complex interrelated systems of floral and faunal communities that welcome water during high discharge events (Bayley, 1995; Wharton et al., 1982). During riverine low stages, floodplains are usually fed by groundwater that maintains a saturated soil. When the water level becomes too low, as for instance during droughts, the hydrological connection between river and floodplain is lost. In these periods, the ground is exposed more frequently, soil moisture tends to evaporate more easily, thus affecting floodplain vegetation. Reduced water availability often leads to a change in the composition of floodplain vegetation, with dramatic consequences for the entire ecosystem (Rood et al., 2008; Horner et al., 2009).

The complexity of floodplain ecological interactions can be explained by means of the flood pulse concept (Junk et al., 1989; Tockner et al., 2000). The pulsing of river discharge during floods stimulates the exchange of energy, matter, and organisms across the river floodplain. This cycling phenomenon of long duration appears to be a dominant factor for the development of biorhythms characteristics of floodplain ecosystems (Tockner et al., 2000). Floodplains store large quantities of organic matter that is slowly decomposed causing a concentration of nutrients. The flood pulse mobilizes and distributes these nutrients through the whole floodplain creating at the same time vital ecosystemic connections. Moreover, part

of the nutrients and organic matter is washed downstream and supports estuarine and marine ecosystems (Apalachicola National Estuarine Research Reserve, 2013).

A loss of tree density in floodplain wetlands would lead to a decrease in tree and leaf litter biomass, which would have additional effects on swamp organisms. Both the loss of litter and the cut off of lateral connections between aquatic and terrestrial environments would result in a loss of substrate (Darst and Light, 2008; Keruzoré et al., 2013), with possible consequences for benthic organisms in the floodplain and, ultimately, in the downstream waters of the river and estuary. Moreover, floods have the ability to affect germination, seedlings survival, regeneration success, and plant succession (Jones et al., 1994). Inundations alter the spatial distribution of habitats, and this is essential to maintain the richness and diversity of species in floodplains (Stallins et al. 2010; Wen et al., 2012). Due to the high level of primary productivity and habitat heterogeneity, floodplains are often considered regional biodiversity hotspots (Tockner and Stanford, 2002; Kooistra et al., 2008). In large subtropical lowland rivers with a constant temperature during the growing season, the connections between the river and its floodplain become more apparent after major inundations.

Here we investigate vegetation dynamics in the floodplain of the Apalachicola River, Florida USA, by exploring the relationship between satellite-derived vegetation indices and river stage. The lower non-tidal reaches of the Apalachicola River underwent several human interventions to maintain a now rarely used inland water transport system, such as dredging and disposal of sediments on the floodplain and straightening of the main river course (Leitman et al., 1983). The hydrological alterations in the Apalachicola River are also directly related to the construction of dams, which regulate the flow preventing frequent floods. Although large floods can still occur because the reservoirs volume is limited, the sediments are trapped by the dams and the river bed is subject to excavation downstream (Darst and Light, 2008). This process makes the existing banks higher relative to the bottom, thus

reducing the hydrological connection between the river and the floodplain. In addition, water withdrawal for agricultural and municipal purposes has increased in the last five decades. All these factors have contributed to declining water levels in the last 30 years, leading to shorter durations for floodplain inundations, a disconnection of the floodplain with the main channel, and eventually to a change in forest characteristics (Light et al., 2006; Stallins et al., 2010; Smith et al., 2013).

In our study, we used the vegetation indicator NDVI (Normalized Difference Vegetation Index) derived from a time series of data from Landsat 5 Thematic Mapper and Landsat 7 Enhanced Thematic Mapper Plus. Field data are typically expensive and are usually collected at small spatial and temporal scales, whereas remote sensing data are ideal for monitoring large areas. Therefore, satellite imagery has become a powerful tool for ecologists and geomorphologists in riverine studies (Kerr and Ostrovsky, 2003; Turner et al., 2003). A common drawback of satellite images is the reliability of the observations. However, new algorithms are available to reduce noise in NDVI time-series, for instance by detecting and disregarding pixels covered by clouds, snow, ice or standing water (Zhu and Woodcock, 2012).

This study has two goals: i) to define a simple index, based on easily available remote sensing information, that relates vegetation dynamics to the hydrological regime; ii) to use the index to interpret the medium-term changes that are occurring in the Apalachicola river floodplain. The methodology used here allows monitoring of the dynamics of swamp vegetation in alluvial river floodplains, so as to better understand this complex ecosystem and to provide critical information for flow regulation.

2 Study site

The Apalachicola River is part of a three-river system (the Apalachicola-Chattahoochee-Flint or ACF basin) that originates in the Appalachian Mountains and drains 50,800 km² within Alabama, Georgia, and Florida (Fig. 1A).

Figure 1: A) Apalachicola-Chattahoochee-Flint watershed in the southeast part of the USA; B) area analysed in this study in the Apalachicola floodplain (delimited by a yellow box). The overall area is 29.44 km² wide.

The Chattahoochee River flows for about 700 km from its source in North Georgia to the Florida border, and serves as the boundary between Georgia and Alabama for part of this distance. It meets sixteen dams, with the last one at Lake Seminole where it joins the Flint River. The Flint River originates south of Atlanta, and flows about 600 km through southwestern Georgia. The Apalachicola River begins at the Flint-Chattahoochee confluence and flows unimpeded through Florida for about 170 km to Apalachicola Bay. The Apalachicola River is 21st in magnitude in terms of flow in the conterminous United States with a mean annual flow of 630 m³/s (average value in the period 1922-1995).

Figure 2: A) Elevation map of the study site (yellow box in Fig. 1B), reconstructed from LiDAR data taken in 2010 (North West Florida Water Management District, 2010) resampled at 30m resolution. The light blue areas near the main river are natural levees with higher elevations and drier conditions. B) Map of vegetation cover in 2003 (Florida Fish and Wildlife Conservation Commission 2003). Three samples of the dominant forest classes are indicated with a box: a) pineland forest; b) bottomland hardwood forest; c) hardwood swamp.

2.1 Dominant vegetation species

Our study site was chosen within the non-tidal reaches of the river, 32 km upstream from the river mouth in Apalachicola Bay (Fig. 1B). According to the Florida Fish and Wildlife Conservation Commission (2003), five vegetation classes are present in the Apalachicola floodplain: cypress swamp, hardwood swamp, bottomland hardwood forest, mixed wetland forest, and pineland forest (FWC 2003, Fig. 2B). The vegetation cover map in Figure 2B was

derived by the FWC from a Landsat ETM+ scene, previous land cover datasets created by different agencies, and a digital orthographic quarter quadrangle aerial photography. Specific areas of the map were visited in the field for ground-truthing (see Stys et al. 2004 for details).

Cypress swamps are found along the investigated segment of the Apalachicola River and in depressions which occur within the floodplain. These forested zones are habitually inundated, with either bald cypress (*Taxodium distichum*) or pond cypress (*Taxodium ascendens*) dominating these wetlands. Furthermore, both understory and vegetated ground cover are rarely present because of the prolonged time of standing water.

The most common tree species of the hardwood swamps are water tupelo (*Nyssa aquatica*) and bald cypress (*Taxodium distichum*) (Edmiston and Tuck, 1987). Both species lose their leaves in winter, and grow in low, saturated, and seasonally inundated soils and in deep swamps. In fact, both the long duration of inundations and the high water levels during floods control forest composition through a process of exclusion, drowning the seedlings of most bottomland hardwood species before they can become established (Hosner, 1960; Light et al., 2006). Bald cypress usually achieves dominance when competing with water tupelo.

However, the adverse condition of moving water in sloughs confines bald cypress trees to the margins and more shallow areas. Water tupelo requires more stringent wet conditions to be competitive at the seedling stage, is less shade-tolerant and favors mineral soils with high silt content (Hook, 1984). As the flood depth in the swamp becomes deeper and the water stagnant, the two species occupy much of the same area (Johnson, 1965). Other hardwood species are also present, the most common are Carolina ash (*Fraxinus caroliniana*), and planer tree (*Planera aquatica*), as well as other tupelo species: swamp tupelo (*Nyssa biflora*), Ogeechee tupelo (*Nyssa ogeche*), and blackgum (*Nyssa sylvatica*).

Reduced inundation periods and higher ground elevation characterize the levees of the Apalachicola River, where sediments tend to settle during overbank flows creating alluvial

deposits (Fig. 2A). As a result, drier conditions and a diverse mosaic of habitats are produced, thus allowing many tree species to survive. Here bottomland hardwood forests thrive (Darst and Light, 2008). In our area of interest this type of forest has a predominance of deciduous trees, such as water hickory (*Carya aquatica*), overcup oak (*Quercus lirata*), sweetgum (*Liquidambar styraciflua*), green ash (*Fraxinus pennsylvanica*), and sugarberry (*Celtis laevigata*) (Leitman et al., 1983).

The category of the mixed wetland forest includes a mix of hardwoods such as bald cypress (*Taxodium distichum*) and pines, primarily loblolly pine (*Pinus taeda*), in which both communities are equally present. This habitat is regarded as a transition environment from hydric to mesic sites. Finally, the pineland forests present in the research area are dominated by loblolly pine (*Pinus taeda*) and longleaf pines (*Pinus palustris*).

2.2 LiDAR and Landsat data

The main goal of our analysis is to determine the relationships between hydraulic conditions and vegetation cover in the Apalachicola floodplain over time. To this end, we rely on 29-year series of satellite data (Landsat Thematic Mapper TM and Enhanced Thematic Mapper Plus ETM+) to reconstruct the vegetation distribution in our study site from 1983 to 2011. Satellite data are combined with a Digital Elevation Model based on LiDAR data collected in 2010 and water level data recorded at a nearby gauging station. LiDAR data for the area of interest are available from the North West Florida Water Management District Public LiDAR Data Server (North West Florida Water Management District 2010). The Digital Elevation Model was derived from bare earth LiDAR points collected in spring 2010 by the Florida Department of Emergency Management. We applied a coordinate transformation to the LiDAR data (NAD83 UTM 16N) to match the datum of the Landsat data (WGS84 UTM 16N). The LiDAR data were then resampled to the 30 m pixel size of the Landsat images

using the standard bicubic interpolation implemented in GRASS GIS 6.4, which uses the 16 nearest cells (Fig. 2A).

Figure 3: Frequency and timing of Landsat TM/ETM+ images used in the study, where each square represents a single Landsat image. The darker the square the higher the number of cloud-free pixels within the research area (see the scale on the right). Note that the density of observations increases after the launch of Landsat 7, but it decreases again after the scan line corrector failure of Landsat 7 that occurred on 31 May 2003.

We processed 462 Landsat Thematic 5 TM (TM5) and Landsat 7 Climate Data Record (CDR) TM/ETM+L1T images from 1983 to 2011 for a Landsat scene centered over Northwest Florida (Path 019 Row 039) (Fig. 3). We did not consider Landsat 7 ETM+ recorded after the 31th of May 2003 because of the scan line corrector failure. The CDR images were screened for clouds and cloud shadows using an implementation of the F-mask algorithm of Zhu and Woodcock (2012). To characterize the vegetation cover we use the Normalized Difference Vegetation Index (NDVI), defined as:

$$NDVI = \frac{(NIR - RED)}{(NIR + RED)} \quad (1)$$

where NIR and RED are surface reflectance of the near infrared and the red band (Landsat Band 4) and the red band (Landsat Band 3). The definition is based on the physical principle that chlorophyll absorbs RED whereas the mesophyll leaf structure scatters NIR (Kerr and Ostrovsky, 2003). NDVI values range between -1 and 1; vegetation has NDVI values

generally above 0.3, while both wet and non-vegetated surfaces have NDVI values either negative or close to 0.

The relationship between NDVI and vegetation productivity is well established, and the link between this index and the fraction of absorbed photosynthetic active radiation has been well documented both theoretically (Sellers et al., 1992) and empirically (Asrar et al., 1984). NDVI is being used worldwide to monitor floodplain dynamics, to assess both abundance and richness of target species (Gould, 2000), to evaluate the response of floodplain vegetation to either land use or climate change, for remote monitoring of forest response to changed soil moisture regimes due to river regulation (Tewari et al., 2003), to assess damage after severe storms and hurricanes (Faulkner et al., 2005), and to determine the relationship between flood frequency and plant growth vigor in large floodplains (Sims and Thoms, 2002).

2.3 Seasonality Index based on Landsat data

In our study we identified three areas 150 x 150 m² in size uniformly covered by pineland forest, bottomland hardwood forest, and hardwood swamp (squares marked a,b,c in Fig. 2B). To determine seasonal variations in NDVI for different vegetation covers, for each pixel of the three areas we made a plot of the NDVI typical year (NTY) by plotting all existing NDVI values over the 29 years according to the day of the year (DOY) it was collected. When multiple values were available for the same DOY, the maximum NDVI value was chosen, since clouds and shadows can be present in some images.

We also calculated the maximum NDVI value in the spring-summer months (defined as the months from May to September) and in the winter months (January to February). Based on these data, we introduce a simple Seasonality Index (SI) as the ratio of the maximum NDVI calculated in winter months (NDVI_w) to the maximum NDVI calculated in spring-summer months (NDVI_s):

216

$$SI = \frac{NDVI_w}{NDVI_s} \quad (2)$$

217

218 **Figure 4:** NDVI Typical year for: A) pineland forest, B) bottomland hardwood forest, and C) hardwood
 219 swamp. We indicate the periods for which we calculate the winter NDVI and the summer NDVI in light grey
 220 and in light green respectively. Each point is the average NDVI value of the areas indicated in Figure 2b for one
 221 Landsat image. The points are ordered by the calendar day when the image was collected, without accounting
 222 for the year. The Seasonality Index, SI, is obtained as the ratio between the maximum NDVI in summer (light
 223 green interval) and the maximum NDVI in winter (light grey interval).

224

225 Different vegetation classes are characterized by different distributions of NDVI values in
 226 the typical year (NTY), as depicted in Fig. 4. Pine trees have a SI close to 1 because NDVI is
 227 approximately constant during the year for evergreen species, while bottomland hardwood
 228 trees and hardwood swamp trees show a lower SI value, with the last type of vegetation being
 229 characterized by the almost complete absence of leaves during winter. By considering the
 230 ratio between $NDVI_w$ and $NDVI_s$, the SI is able to capture some important factors that

characterize the different vegetation classes, and a significant change in SI may imply a shift in the type of vegetation cover. The SI is defined every year for each pixel, thus it can be used to represent both spatial and temporal variations of vegetation canopy. We exploited these features and used the SI as a tool to provide a classification of the vegetation in the study area.

2.4 Water level data

Water level data are available from the automated U.S. Geological Survey station 02359170 of the Apalachicola River near Sumatra. The present analysis is based on the series of daily values of water level for this station. We calculated the maximum, mean, and minimum water level values for each month of the 29-year period corresponding to the available NDVI dataset. The typical range of monthly averaged variations of water level is shown in Fig. 5A: peak water levels occur in February, March and April (FMA), when the river stage is on average higher. According to the flood pulse theory, severe cyclic floods are essential to keep the floodplain's ecosystems vital and connected (Junk et al., 1989). Therefore, we consider this period critical for the development of the vegetation in the floodplain. Moreover, referring to the same period, we calculate the number, n_w , of days for which the water level was above a given value in each year. The range of water levels used in this analysis varied from 1m to 3m above mean sea level, with a vertical interval of 0.2m. This range encompasses both the averaged minimum and maximum water levels (Fig. 5B). It is important to note that hydrological data are less reliable for the low-laying swamps due to individual site characteristics, which can be highly variable (see Darst and Light, 2008).

Figure 5: A) Monthly values of minimum, average, and maximum water level in the Apalachicola River near Sumatra, Florida, averaged from 1983 to 2011; B) Duration of minimum, average, and maximum water levels in February, March, and April.

2.5 Correlation between NDVI, SI and water levels

To explore relationships between vegetation cover and water regime in the area of interest we computed the Pearson's linear correlation coefficient between NDVI, SI, and water levels. According to previous findings (Darst and Light, 2008; Stallins et al., 2010; Smiths et al., 2013), a reduction in water level during low flows, which occurs almost 80% of the time in the Apalachicola River, is deleterious to the ecosystem. In fact understory composition and structure may change in swamps as drier conditions facilitate the establishment of bottomland hardwood vegetation as well as generalist understory ground cover (Stallins et al. 2010).

In this analysis, we considered the 29-year long datasets of winter minimum water level values, summer maximum NDVI values, and SI values for all the pixels in the study area. Similarly, we calculated the Pearson's linear correlation between the number of days in winter, n_w , with water levels above a given value and both NDVI and SI.

3 Results

3.1 Forest classification using the SI

We compared the distribution of vegetation cover provided by FWC in 2003 to the SI map in the same year, and defined SI ranges for each class to match the FWC vegetation cover map (Table 1). The SI thresholds were varied to determine what values yield the highest overlap in terms of area between the FWC map and our classification. For each set of SI thresholds, a confusion matrix was obtained (see Table 2 as an example) and the overall accuracy computed as the percentage of area that have been correctly labelled (based on SI) compared to the classes adopted as a reference (the sum of pixels on the trace of the confusion matrix divided by the total number of pixels). The trace of the confusion matrix defines the overall accuracy: values close to 100% suggest that the method is able to classify correctly the whole class as defined in the FWC map. The misclassified pixels can be examined by evaluating each row. Considering the third row of Table 2 as an example, 79% of the pixels belonging to the bottomland hardwood forest (BHF) are correctly classified, while 16% of the BHF in the FWC map is classified as HS using our method, 4% as PF, and 1% as water and unclassified areas. The set of chosen SI thresholds (Table 1) is the one leading to the highest percent of pixels correctly classified. The corresponding confusion matrix is reported in Table 2 and the agreement with the FWC map is equal to 81%.

Note that the 2003 SI map (Fig. 6A) has been categorized with only 4 classes (Fig. 6B) based on the SI thresholds identified in Table 1. The SI effectively distinguishes areas

covered with pine trees ($SI \approx 1$) from bottomland hardwood trees ($SI \approx 0.5$) and hardwood swamp trees ($SI \approx 0.3$). Negative SI values indicate the presence of standing water in the winter season (Table 1). The SI was not able to correctly identify cypress forest and mixed wetland forest, so we reclassified the FWC map in the three categories (pine forest, bottomland hardwood forest, and hardwood swamp) instead of five. Our choice also took into account that the areas classified as mixed wetland forests did not have a dominant species and the soil condition could have been either hydric or mesic (Gilbert and Stys, 2004). Moreover, the research area encompasses floodplain swamps and pine uplands, that are essential valuable components of an intertwined ecological system that also supports economic and recreational activities downstream in the bay (Apalachicola National Estuarine Research Reserve, 2013). Therefore, the mixed wetland forest was grouped with the bottomland hardwood forest, rather than with the pineland forest, due to the higher number of shared tree species. In addition, the limited areas covered by cypress forest were discarded from the analysis. Therefore, we rely on SI to track the dynamics of the three major vegetation categories, and to study changes in vegetation cover and their relationship with water levels.

Table 1. Classes used in the analysis and SI thresholds for forest classification. For each class, the dominant tree species are listed.

	Type	Tree species	Thresholds
Class 1	Pineland forest (PF)	<i>Pinus palustris</i> <i>Pinus taeda</i>	$SI \geq 0.8$
Class 2	Bottomland hardwood forest (BHF)	<i>Carya aquatica</i> <i>Quercus lirata</i> <i>Liquidambar styraciflua</i> <i>Fraxinus pennsylvanica</i> <i>Celtis laevigata</i>	$0.5 \leq SI < 0.8$
Class 3	Hardwood swamp (HS)	<i>Nyssa aquatica</i> <i>Taxodium distichum</i> , <i>Fraxinus caroliniana</i> <i>Planera aquatica</i> <i>Nyssa biflora</i> <i>Nyssa ogeche</i> <i>Nyssa sylvatica</i>	$0 < SI < 0.5$
Class 4	Water and unclassified areas (W-N)		$SI \leq 0$

Table 2. Confusion matrix between the classes of the FWC map and the classes identified with the Seasonal Index introduced herein (classes refer to vegetation types as in Table 1). Each row contains the types of vegetation as reported in the FWC map, whereas each column returns the vegetation class derived from the SI. The units are in number of pixels. The overall accuracy is equal to 81%.

Class	Water and unclassified areas	Hardwood swamp	Bottomland hardwood forest	Pineland forest	Total
River	917 27%	829 30%	1118 21%	759 21%	3623 4%
Hardwood swamp	263 1%	20188 90%	1948 1%	67 9%	22466 67%
Bottomland hardwood forest	35 1%	815 16%	3983 79%	235 4%	5068 22%
Pineland forest	1 0%	6 0%	282 18%	1320 82%	1609 7%
Total	1216 27%	21838 69%	7331 15%	2381 5%	32766 Accuracy: 81%

Figure 6: A) Spatial distribution of Seasonal Index (SI) in 2003. B) Categorized SI map for the same year, to be compared with the FWC map of Fig. 2B. The range of the SI categories is reported in Table 1.

Our classification was based on the SI data of one year (2003, the year when the FWC map was made) and can therefore be sensitive to the specific hydrological conditions of that year. Wetter or drier conditions can in fact affect the NDVI values in summer, and therefore the entire classification. To further investigate the validity of our method, the classification based on the SI index for the year 2011 was validated against a high resolution aerial image of the Apalachicola forest collected on April 10 2011 with a spatial resolution of 0.61m (DigitalGlobe, courtesy Google Earth). From the categorized SI map of 2011 we created a stratified sample for each vegetation class (25 pixels for the water and unclassified areas, 100 for the hardwood swamp, 50 for the bottomland hardwood swamp, 25 for the pineland forest), following the methodology reported in Olofsson et al. (2013, 2014). We then conducted a visual assessment of the vegetation present in each 30 x 30 m² pixel after projecting the sample pixels on the aerial photograph.

We report in Table 3 the four-class error matrix in which we allocated our estimates of accuracy. Each row contains the types of vegetation as visually identified in the reference aerial photograph, whereas each column returns the vegetation class derived from the SI. The classification yields an overall accuracy of 87%, with a user's accuracy between 66% and 94% and a producer's accuracy between 74% and 89%.

Table 3. Error matrix with cell entries containing the estimated proportion of area, with a 95% confidence interval. Reference classes are the rows (aerial photograph) while the map classes are the columns (vegetation classes derived from the Seasonal Index).

Class	River	Hardwood swamp	Bottomland hardwood forest	Pineland forest	Number of pixels	W_i	User's	Producer's	Overall
River	0.031	0.010	0.002	0	25	0.043	0.72 ± 0.18	0.74 ± 0.08	0.87 ± 0.04
Hardwood swamp	0.006	0.629	0.033	0	100	0.669	0.94 ± 0.05	0.89 ± 0.04	
Bottomland hardwood forest	0.004	0.060	0.141	0.008	50	0.213	0.66 ± 0.13	0.77 ± 0.20	
Pineland forest	0	0.003	0.006	0.066	25	0.075	0.88 ± 0.13	0.88 ± 0.01	
Total	0.042	0.702	0.182	0.0742	200	1			

3.2 The use of SI for mapping spatial and temporal changes

In Fig. 7A we show temporal variations in the fractional area of each vegetation class during the study period, reconstructed using the SI. The data are smoothed with a moving average of three years. As water levels decrease the hardwood swamp cover shrinks ($R^2=0.74$, $p<0.005$), while the bottomland hardwood forest widens. The area covered with pineland trees is increasing as well, likely as a result of both the natural succession of a young forest and a diminished exposure to floods (Stallins et al., 2010), with the second factor being prevalent for the pineland forest at lower elevation. This result suggests the transition of the vegetation in the floodplain to a drier composition, in agreement with Darst and Light (2008) studies on changes in forest composition and trees abundance.

Figure 7: A) Fraction of each vegetation class in the study area from 1987 to 2008. Each data point is the average of three years. The standard deviation is also indicated. B) Spatial distribution of average SI and D) categorized vegetation map for the period 1986-1988. C) Spatial distribution of average SI and E) categorized vegetation map for the period 2009-2011. The classification reported in C and E was validated with a high resolution aerial photograph in 2011 (see table 3).

The remote sensing images detect the expansion of bottomland hardwood forest, colored in blue, from the elevated banks of the river into the nearby swamp (Fig. 7D, E). In the southwest corner of the study site we detect an oxbow lake whose banks are colonized in time by bottomland hardwood forest (Fig. 7E), while the natural pineland forest vegetation (in yellow) in the northeast corner is spreading. If the floodplain vegetation classification by means of SI is reliable, then we can quantitatively assess how the vegetation has changed in

the last 20 years (Table 4). The area loss is calculated simply by multiplying the number of pixels that changed category by the area of each pixel.

Table 4. Area loss for each vegetation category between 1986-1988 and 2009-2011 (classes refer to vegetation types as in Table 1).

Vegetation change	Area loss (km ²)	Area loss (%)
Hardwood swamp (HS) to bottomland hardwood forest (BHF)	4.145	14.05
Bottomland hardwood forest (BHF) to hardwood swamp (HS)	0.036	0.12
Bottomland hardwood forest (BHF) to pineland forest (PF)	0.836	2.83
Pineland forest (PF) to bottomland hardwood forest (BHF)	0.274	0.93
Pineland forest (PF) to hardwood swamp (HS)	0	0
Hardwood swamp (HS) to pineland forest (PF)	0	0

We also analyzed the influence of elevation on the spatial pattern of vegetation types. In Fig. 8A we show a plot of SI, measured in 2010, for all pixels in our study area, as a function of ground elevation derived from the LiDAR collected in 2010. The plot suggests that elevation is an important parameter controlling vegetation distribution: as the elevation increases SI increases too ($R^2=0.64$, $p<0.05$). In floodplains, anaerobic soil conditions are among the most important limiting factors in the distribution of forest types (Edmiston and Tuck, 1987) and are related to the frequency of inundation that in turns depends on

topography (Stallins et al., 2010). The three vegetation covers are also segregated in terms of elevation (Fig. 8B).

Figure 8: A) Seasonality Index (SI) in 2010 as a function of elevation. Each point corresponds to a pixel in the research area. B) Box and whisker plot of elevation data for each vegetation category indicated in Table 1.

3.3 The effect of water levels

By correlating the minimum water level in winter months (February to April, FMA), $H_{min,w}$, with $NDVI_s$ we aim at understanding whether reduced winter flows are detrimental for the overall density of the vegetation (Weier and Herring, 2000). In Fig. 9A we plot the correlation coefficient between $H_{min,w}$ and $NDVI_s$ for each pixel in the study area. We disregarded pixels that are not statistically significant ($p > 0.05$, indicated by white areas in Fig. 9). For each pixel, the number of data points used in the correlations ranges from 18 to

24 (some pixels had either persistent cloud cover or stagnant water that reduced the number of Landsat images available in one year).

Figure 9: Map of the statistically significant correlation coefficients r between the maximum NDVI values in spring-summer months (May to September) and: (A) the time series of minimum water level in winter months (February to April); number n_w of days with water level above (B) 1.4 m, (C) 1.6 m, and (D) 1.8 m. Only significant values with $p < 0.05$ are plotted.

In Fig. 9A we notice few areas with significant negative correlation in the north part and in the center of the floodplain. A negative value suggests that a decrease in $H_{min,w}$ results in an increase in NDVI_s. In Fig. 9B-D we plot the correlation coefficient between NDVI_s and the number n_w of days with water level above a specific threshold. This analysis aims to assess whether the frequency of floods above a particular water level can trigger a vegetation change. We hypothesize that when the water level reaches a specific threshold in the main channel, the lateral connections with the floodplain become active. Above this water level riverine water starts flowing through sloughs and is able to alter the moisture content at the base of floodplain trees (Smith et al., 2013). We find that, for a range of water levels spanning from 1.4 m to 1.6 m a.s.l., a high negative correlation is present and widespread. As we reach the water level of 1.4m, even locations in the inner portion of the floodplain are significantly correlated (Fig. 9B). Thus, for the locations indicated in Fig. 9B and 9C a reduction of days with averaged minimum river level higher than 1.4 m during the wet season translates in an increase in NDVI in summer.

417

418

419

420

421

Figure 10: Map of statistically significant correlation coefficients r between SI and: (A) the time series of minimum water level in winter months (February to April); number n_w of days in winter with water level above (B) 1.4 m, (C) 1.6 m, (D) 1.8 m.

422

423

424

425

Moreover, Figure 10A shows that a significant negative correlation exists between minimum water levels in winter (FMA) and SI. If SI accurately classifies the three vegetation classes, our analysis would suggest that a decrease in water levels in the main channel leads to the transition to types of vegetation typical of drier floodplains. Areas with high correlation

coefficients, in absolute value, are located near the bottomland hardwood forest. Here a decrease in minimum water level triggers a significant increase in SI, possibly showing a fast conversion to a drier forest composition. We also analyzed the relationship between SI and n_w . We stress that only in case when the SI is able to reliably discern between the three types of forest, our last analysis could determine whether the vegetation composition is sensitive to the frequency of particular water levels. Figures 10B and 10C show that most of the research area manifests a high and negative correlation with respect to the water level threshold of 1.4 m. Therefore, water level values below this particular threshold may trigger a change in forest composition.

Maxwell and Knapp (2012) shows that *Nyssa ogeche* growth is sensitive to the amount of rainfall from the previous summer rather than recent winter rainfall. To explore this lagged relationship we performed the same analyses of Figure 9 and 10 but considering the water levels of the previous summer. We find low correlation coefficients, with r never exceeding 0.7. We therefore conclude that winter water levels exert a higher control on the SI index with respect to summer water levels.

4 Discussion

The non-tidal reaches of the Apalachicola River have been experiencing a decrease in frequency and duration of floodplain inundation and longer periods of low discharge mainly due to human interventions throughout its watershed (Light et al., 2006). Therefore, the quantity and quality of floodplain habitats for floral and faunal communities is declining and changing in response to drier conditions and loss of floodplain connectivity with the main river. Based on our vegetation monitoring method, we have linked pineland forests with high elevations, bottomland hardwood forests with intermediate elevations and hardwood swamps with low elevations. Differences in vegetation cover are due to the flood-tolerance of each

group of species. In fact, as the elevation increases the soil moisture is likely to decrease since ponding water is present for shorter periods. Moreover, at higher elevations a greater number of surviving seedlings can be found rather than in flooded sites (Jones et al., 1994).

Our results are in agreement with the findings of Stallins et al. (2010), which indicate that vegetation species experiencing the largest loss in the recent past live at the lowest elevations of the floodplain, where there are frequently submerged swamps. Stallins et al. (2010) report that the decline is affecting all size classes of the genus *Nyssa*, but particularly *Nyssa aquatica*. Moreover, the drying of the floodplain has reduced the density of typical tree species, slowed their growth and altered the ecological succession. Our analysis based on the SI classifier confirms this negative trend. In fact, a large area of hardwood swamp (light brown in Fig. 7D and 7E) has likely disappeared in the three decades analyzed here. Swamp vegetation is slowly being replaced by less flood tolerant trees, which are encroaching from the bottomland hardwood forest dominating the river levees, where trees are gaining basal area (Smith et al., 2013). For example, Darst and Light (2008) indicate that water oak became the dominant species in many areas in 2004, while it was not dominant in 1976.

The variation in habitat quality may influence vegetation succession. Thus, as the water level declines in the Apalachicola River, its lateral connections with the floodplain may be lost and both macro and micro-topography will play an important role in determining flood exposure and moisture availability. Consequently, macro-hydrologic and macro-climate variables may no longer control the tree growth in the floodplain. On the contrary, the vegetation composition and structure will depend on local parameters (Stallins et al., 2010). Currently, the floodplain forest is lacking of indicator species, and tree species typical to the swamp habitat are gradually declining (Stallins et al., 2010). Perhaps, in subtropical regions, the steady decline in water flows will eventually lead to less compositional differences within a drier floodplain forest.

Darst and Light (2008) indicate that the density of trees in the Apalachicola swamps significantly decreased by 37 percent from 1976 to 2004, leading to encroachment by grasses and sedges. Replacement of canopy trees with marsh grasses and sedges likely increases the SI index, because of their limited seasonal variations in aboveground biomass at these latitudes (Brinson et al 1981).

Floodplain zones having both high elevation and high SI values are not affected by the change in hydrological regime because they are likely established by bottomland hardwood forest (light blue regions in Fig. 7D and 7E). At lower elevations, the current tendency for the forest composition is to turn to species that establish on drier soil, with bottomland hardwood forest colonizing areas adjacent to the river levees and banks. These places are more affected by the decline in water level and will probably be the first to shift in vegetation composition. As a result, the boundary that separates swamp tree species from bottomland hardwood ones has been expanding and moving away from the river levees, reducing the extent of the swamp. The trend of SI distribution for each vegetation class as defined in our analysis, shown in Fig.7A, highlights this transition.

As indicated by Stallins et al. (2010), the floodplain forest is becoming compositionally noisier (with less fidelity of tree species to habitats) and less consistently structured. This process can accelerate the increase in SI, because the absence of indicator species and the presence of several new vegetation species would average out any seasonal signal in NDVI. An important question that should be addressed in future research is how fast the forest responds to a change in hydrological regime. A delay between the onset of drier conditions and variations in forest composition is to be expected, and it would be interesting to see whether it can be captured by remote sensing images.

Our data show that the pineland forest, colored in yellow in Fig. 7D and 7E, is expanding in the northeast corner, which is part of the upland forest adjacent to the floodplain. Here tree

growth may not be related to the river dynamics given the high elevations of the soil, and vegetation composition may depend mainly on local conditions (Stallins et al., 2010).

The frequency of inundations is the strongest hydrologic predictor of growth for all the species situated within the research area (Smith et al., 2013). We find that fewer inundation events have resulted in the increase of maximum summer NDVI ($NDVI_s$) in some parts of the swamp. As reported by Sims and Thoms (2002), in floodplains less frequently inundated there is more vigorous vegetation due to oxygenated soil and less disturbances affecting plant productivity. The correlation between $NDVI_s$ and the number n_w of days of high flow reveals that the vegetation canopy is sensitive to events with water levels higher than 1.6m. This might indicate that above this threshold sloughs connect the river to its floodplain, so water can flood a larger part of the floodplain affecting its vegetation. A reduction of flow events having water elevations above 1.6m would explain the increase in summer $NDVI_s$ within these disconnected area. The most negative correlation coefficients are found in the vicinity of currently dried sloughs. Dense grasses and sedges or less shade-tolerant trees are possibly colonizing these patches of bare ground (Light and Darst, 2008), increasing the NDVI values. These species are sensitive to flooding, thus explaining the negative correlation coefficients of Figure 9. The highly negative correlation between SI and $H_{min,w}$ may imply that locations with an elevation between 1.4 m and 1.6 m are more threatened by the decline in water level, and can be prone to vegetation change.

It takes a long period for the floodplain to reach a state of equilibrium between the hydroperiod and its vegetation composition and structure. When this equilibrium is lost the vegetation composition changes to reach a new dynamic configuration. Here we have shown how satellite data can help to observe the systematic responses of a floodplain to alterations to riverine flows in subtropical climates. On the other hand, there are some shortcomings in using satellite data. First, the number of usable images is relatively small because of the low

frequency of Landsat 7 ETM+ data (16-day revisit period) in conjunction with frequent cloud cover in subtropical regions. It is possible to increase the number of images using MODIS satellite imagery (1-day revisit period) instead of LANDSAT, however MODIS has a lower spatial resolution (250m rather than 30m pixels). Another problem is related to the fact that it is not always possible to detect and eliminate uncertainties in remotely sensed measures, like cloud shadows and the presence of standing water that tend to reduce NDVI values. For example, NDVI has been used to define the extent of floods in wetlands (Powell, 2014). In a similar way SI values can be affected by soil moisture. However, edaphic conditions ultimately control vegetation in floodplains (Bledsoe and Shear, 2000), and therefore SI, by possibly incorporating both vegetation and soil characteristics, can be still used as a proxy to identify different vegetation covers and their temporal dynamics.

Conclusions

This paper presents a remote sensing approach to monitor the dynamics of bottomland hardwood forests and hardwood swamps in the Apalachicola floodplain, Florida USA. The simple tools and indices proposed in this paper may prove valuable for management of floodplain areas located in humid regions. Given the possibility of obtaining the necessary datasets in a straightforward and cost-effective manner, remotely sensed data are useful for monitoring large areas difficult to reach.

We used NDVI data, derived from Landsat TM/ETM+ time series, to categorize different vegetation covers. The novel Seasonality Index (SI) introduced herein is a straightforward index to monitor vegetation dynamics as it may distinguish between hardwood swamp and bottomland hardwood forests. In general, the SI index is excellent at separating forest canopies that have a specific seasonal signal, such those reported in Figure 4. A preliminary

analysis using the NDVI typical year for each vegetation group, as shown in Figure 4, would then determine the applicability of the SI to a specific region.

Assuming that the classification of floodplain vegetation with the SI is reliable, our analysis indicates that substantial amounts of hardwood swamp have been replaced by bottomland hardwood forest in the last thirty years due to a reduction in water flow in the main river and the consequent drying process occurring in its floodplain.

Correlation between minimum water levels in winter and SI could highlight areas undergoing change because of flow alterations. The most sensitive swamp areas are located both in the vicinity of lateral channels activated at moderate water levels and in the vicinity of river banks. The floodplain vegetation composition and structure is driven by multiple parameters. The timing and frequency of inundations and the height of the water levels within the floodplain affect the edaphic conditions and alter the dispersal of seeds and the establishment and recruitment of new individuals (Maxwell and Knapp, 2012). Therefore, the ground elevation plays an important role in determining the vegetation composition: the SI index seems to indicate that the higher banks of the Apalachicola river are dominated by bottomland hardwood forest while the low floodplain areas are characterized by hardwood swamp. The boundary of these two vegetation classes has changed in the last 30 years, with an increase in bottomland hardwood forest and a decrease in swamp forest.

The Apalachicola river basin is considered one of the planet's biodiversity hotspots, with rich and diverse endemic vegetation colonizing the floodplain, ravines and bluffs of the river (Blaustein 2008). If our results are confirmed for the entire river basin, they could indicate large-scale temporal variations in key ecosystems, possibly threatening the entire ecosystem and its biodiversity.

Acknowledgments. This research was partly supported by the donors of the American Chemical Society Petroleum research Fund award # 51128-ND8 and the NSF award DEB-0621014 (VCR-LTER program).

References

- Amoros, C., & Bornette, G. (2002). Connectivity and biocomplexity in waterbodies of riverine floodplains. *Freshwater Biology*, 47(4), 761-776.
- Apalachicola National Estuarine Research Reserve, June 2013, Management Plan, Tallahassee, FL.
- Asrar, G., Fuchs, M., Kanemasu, E. T., & Hatfield, J. L. (1984). Estimating absorbed photosynthetic radiation and leaf area index from spectral reflectance in wheat. *Agronomy journal*, 76(2), 300-306.
- Bayley, P. B. (1995). Understanding large river: floodplain ecosystems. *BioScience*, 153-158.
- Blaustein, R. J. (2008). Biodiversity hotspot: the Florida panhandle. *BioScience*, 58(9), 784-790.
- Bledsoe, B. P., & Shear, T. H. (2000). Vegetation along hydrologic and edaphic gradients in a north carolina coastal plain creek bottom and implications for restoration. *Wetlands*, 20(1), 126-147.
- Brinson, M. M., Lugo, A. E., & Brown, S. (1981). Primary productivity, decomposition and consumer activity in freshwater wetlands. *Annual Review of Ecology and Systematics*, 123-161.
- Darst MR, Light HM. (2008). Drier Forest Composition Associated with Hydrologic Change in the Apalachicola River, Florida. U.S. Geological Survey Scientific Investigations Report 2008-5062; 81pp.
- Edmiston, H.L. and H. Tuck (1987). Resource Inventory of the Apalachicola River and Bay Drainage Basin. Office of Environmental Services, Florida Game and Fresh Water Fish Commission, Apalachicola, Florida. 303 p.
- Faulkner, S., Barrow, W., Couvillion, B., Conner, W., Randall, L., & Baldwin, M. (2005). Impacts of Hurricane Katrina on floodplain forests of the Pearl River. Farris GS, US Geological Survey et al (eds) *Science and the storms: the USGS response to the hurricanes* of, 124-130.
- Florida Fish and Wildlife Conservation Commission (2003) retrieved from http://ocean.floridamarine.org/TRGIS/Description_Layers_Terrestrial.htm on 1 Sept. 2014.
- Gibson CA, Meyer JL, Poff NL, Hay LE, Georgakakos A. (2005). Flow regime alterations under changing climate in two river basins: implications for freshwater ecosystems. *River Research And Applications* 21(8): 849-864.
- Gilbert, T., Stys, B., (March, 2004). Descriptions of Vegetation and Land Cover Types Mapped Using Landsat Imagery. Florida Fish and Wildlife Conservation Commission

- 613 Gould, W. (2000). Remote sensing of vegetation, plant species richness, and regional
614 biodiversity hotspots. *Ecological applications*, 10(6), 1861-1870.
- 615 Hook, D. D. (1984). Waterlogging tolerance of lowland tree species of the south. *Southern*
616 *Journal of Applied Forestry*, 8(3), 136-149.
- 617 Horner, G. J., Baker, P. J., Mac Nally, R., Cunningham, S. C., Thomson, J. R., & Hamilton,
618 F. (2009). Mortality of developing floodplain forests subjected to a drying climate and
619 water extraction. *Global Change Biology*, 15(9), 2176-2186.
- 620 Hosner, J. F. (1960). Relative tolerance to complete inundation of fourteen bottomland tree
621 species. *Forest Science*, 6(3), 246-51.
- 622 Jones, R. H., Sharitz, R. R., Dixon, P. M., Segal, D. S., & Schneider, R. L. (1994). Woody
623 plant regeneration in four floodplain forests. *Ecological monographs*, 345-367.
- 624 Johnson, R. L. (1990). *Nyssa aquatica* L. water tupelo. Burns, RM, Honkala, BH (Tech.
625 Coords.), *Silvics of North America*, 2, 474-478.
- 626 Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river-floodplain
627 systems. *Canadian special publication of fisheries and aquatic sciences*, 106(1), 110-127.
- 628 Keddy, P. A. (2010). *Wetland ecology: Principles and conservation*.
- 629 Kerr, J. T., & Ostrovsky, M. (2003). From space to species: ecological applications for remote
630 sensing. *Trends in Ecology & Evolution*, 18(6), 299-305.
- 631 Keruzoré, A. A., Willby, N. J., & Gilvear, D. J. (2013). The role of lateral connectivity in the
632 maintenance of macrophyte diversity and production in large rivers. *Aquatic*
633 *Conservation: Marine and Freshwater Ecosystems*, 23(2), 301-315.
- 634 Kooistra, L., Wamelink, W., Schaepman-Strub, G., Schaepman, M., van Dobben, H.,
635 Aduaka, U., & Batelaan, O. (2008). Assessing and predicting biodiversity in a floodplain
636 ecosystem: Assimilation of net primary production derived from imaging spectrometer
637 data into a dynamic vegetation model. *Remote Sensing of Environment*, 112(5), 2118-2130.
- 638 Leitman, H. M., Sohm, J. E., & Franklin, M. A. (1983). Wetland hydrology and tree
639 distribution of the Apalachicola River flood plain. Florida. In *Geological survey water-*
640 *supply paper* (Vol. 2196). US Government Printing Office.
- 641 Light HM, Darst MR, Grubbs JW. (1998). *Aquatic Habitats in Relation to River Flow in the*
642 *Apalachicola River Floodplain, Florida*. U.S. Geological Survey Professional Paper 1954;
643 78pp.
- 644 Light HM, Vincent KR, Darst MR, Price FD. (2006). *Water-Level Decline in the*
645 *Apalachicola River, Florida, from 1954 to 2004, and Effects on Floodplain Habitats*. U.S.
646 *Geological Survey Scientific Investigations Report 2006-5173*; 83pp.
- 647 Light, H. M., Darst, M. R., Lewis, L. J., & Howell, D. A. (2002). Hydrology, vegetation, and
648 soils of riverine and tidal floodplain forests of the lower Suwannee River, Florida, and
649 potential impacts of flow reductions. *US Geological Survey professional paper*, (1656A).
- 650 Maxwell, J. T., & Knapp, P. A. (2012). Reconstructed tupelo-honey yield in northwest florida
651 inferred from nyssaogeche tree-ring data: 1850-2009. *Agriculture, Ecosystems and*
652 *Environment*, 149, 100-108.
- 653 Mertes, L. A., Daniel, D. L., Melack, J. M., Nelson, B., Martinelli, L. A., & Forsberg, B. R.
654 (1995). Spatial patterns of hydrology, geomorphology, and vegetation on the floodplain of

655 the Amazon River in Brazil from a remote sensing perspective. *Geomorphology*, 13(1),
656 215-232.

657 North West Florida Water Management District, (2010) retrieved from
658 <http://www.nwfwmdlidar.com> on 1 Sept. 2014

659 Olofsson, P., Foody, G. M., Herold, M., Stehman, S. V., Woodcock, C. E., &Wulder, M. A.
660 (2014). Good practices for estimating area and assessing accuracy of land change. *Remote*
661 *Sensing of Environment*, 148, 42-57.

662 Olofsson, P., Foody, G. M., Stehman, S. V., & Woodcock, C. E. (2013). Making better use of
663 accuracy data in land change studies: Estimating accuracy and area and quantifying
664 uncertainty using stratified estimation. *Remote Sensing of Environment*, 129, 122-131.

665 Osborne, P. E., &Suárez-Seoane, S. (2007). Identifying core areas in a species' range using
666 temporal suitability analysis: An example using little bustard *tetrax tetrax* L. in Spain.
667 *Biodiversity and Conservation*, 16(12), 3505-3518.

668 Parolin, P., Lucas, C., Piedade, M. T. F., &Wittmann, F. (2010). Drought responses of flood-
669 tolerant trees in Amazonian floodplains. *Annals of botany*, 105(1), 129-139.

670 Powell, S. J., Jakeman, A., &Croke, B. (2014). Can NDVI response indicate the effective
671 flood extent in macrophyte dominated floodplain wetlands? *Ecological Indicators*, 45, 486-
672 493.

673 Richard, Y., Martiny, N., Fauchereau, N., Reason, C., Rouault, M., Vigaud, N., &Tracol, Y.
674 (2008). Interannual memory effects for spring NDVI in semi-arid south africa. *Geophysical*
675 *Research Letters*, 35(13704)

676 Rood, S. B., Pan, J., Gill, K. M., Franks, C. G., Samuelson, G. M., & Shepherd, A. (2008).
677 Declining summer flows of Rocky Mountain rivers: Changing seasonal hydrology and
678 probable impacts on floodplain forests. *Journal of Hydrology*, 349(3), 397-410.

679 Sellers, P. J., Berry, J. A., Collatz, G. J., Field, C. B., & Hall, F. G. (1992). Canopy
680 reflectance, photosynthesis, and transpiration. III. A reanalysis using improved leaf
681 models and a new canopy integration scheme. *Remote sensing of environment*, 42(3), 187-
682 216.

683 Sims, N. C., &Thoms, M. C. (2002). What happens when flood plains wet themselves:
684 vegetation response to inundation on the lower Balonne flood plain. *International*
685 *Association of Hydrological Sciences, Publication*, 276, 195-202.

686 Smith, M. C., Stallins, J. A., Maxwell, J. T., & Van Dyke, C. (2013). Hydrological shifts and
687 tree growth responses to river modification along the Apalachicola River, Florida.
688 *Physical Geography*, 34(6), 491-511

689 Sparks, R. E. (1995). Need for ecosystem management of large rivers and their
690 floodplains. *BioScience*, 168-182.

691 Stallins, J. A., Nesius, M., Smith, M., & Watson, K. (2010). Biogeomorphic characterization
692 of floodplain forest change in response to reduced flows along the Apalachicola River,
693 Florida. *River research and applications*, 26(3), 242-260.

694 Stys, B., R. Kautz, D. Reed, M. Kertis, R. Kawula, C. Keller, and A. Davis. 2004. Florida
695 vegetation and land cover data derived from 2003 Landsat ETM+ imagery. Florida Fish
696 and Wildlife Conservation Commission, Tallahassee, Florida, USA.

697 Tewari, S., Kulhavy, J., Rock, B. N., & Hadas, P. (2003). Remote monitoring of forest
698 response to changed soil moisture regime due to river regulation. *J. For. Sci.*, 49(9), 429-
699 438.

700 Tockner, K., & Stanford, J. A. (2002). Riverine flood plains: present state and future
701 trends. *Environmental conservation*, 29(03), 308-330.

702 Tockner, K., Malard, F., & Ward, J. V. (2000). An extension of the flood pulse
703 concept. *Hydrological processes*, 14(16-17), 2861-2883.

704 Turner, W., Spector, S., Gardiner, N., Fladeland, M., Sterling, E., & Steininger, M.
705 (2003). Remote sensing for biodiversity science and conservation. *Trends in ecology &*
706 *evolution*, 18(6), 306-314.

707 Weier, J., Herring, D., Measuring Vegetation (NDVI & EVI), (2000). Retrieved from:
708 <http://earthobservatory.nasa.gov/Features/MeasuringVegetation/printall.php>

709 Wen, L., Yang, X., & Saintilan, N. (2012). Local climate determines the NDVI-based primary
710 productivity and flooding creates heterogeneity in semi-arid floodplain
711 ecosystem. *Ecological Modelling*, 242, 116-126.

712 Wharton, C. H., Kitchens, W. M., Pendleton, E. C., & Sipe, T. W. (1982). Ecology of
713 bottomland hardwood swamps of the southeast: a community profile (No. FWS/OBS-
714 81/37). Georgia Univ., Athens (USA). Inst. of Ecology; Fish and Wildlife Service, Slidell,
715 LA (USA). National Coastal Ecosystems Team; Wabash Coll., Crawfordsville, IN
716 (USA). Dept. of Biology.

717 Zhu, Z., & Woodcock, C. E. (2012). Object-based cloud and cloud shadow detection in
718 Landsat imagery. *Remote Sensing of Environment*, 118, 83-94.