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Research papers

Using biodiversity response for prioritizing participants and service provisions in a payment-for-water-storage program in the Everglades basin

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ABSTRACT

Hydrological changes can lead to biodiversity responses that are complex and challenging to quantify. We present a framework that uses biodiversity as a criterion to address “where to buy” and “how much to buy” in a payment-for-environmental-services (PES) program focused on water storage service in the Everglades basin, USA. The PES program was designed to pay for added water storage on private cattle ranchlands by raising the spillage level in drainage control structures to reduce surface flows. We predicted that increased hydration of previously drained wetlands would benefit biodiversity, a previously unquantified but desirable co-benefit of the original program, and that a PES program offering bundled services (e.g., storage and biodiversity) can better achieve restoration goals. We quantified desirable biodiversity services (abundance of native flora and fauna such as cover of wetland and forage plants, and abundance of fishes, amphibians, and macroinvertebrates), dis-services (e.g., abundance of invasive plants and mosquitos), and hydrologic signatures (e.g., wetland water depth, inundation area and duration) at four ranches to develop eco-hydrological relationships (models) between hydrological changes and biodiversity responses. Next, a hydrologic model (MIKE-SHE/MIKE-11) was used to predict surface and subsurface water levels and flows and resulting wetland hydrologic signatures for 13 water storage alternatives on the ranches, which were used as example PES proposals. A decision-support-system (DSS) was developed to integrate (i) storage predicted by the hydrologic model, (ii) biodiversity responses predicted by eco-hydrologic models, and (iii) a user-defined preference scheme to assign importance weights to storage, biodiversity, and implementation cost. The DSS calculated a cumulative score for ranking PES proposals. By considering desirable services and dis-services, stakeholders can decide on their preferred level of services, e.g., buyer(s) may settle for less storage if there is a gain in desirable biodiversity. The DSS can identify trade-offs among services, helping stakeholders negotiate.

1. Introduction

Agricultural lands have traditionally been valued for food, fiber, and fuel production (Clément et al., 2017; Pastor et al., 2019) but their role in regulating and sustaining environmental services and enhancing biodiversity has also been receiving increased attention (Norris, 2008; Gonthier et al., 2014; Huntsinger and Oviedo, 2014; Page and Bellotti, 2015; Boughton et al., 2019). Agricultural lands, including ranches, often contain wetlands which are considered biodiversity hotspots and a hub of environmental services (Swain et al. 2013). However, loss of these wetlands to support agricultural intensification has also received widespread attention (Thiere et al., 2009). Almost 35% of wetlands have been lost between 1970 and 2015, worldwide (WWF, 2018). Loss of wetlands has significantly impacted freshwater biodiversity (Kingsford et al., 2016; Arntzen et al., 2017) and globally, 25% of wetland-dependent plant and animal species are at risk of extinction (Ramsar

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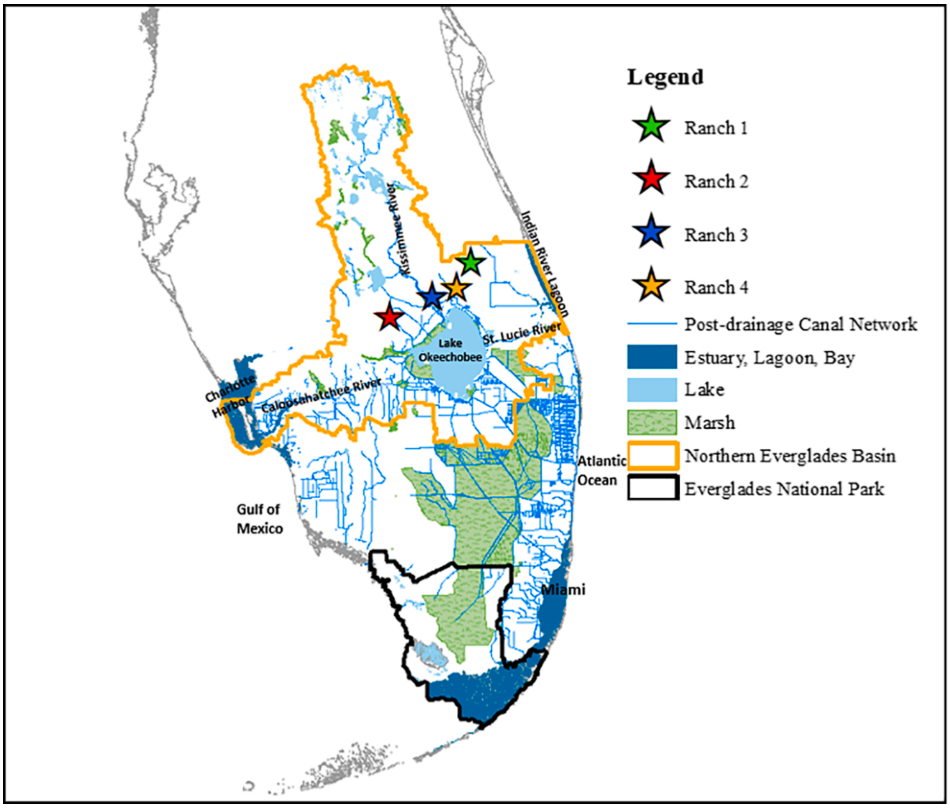

Fig. 1. Locations of study sites relative to Lake Okeechobee, FL.

Convention on Wetlands, 2018). Draining wetlands, for agricultural and urban development, has led to degradation of ecosystems in watersheds such as the Everglades (NAS, 2021), Chesapeake Bay (Steven and Lowrance, 2011), Mississippi River (Gleason et al., 2011), and China's Yangtze River (An and Verhoeven, 2019).

The payment-for-environmental-services (PES) concept is increasingly becoming a sought-after ecosystem restoration approach (Yin and Zhao, 2012; Huang et al., 2018; Ruggiero et al., 2019) and is applicable to wetland restoration projects. The philosophy of PES programs is that in certain situations, providing financial incentive is more effective than implementing a regulatory measure (Zanella et al., 2014). Promoted as a "market-like" approach, PES programs are usually conceptualized with a sense of simplicity. However, implementing them is complicated (Zanella et al., 2014). The foundation of PES programs is their voluntary nature such that a seller willingly sells a well-defined service to a buyer (Wünder, 2005). Although a core principle of the PES theory, there are exceptions to the voluntary participation especially when intermediary parties between the seller and buyers are involved (Vatn, 2010). For our study, we consider the state (public) as the buyer and the voluntary nature of the PES programs holds true. The willingness to participate provides freedom to the sellers, however, it also puts them all in one pool, making it harder to compare cost efficiency and service provision levels (Jack and Jayachandran, 2019). While quantity of service provision is the primary criterion for awarding PES contracts, the synergies and trade-offs among other associated services and the potential risk of dis-services because of interactions among physical and ecological systems, receives little to no attention (Wünscher et al., 2008). Reasons for this lack of attention mostly stems from the difficulty in quantifying the trade-offs and synergies.

Changes in hydrologic patterns are known to impact wetland ecological processes and biodiversity (Konar et al., 2013). Although hydration of drained wetlands may lead to enhanced wetland habitat and desirable biodiversity, recent studies have also linked alterations in hydrologic conditions and patterns to increases in invasive species (Stokes et al., 2010; Catford et al., 2011; Nielsen et al., 2013). The conflict between positive and negative impacts of wetland hydration on biodiversity needs to be evaluated by simultaneous consideration of services and dis-services. The extent of impacts, positive and negative, may change from site to site for the same or different levels of hydrologic changes depending on multiple factors including land use and management. For example, Boughton et al. (2019) showed that ranch management practices in Florida can affect biodiversity responses to hydrological changes. Therefore, the optimum level of hydration that supports native flora and fauna while minimizing invasive species may differ by location.

The challenge for PES in wetland settings targeting biodiversity is that the ecological interactions are complex and can be difficult to generalize, especially as a response to something as complex as changing water storage on landscape that changes water availability including area, volume and hydroperiod on wetlands. It has been proposed that PES accounting should only consider the final "service" product in favor of greater traceability of the program impact (Brauman et al., 2007). It is also suggested that instead of considering PES programs purely for their "paid service' and economic benefits, they be designed with consideration of how biodiversity responds to management actions (Bullock et al., 2011).

The Everglades is a globally important watershed and public interest in hydrological restoration makes it an appropriate ecosystem in which the role of PES can be evaluated. The Everglades was once, and to an extent still is, a vast network of wetlands and rivers supporting rich subtropical vegetation and wildlife (Lemaire and Sisto, 2012). Dramatic alteration of landscape drainage to improve flood management, urban water supply, and agricultural production has led to the decline of the Everglades ecosystem. Historically, the Kissimmee River, Lake Okeechobee, and Everglades constituted the greater Everglades ecosystem which accounts for most of the sub-tropical south Florida area (Fig. 1). Pre-1920s the Kissimmee River formed a significant proportion of the headwaters of the greater Everglades system feeding Lake Okeechobee

(Fig. 1; McVoy et al., 2011; Steinman and Rosen, 2000). In addition to flowing south through sheet flow, the lake water also moved slowly through the Caloosahatchee River and downstream through other small rivers to coastal estuaries and eventually to the Gulf of Mexico and the Atlantic Ocean (McVoy et al., 2011). Development of extensive ditch networks to drain the watershed began in the 1880s (Fig. 1) which resulted in fast movement of unnaturally high flows and nutrient loads to the Lake, rivers, estuaries, and the southern Everglades (Steinman and Rosen, 2000; Graham et al., 2015).

Water flowing into Lake Okeechobee from the north (Fig. 1) accounts for 92% of the total inflow to the lake. Current estimates are that storage of 750,000 acre-feet of water is required, north of the lake, to significantly reduce excessive discharges from the Lake to the estuarine systems and the Everglades (SFWMD, 2021a). While several state-funded projects including reservoirs and aquifer storage recovery (ASR) have been planned or started, they are not enough to meet the water storage goals, hence there is a great deal of interest in developing PES solutions for the watershed to complement other initiatives.

In 2005, a coalition of state agencies, environmental organizations, researchers, and private landowners developed and implemented a PES program known as the Florida Ranchland Environmental Services Program (FRESP) in the Northern Everglades basin (Bohlen et al., 2009). The idea of FRESP was to hold water on private ranchlands by reducing surface flows from them (Lynch and Shabman, 2011). The rationale behind FRESP was to obtain some of the needed storage by dispersing it throughout the Northern Everglades basin, complementing other water storage systems such as reservoirs and constructed treatment wetlands. Under FRESP, ranchers installed discharge control structures, commonly known as culverts with riser boards (CRB) and raised the spillage level by adding boards. The seller of services was the rancher, and the buyer was the state of Florida through the South Florida Water Management District (SFWMD) that is responsible for managing and protecting water resources in the Everglades basin. The FRESP, first of its kind PES implemented within the United States, was a pilot program that laid the foundation for the basin-wide launch of the Northern Everglades - Payment for Environmental Services (NE-PES) program (Bohlen et al. 2009). The NE-PES now has 15 sites and provides an estimated 102,512 acre-feet of water storage annually (SFWMD, 2021b). Under the NE-PES the term “water retention” better describes the “storage” service since the surface water that is retained eventually leaves the ranch, through subsurface flow and evapotranspiration. The term water retention has been simplified to a more generic term of “water storage” for communicating to general public and other stakeholders and is used here onwards. Increasing the spillage level on ranch lands was expected to alter the hydrological linkages between uplands and wetlands which in turn affects vegetation and wildlife diversity (Shabman et al., 2013).

In the case of the NE-PES, trade-offs and synergies exist between storage level, and desirable and undesirable biodiversity (Boughton et al 2019). There is a difference of opinion among researchers regarding biodiversity’s status as an ecosystem service (Haines-Young and Potschin-Young, 2010; Mace et al., 2012; Duncan et al., 2015). The disagreement stems from a variety of reasons; primary reason being the complexity in biodiversity valuation due to a host of parameters and its trans-disciplinary nature (Mace et al., 2012). However, we believe that the status of biodiversity as a service should be program specific as suggested by Mace et al. (2012). Biodiversity can be considered as a regulator of ecosystem functions (e.g., carbon sequestration), stand-alone ecosystem service (e.g., restoration of native flora and fauna) or a valued product (e.g., aesthetically pleasing view facilitating tourism).

Although biodiversity enhancement was an expected impact of reducing ranch-scale drainage which in turn increased the wetland hydration under the payment for water storage program, it was recognized as difficult to quantify. There was no provision to include biodiversity as a factor either for deciding where to implement a PES contract or how much to pay for it. More importantly, there was no willing buyer (e.g., conservation stakeholder group such as an NGO or a federal/state agency) for enhanced biodiversity. Even if there were willing buyers, “double dipping” would be a legitimate concern. “Double dipping” i.e., selling environmental benefits from a PES program in multiple markets, is typically not allowed (Woodward, 2011). Given these concerns, we decided at a later stage that biodiversity be considered a non-economic but valued product of the program, and relationships between hydrological and ecological changes be developed to quantify biodiversity response to increase in water storage. Although, biodiversity enhancement is not a benefit for which there is a buyer, we argue and demonstrate that these relationships should be the drivers of decisions about “where to implement (site selection) projects” and “how much to store (spillage level).” Furthermore, multiple stakeholder preferences play a key role in implementation of PES programs. In case of the NE-PES, the diversity of stakeholders (e.g., agricultural producers, local government, environmental agencies, and NGOs) in the coalition is significant, and their preferences for service valuation is likely to vary. This makes the decision-making process, where biodiversity is also a factor, more challenging from a policy standpoint. The correlation between water storage and biodiversity enhancement is not necessarily linear which is why selecting sites solely based on amount or unit price of storage may not work for a multi-service PES program targeting both water storage and biodiversity enhancement. Therefore, a tool to quantify trade-offs between storage and various biodiversity indicators on the basis of user preference is needed. Our overarching goal was to develop an approach that is a result of the interplay between physical (water storage), ecological (biodiversity), agricultural (food production), and social (stakeholder preference) components that articulates the synergies and trade-offs for all. The steps toward this goal included development of models to mathematically quantify the relationships between the said components and a decision support system (DSS) tool to integrate them with stakeholder preference to identify desirable sites among a pool of eligible PES applicants, and optimum water storage levels. Biodiversity enhancement as a result of water storage under the PES scheme is considered a non-economic but valued product in this study. However, if there were willing buyers for it, biodiversity could be considered a focal ecosystem service and the mathematical models could be used to quantify the service provisioning.

Our specific objectives are to: 1) Refine eco-hydrological relationships drawn from our earlier work on four ranchlands in the Everglades basin by Boughton et al (2019); 2) develop eco-hydrological models for predicting biodiversity response to changes in water storage; 3) develop a DSS that combines predictions from eco-hydrological and hydrologic models with user-defined weighting schemes representing their preference for offered ecosystem services; 4) use the DSS to assist in ranking sites interested in participating in the PES program as well as deciding the level of water storage service that would maximize the desirable and reduce the undesirable biodiversity response; and 5) identify the actors and their preferences when selecting PES projects and demonstrate how trade-offs can be negotiated given the assimilation of these proclivities in the DSS tool.

Table 1
Study site characteristics.

Study Site	Ranch Area (ha)	Ranch Type	Wetlands Monitored	Study Wetland Area (ha)
Ranch 1	1,358	Semi-native pasture	3	1.28 – 4.56
Ranch 2	4,250	Semi-native/ Improved pasture	8	0.24 – 0.78
Ranch 3	275	Improved pasture	2	7.5 – 12.9
Ranch 4	3,683	Improved Pasture	2	6.55 – 31.03

Table 2
Grouped biodiversity metrics for eco-hydrological model development. Plant cover refers to percent cover of plant type per square meter (percent cover/m^2) and abundance refers to the number or count of each species per unit area. Data were the average of all square meter plots sampled per wetland for a sampling event (see Boughton et al. (2019) for details).

Grouped Biodiversity	Description*
Plants	
Forage	Average cover of all forage plants
Wetland	Average cover of all emergent macrophytes
Exotic	Average cover of all non-native plants
Animals	
Fish	Average abundance of all fish species
Amphibian (frog)	Average abundance of all amphibian species
Macroinvertebrate	Average abundance of all macroinvertebrate species
Mosquito	Average abundance of all mosquito species

* See Appendix Table A-1 for details on individual measured plant and animal species

2. Methods

2.1. Study sites and site selection

Four ranches (Ranch 1 through 4) in the Everglades basin were selected for ecological measurements to evaluate the effects of changes in ranch-scale water storage on biodiversity of wetlands embedded within ranchlands and develop eco-hydrologic models for predicting the biodiversity responses (Fig. 1; Table 1). Site selection was based on participation in the FRESP pilot PES program, availability of hydrologic data, landowners' willingness, and/or representativeness. Ranch 1, 2, and 4 participated in the NE-PES program while Ranch 3 was the study site for another similar ranchland water storage research project (Shukla et al., 2011; Shukla et al. 2014; Goswami and Shukla, 2015; Wu et al., 2016). Land use, land management, wetland biodiversity, and hydrology of the four ranches selected for the study represent typical ranchlands within the Northern Everglades basin. Each is a representative cow-calf ranching operations, maintaining a permanent herd of breeding cows to raise calves for transfer to stocker grass and feedlots and eventual market.

Land use at the four ranches comprised of semi-native pastures and improved pastures. Improved pastures receive regular inputs of fertilizers and amendments (e.g., lime) and have higher density of ditches and swales to facilitate both drainage as well as irrigation (sub-irrigation). Semi-native pastures represent a lower intensity grazing operation compared to improved pastures. Improved pastures also involve establishment of agronomic forage grasses and have higher cattle stocking density as compared to semi-native pastures which makes them a more intensive production system.

With a subtropical climate, the average rainfall for the basin is 1362 mm/year. Average minimum temperature of the region is 17 °C (62°F), and the average maximum temperature is 29 °C (84°F). The dominant soil type at the four ranches was spodosol; sandy soils with nearly level topography, shallow water table, and poor drainage.

The outflow locations at all ranches were equipped with CRB structures that enabled the changes in the spillage level at the outlet to reduce surface outflow volume and rate and increase the surface and subsurface storage. Ecological and hydrologic data were collected at 15 wetlands (Table 1) on the four ranches; these wetlands varied in area and hydrologic conditions (Table 1). Selecting different types of ranches as well as wetlands allowed us to develop generic eco-hydrological models that are representative of the Northern Everglades basin. The 15 wetlands ranged from 0.24 − 31 ha in size and were previously drained through intense ditch networks for maximizing pasture area.

Table 3
Hydrologic variables used as input for developing wetland scale eco-hydrological models.

Hydrologic Variable	Definition
Days Connected (DC)	The number of days a wetland is connected to the main ditch network within the ranch. A wetland is considered connected if the difference between the maximum water level in the wetland is greater than the spillage level of the ditch network.
Days Not Connected (DNC)	The number of days a wetland is not connected to the main ditch network. A wetland is considered not connected if the maximum water level in the wetland is less than the spillage level of the ditch network.
Time Inundated (TI)	The number of days a wetland is inundated. Time inundated was derived by a comparative analysis of lowest elevation observed in the wetland and 15-minute water elevation data collected at the wells that recorded surface/groundwater levels (Shukla et al., 2011). Time inundated was the length of time in days that the difference between water elevation and the minimum elevation in the wetland was > 6 in. (15 cm).
Inundation Area (IA)	Area of the wetland covered in water. Measured water depths and the DEMs (LiDAR and ground topographic survey) were combined within ArcGIS v.10.1 to estimate inundation area. Inundation areas were estimated for wetlands on four ranches for the dates of ecological sampling, calculated using a Python script within ArcGIS using water depth.
Volume (Vol)	Volume of water in the wetland was calculated by developing Volume-area-depth equations.
Maximum Depth (MaxDepth)	The maximum depth in a wetland for a specific sampling date.

2.2. Selection and categorization of biodiversity and agricultural production indicators

Biodiversity response indicators were chosen based on their ease of measurement, relevant for the scale of measurement, expected response to changes in wetland hydrology, significance to maintaining ranch operations under extended hydroperiods, and significance for other higher trophic level organisms (e.g., mosquito larvae and fish, and fish and macroinvertebrates attracting wading birds), as well as ability to develop empirical relationships that can be used within the DSS. Specific biodiversity indicators included percent cover of native wetland plants, and exotic plants, as well as abundance of fish, amphibians, aquatic macroinvertebrates, and mosquitos (Table 2). The percent cover of planted forages was used as an indicator of the NE-PES project's impact on agricultural production. The wetland plant indicator included emergent macrophytes (e.g., *Sagittaria graminea, Sagittaria lancifolia, Sagittaria latifolia, Pontedaria cordata, Juncus effusus, Thalia geniculata*). The exotic plant indicator group included non-native plants (e.g., *Ludwigia peruviana*). The forage indicator group included all forage species, planted or wetland (e.g. *Panicum hemitomon*) grasses known to be palatable to cattle. The animal responses were grouped into fish, amphibians, macroinvertebrates, and mosquitos. A full list of plant and animal species are presented in the Appendix (Table A-1). Detailed methods for ecological data collection and analyses are provided in Boughton et al. (2019).

2.3. Ecologic and hydrologic data collection for Eco-hydrological modeling

Wetlands were sampled multiple times (1–9, median = 6 per wet season, June-October) for measuring hydrologic (surface water elevation) and biodiversity indicators during the wet season for two years (2010–2011) to account for seasonal variation. From late fall through the spring months (November – April), the region typically has cooler temperatures receiving only 30% (408 mm) of the total annual rainfall. This results in drought-like conditions; shallow wetlands dry out and ecological activity slows. The summer and early fall months (May – October) are typically warmer with most of the rainfall occurring in this time (70%; 953 mm) resulting in hydrated and ecologically active wetlands. June through mid-November is hurricane season in the region which can result in flooded conditions. The monitoring period included periods of prolonged drought and flooding thus making the eco-hydrological models appropriate for variety of weather and hydrologic conditions. Specific hydrologic variables were either measured or derived from measured data are presented in Table 3.

2.3.1. Topography

The digital elevation models (DEMs) for Ranch 1 and Ranch 4 were developed using the topographic data collected during a ground survey using a Trimble S6 (Trimble, Westminster, CO, USA). The LiDAR data collected by the National Center for Airborne Laser Mapping at University of Florida were used to develop DEMs for Ranch 2 and 3 (Guzha and Shukla, 2012; Shukla et al., 2011). These DEMs were used for estimating hydrologic variables (e.g., inundation area and time inundated) and calibration and validation of hydrologic models. The vertical accuracy of topographic data was 10–20 cm.

2.3.2. Surface and groundwater levels

Manual water depth measurements were collected in the wetlands between 2010 and 2012. Depth of water in the wetland was measured using a metric ruler at each sampling location at the time of ecologic sampling. The sampling locations were randomly selected and were recorded using a Trimble GeoXT GPS (Trimble, Westminster, CO, USA). Maximum wetland depth was determined for each sampling date by comparing all depth measurements collected during a particular day. In addition to the event-based manual measurements, 15-minute groundwater/surface water levels were measured using pressure transducers during the 2010–2012 period. Higher-frequency water elevation (above mean sea level) data were used to calibrate and validate the hydrologic model as well as estimate the hydrologic connectivity between upland and wetland areas for Ranches 2 and 3 discussed in section 2.4 below.

2.4. Wetland Inundation, Volume, and hydrologic continuum

The hydrologic variables used in the study were chosen based on available literature and discussion between co-authors (hydrologists and ecologists). Using the measured wetland water depth and groundwater elevations, other hydrological parameters (e.g., days inundated, wetland water volume) needed for the eco-hydrologic models were derived (see details in Table 3).

2.5. Development of Eco-hydrological models

Development of eco-hydrological models is an extension of ranch-specific relationships developed by Boughton et al. (2019) which identified the synergies and trade-offs among water storage and biodiversity services. The ecologic variables/biodiversity indicators in our study were regrouped from the standpoint of using biodiversity for inclusion in the DSS for stakeholders designing PES programs and/or selecting PES sites and a specific water storage service level (spillage level within the CRB). We developed mathematical functions based on the eco-hydrological relationships as they related to the new groupings, essential for developing the DSS. The most informative mathematical models and variables were identified based on Akaike's Information Criterion (AIC; Burnham and Anderson, 2001).

The AIC identifies the best model from a set of possible models by assessing how well each explains the variation in the data while penalizing for model complexity. Generalized linear mixed-effects abundances were used because wetlands were random effects across the landscape and within ranches, meaning the wetlands were considered a sample from a population of wetlands. Negative binomial, Gaussian, or zero-inflated negative binomial distributions were used where applicable.

Potential models included additive and interactive effects, though interactions were included only if it was expected that they were important. In all sets of models, a null model was compared that included an intercept term only (e.g., amphibian = 1) and univariate models which contained each independent (hydrologic) variable only (e.g., only TI, only Vol, only Maxdepth, etc.). Furthermore, because land use history on the ranches differs, ranch effects were considered in model development. Ranch effect variables were Boolean, one (1) if present and zero (0) if not. If these models are applied for a specific ranch with pastures that have been described similarly to those in Table 1, then one (1) should be used, otherwise the ranch effect should be zero (0). All analyses were conducted in R v 2.15.3 using the libraries glmmADMB (Fournier et al., 2012) and bbmle (Bolker, 2014) for wetland scale analyses of mean counts (abundances) of animals and mean plant cover. To scale from point to wetland scale, mean counts of animals and mean plant cover were used and were the average of the animal abundance or plant percent cover for the measurement date, respectively. Once the most informative models were identified from the AIC analysis, best models were analyzed and estimates (i.e., coefficients) and p-values were interpreted. The p-value was used to ensure that the independent variables included in the models were meaningful in predicting the biodiversity responses. Model diagnostics included plots of residuals vs. predicted values and quantile–quantile (q-q) plots, where q-q plots were used to verify data distributions. Omega values were computed to assess goodness of fit (Xu, 2003); the residual variance of the full model was compared against the residual variance of a (fixed) intercept-only null model. For models where omega values could not be computed (all plant models), the residual variance of the full model was compared to the residual variance of the intercept-only null model (Bolker, 2014). The root mean square error for each model was calculated to evaluate the model performance.

2.6. Hydrologic modeling

Hydrologic variables (e.g., days connected; Table 3) needed for predicting biodiversity responses (e.g., fish abundance) for different DSS applications were estimated using daily predictions (e.g., wetland water depth) from a hydrologic model, MIKE-SHE/MIKE-11. There are other models which could be used for the purpose of predicting water levels and flows but we were able to capitalize on two previous MIKE-SHE/MIKE-11 modeling studies conducted at Ranch 2 (Shukla et al., *unpublished data*) and Ranch 3 (Wu, 2014) to evaluate the effects of wetland water retention on surface and subsurface flows at wetland and ranch scales. Ranch 2 monitoring started in 2008 as part of the FRESP pilot PES program (Shukla, 2012). Ranch 3 monitoring started in 2003 as part of a wetland water retention project by the University of Florida (Shukla et al., 2011, Shukla et al. 2014).

MIKE-SHE is an integrated, physically based model (Refsgaard and Storm, 1995; Jaber and Shukla, 2012) capable of simulating the surface and subsurface processes including recharge and evapotranspiration. It can simulate surface and subsurface water interactions to predict surface and ground water levels and flows to rivers, flood plains, canals, and other inland water bodies (Jaber and Shukla, 2012). It is suitable for simulating the highly interactive surface water and ground water system prevailing in the Everglades basin with shallow water table, conductive sandy soils, and highly engineered drainage system of ditches, canals, and pumps to achieve multiple goals of flood control, land drainage, and water supply. MIKE-SHE can be coupled with a 1-D river (channel) hydraulics model MIKE-11 to simulate flows from drainage networks containing hydraulic structures (e.g., culvert, flashboard structures, pumps, and gates) prevalent in the Everglades basin. This feature of MIKE-11 is especially useful in accurately representing number of boards in the CRB structures installed at the ranch outlet. Given that MIKE-SHE/MIKE-11 is a spatially explicit model, appropriate cell sizes can be used to represent ponded features (e.g., wetlands, ditches) of

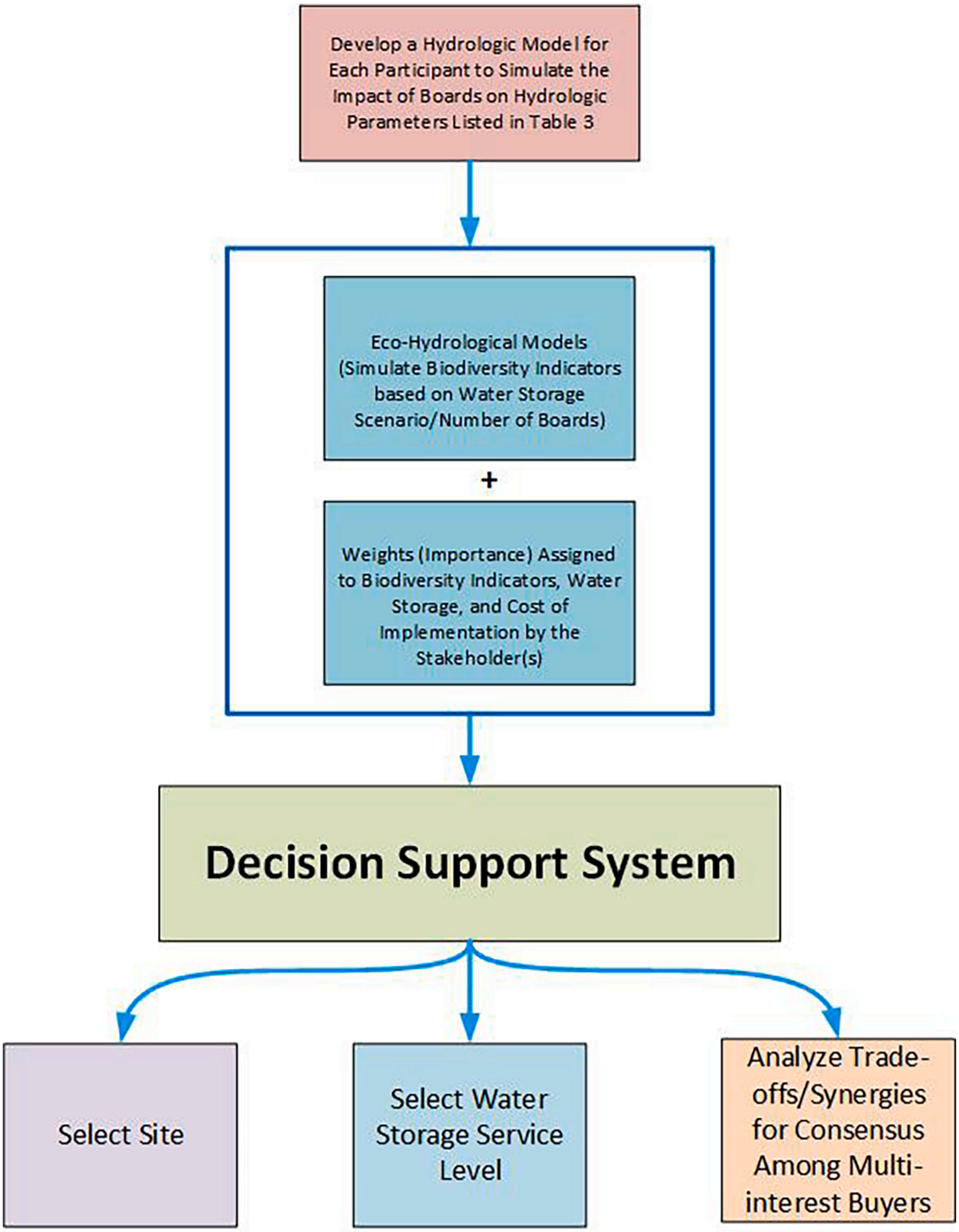

Fig. 2. A flowchart showing the requirements (hydrologic metrics, eco-hydrologic models, and user-defined weights of importance) for development and use of the DSS for payment-for-environmental-services projects.

varying area. The coupled model enables the quantification water retention (storage), wetland hydroperiod, and other hydrologic signatures (Table 3) at wetland and ranch scales for their use by the eco-hydrological models within the DSS.

The MIKE-SHE/MIKE-11 models were developed and evaluated using topographic, land use, weather, soil hydraulics, channel geometry, drainage structures and operation, and surface water and ground water level data collected at Ranch 2 and 3. Prediction statistics for calibration and validation periods, reported in Shukla et al. (unpublished data) and Wu (2014), are briefly discussed here.

The Nash-Sutcliffe efficiency (NSE) and Percent Bias (PBIAS) were used to evaluate model performance. The Nash-Sutcliffe efficiency (NSE) is a statistic that normalizes the observed and simulated data and calculates the amount of variance in simulated data compared to variance in measured data and shows how well the scatterplot of observed versus simulated data fits the 1:1 line (Moriasi et al., 2007). Values of NSE can range from - ∞ to 1 with 1 being the best possible value. Although the NSE is widely used to assess model performance, poor hydrologic models can have a high NSE (Jain and Sudheer, 2008) therefore other performance indicators should also be considered (Clark et al., 2021). Percent bias (PBIAS) is especially useful since it can clearly indicate poor model performance (Gupta et al., 1999). The PBIAS is a measure of how much the simulated data tends to be overestimated (negative values) or underestimated (positive values) as compared to the

Table 4
Weighting schemes based on buyer's preference – biodiversity response vs water storage. A weight of 1 represents least preferred and 4 represents the most preferred.

Criteria	Business as Usual	Importance Weighting Schemes	
		Water Storage Focused	Biodiversity Focused
Water Storage Payment	4	4	1
Cost of Implementation	0	1	1
Wetland Plant Abundance	0	1	4
Forage Plant Cover	0	4	4
Exotic Plant Cover	0	1	1
Fish Abundance	0	1	4
Mosquito Abundance	0	1	1
Amphibian Abundance	0	1	4
Macroinvertebrate Abundance	0	1	4

observed data, and a value of 0.0 is the best possible value. Model performance at Ranch 2 and 3 were based on the criteria developed by Moriasi et al. (2015): *very good* ($1 \geq$ NSE > 0.75; PBIAS $< \pm 5$), *good* ($0.75 \geq$ NSE > 0.65; $\pm 5 \leq$ PBIAS $\leq \pm 10$), and *satisfactory* ($0.65 \geq$ NSE > 0.50; $\pm 10 \leq$ PBIAS $\leq \pm 15$).

The MIKE-SHE/MIKE-11 calibration period for Ranch 2 was August 1, 2008 to May 31, 2009 and the validation period was June 1, 2009 to May 31, 2011. The model performance at Ranch 2 was "satisfactory" per NSE (0.59) and "very good" per PBIAS (0.71) in simulating ditch water levels (Shukla et al., unpublished data). For the validation period, model performance was rated "very good" (NSE = 0.82; PBIAS = -0.35). For Ranch 3, model performance was rated "very good" for both calibration (October 1, 2008 to October 31, 2009; NSE = 0.90; PBIAS = -0.54) and validation (November 1, 2009 to May 31, 2011; NSE = 0.88; PBIAS $<$ 0.1) periods (Wu, 2014). The PBIAS values were much lower than the minimum threshold for "very good" performance (PBIAS < 5). Overall, the model performed well in predicting the ditch water levels, the main driver for the wetland water availability and used for estimating ranch-scale storage.

2.7. Development of decision support system

The decision support system (DSS) tool was developed to integrate trade-offs among biodiversity services and dis-services with water storage services for selecting the PES sites as well as the level of service. The DSS utilizes hydrological and eco-hydrological models developed using data from the pilot PES sites which participated in the FRESP and one wetland water retention site (Fig. 2). The DSS was designed to help: 1) choose where to implement the water storage alternatives i.e., prioritizing PES proposals based on biodiversity response, 2) compare water management alternatives possible at a participating ranch and identify water storage alternative which maximizes the services (e.g., increase in fish population) while minimizing the biodiversity disservices (e.g., increase in exotic vegetation cover, increase in mosquito abundance), and 3) identify the most desirable alternative in a case when the buyer is a coalition of stakeholders with diverse interests (e.g., agricultural production, water storage and availability and maintaining minimum/maximum freshwater flows to estuaries, environmental protection, tourism, etc.), by providing DSS rankings for negotiations to build a consensus (Fig. 2). An example of multi-interest buyer would be the water management districts in Florida whose governing boards include members from different counties representing the interests of residents of their respective regions/counties and industries driving their local economies.

The decision support system (DSS) developed for our study was modified from Thornton et al. (2005) which is a multicriteria decision analytics (MCDA) tool that uses a spreadsheet approach. The simpler spreadsheet (MS-Excel) approach was adopted to make it easy-to-use for the buyers, sellers, and other stakeholders that included ranchers/farmers, NGOs and state agencies. We modified this MCDA tool by adding biodiversity (eco-hydrological models), hydrologic (water storage and other water availability predictions from MIKE-SHE/MIKE11), and economic (implementation cost, payment for water storage) components. We also added the criteria to evaluate the preferences of stakeholders for the PES program. The criteria included water retention payment to ranchers (sellers), cost of implementation, wetland plant abundance, forage cover, exotic plant cover, fish abundance, mosquito abundance, amphibian abundance, and macroinvertebrate abundance. The stakeholders can assign importance to the criteria through simplified weights of importance (Table 4). Weights can range from 1 to 4, where 1 is least important and 4 is most important. The DSS uses three techniques for weighting and ranking alternative decisions: the weighted average method (WAM) (Abdelrahman et al., 2008), the discrete compromise programming (CP) method (Zeleny, 1973), and the preference ranking organization method for enrichment evaluation (PROMETHEE) (Brans and Vincke, 1985). The DSS penalizes the alternative which increases the disservices (e.g., mosquito abundance) by assigning a negative weight to the specific disservices when calculating a normalized score used by the three weighing techniques. These different techniques allow users to quickly assess the potential sensitivity or robustness of alternative ranking techniques given specific decision criteria values.

The DSS framework was built based on two main assumptions: 1) the eco-hydrological models are applicable to all ranches and hydrologic conditions observed within the basin i.e., they can simulate the biodiversity response for any water storage/number of boards alternative, and 2) the hydrologic variables required to predict biodiversity responses are provided by the PES applicants (landowners) in addition to the water storage so the buyer (e.g., state agency) can utilize the DSS to rank the proposals (Fig. 2). Alternatively, the predictions of the biodiversity and storage can be made by the buyer. The hydrologic variables can be predicted through a variety of hydrologic models chosen by the ranchers or their consultants. Irrespective of the method chosen to compile hydrologic parameters, some monetary investment will be required from the applicants responding to a call for proposals by the buyers. All applicants for the pilot FRESP hired an engineering consultant that provided the topographic maps and estimates of storage. Details on different components of the DSS, linkages and their use in tradeoff analyses are presented in Appendix (Table B-1).

2.8. Using decision support system

The availability of the MIKE-SHE/MIKE-11 models for Ranch 2 and 3 enabled us to develop a variety of water storage alternatives. Each modeled alternative represented a specific spillage level implemented in the field by adding boards in the CRB structure. Seven water storage alternatives were considered at Ranch 2 and six at Ranch 3. A total of 13 alternatives were simulated using MIKE-SHE/MIKE-11. For demonstrating the use of the DSS for selecting PES sites, these 13 alternatives were assumed to represent 13 different ranches. The alternatives at Ranch 2 included CRB structure with 0 (baseline), 1, 2, 3, 4, 7, and 8 boards (each board was 15.24 cm high). The alternatives at Ranch 3 included CRB structure with 1, 2, 3, 4, 5, and 6 boards. Due to differences in ditch (e.g., depth and width) and pasture (ground elevation, management) characteristics, the water storage alternatives (number of boards) at Ranches 2 and 3 did not correspond to the same spillage level.

2.8.1. DSS use for selecting PES sites

As described earlier, field-verified MIKE-SHE/MIKE-11 models were not available for Ranch 1 and 4. To demonstrate how different services and dis-services can be combined for decision making for a PES program using the DSS, we assumed that the 13 water storage alternatives represented 13 PES proposals or sites (ranches). Although the 13 proposals

Table 5
Eco-hydrological models developed using Akike information criterion (AIC) model selection techniques. TI = Time inundated (days); Vol = Wetland water volume (m^3); GDD = Growing degree days (days); DC = Days connected (days); Maxdepth = Maximum wetland depth (cm); Fish = fish count per unit area. Ranch2, Ranch3, and Ranch4 are ranch effect variables defined by "Ranch Type" in Table 1. The ranch effect variables are Boolean variables – their value is 1 if the ranch type being evaluated is the same as those described in Table 1 for respective ranches and 0 if not.

Ecological Response	N[a]	Hydrologic Variables	R^2	AIC Value	Model	RMSE	P Value	Units
Log (Forage)*	21	TI	0.60	250.2	$2.53 + 9.57E\text{-}03{*}TI - 4.17E\text{-}05{*}TI^2$	0.74	<0.001	Percent cover of forage per m^2
Log(Exotic)*	22	Vol	0.55	284	$0.52–4.57E\text{-}05{*}Vol + 1.52E\text{-}09{*}Vol^2 + 0.27{*}Ranch2 + 0.99{*}Ranch3 + 0.15{*}Ranch4$	0.94	<0.001	Percent cover of exotic plants per m^2
Log (WetlandPlants)†	27	TI	0.77	225.4	$0.33 + 0.003{*}TI + 0.022{*}GDD$	0.59	<0.001	Percent cover of wetland plants per m^2
Fish†	46	TI	0.92	303.1	$6.12 + -0.3{*}TI + 1.58E\text{-}03{*}TI^2 + -2.33E\text{-}01{*}GDD + 1.09E\text{-}02{*}TI{:}GDD - 5.45E\text{-}05{*}TI^2{:}GDD$	2.53	<0.001	Fish count per m^2
Amphibians†	22	TI, DC	0.53	175.9	$-1.13–0.007{*}TI - 0.004{*}DC + 1.64{*}Ranch2 + 2.14{*}Ranch3 + 0.48{*}Ranch4$	1.06	0.007	Amphibian count per m^2
Macroinverts†	47	Maxdepth,	0.69	355.6	$-0.71 + 0.08{*}Maxdepth + -0.001{*}Maxdepth^2$	1.59	<0.001	Macroinvertebrate count per m^2
Log(Mosquito)†	9	*na*	0.16	213	$-0.21–0.09{*}Fish$	1.12	<0.001	Mosquito count per m^2

[a] N is the number of models assessed.
*Equations developed using data from Boughton et al. (2019) and are unique to this study.
†From Boughton et al. (2019).

were created by using the varying spillage levels from only two ranches (Ranch 2 and 3), they are likely to be representative of the hydrology and biodiversity for the ranchlands in the basin. The primary reason for the 13 proposals being representative of the basin, especially the northern part where most of the additional water storage is needed, is its low topographic relief. Florida is the flattest state in the United States (Dobson and Campbell, 2014) with the Everglades characterized by an extremely low slope of 3 cm per km (https://www.floridamuseum.ufl.edu/southflorida/regions/everglades/). Topographic homogeneity has been shown to reduce variability in water availability and biodiversity (Jobbágy et al., 1996). Therefore, we assume that 13 proposals created using 13 spillage levels modeled at Ranch 2 and 3, are likely to be typical of most of the basin.

2.8.2. Selecting water storage alternative

In addition to site selection, the DSS was also used to rank the plausible water storage alternatives at the two ranches (2 and 3). Multiple spillage levels are possible at a site. Each spillage level, referred to as the water storage alternative, would lead to a different response in biodiversity indicators due to changes in hydrologic variables used by the eco-hydrologic models. The DSS was used to rank the said alternatives for the selected site. Rankings were created and analyzed from two buyer perspectives – biodiversity response and water storage (Table 4). Seven alternatives were ranked for Ranch 2 and six for Ranch 3.

It is to be expected that uncertainty in hydrologic model predictions will propagate to biodiversity response predictions and subsequently to DSS-based decision making. The ditches at ranch outlets are hydraulically connected to the wetlands. Therefore, the assumption here is that MIKE-SHE/MIKE-11 model performance and errors in simulating the ditch levels are similar to those for the wetland water levels, the derived hydrologic signatures, and the predicted biodiversity measures used within the DSS to rank the sites and the water management alternatives. Like all models, MIKE-SHE model's performance is likely to vary by site. In some cases, errors in predicting hydrologic signatures may be considerably different than the errors in predicting the ditch water levels. However, for the purposes of the DSS, these errors were considered acceptable because the primary use of DSS is ranking PES proposals and water retention alternatives and not deciding payments for the service.

2.8.3. Use of DSS for Multi-Party buyers

Typical buyers in PES programs are government agencies and in the case of multiple services, such as the one presented here, buyers are likely to be a coalition of different government agencies and non-governmental organizations. We present an example where the DSS could be utilized in quantifying trade-offs and making an informed decision in a situation where three agencies with varied preferences form a coalition to buy services of water storage and biodiversity enhancements.

3. Results and discussion

3.1. Biodiversity response to water storage

Significant relationships were observed between specific ecological and hydrological variables for all biodiversity indicators except mosquito abundance (Table 5). Forage increased with time inundated (TI) quadratically (Table 5), indicating that there is a threshold TI value at which cover of forage plants peaks. This may be because flooding intolerant (e.g., *Paspalum notatum*) and flooding tolerant (e.g., *Panicum hemitomon*) forage species overlap at intermediate water availability level. As TI increases, forage also increases (an economic benefit for rancher) until a peak length of TI is reached, after which, forage growth starts to decline (an economic disadvantage) due to flooding stress and a shift to emergent macrophyte species that are poor forage for cattle. This suggests that a spillage level that provides maximum forage may not necessarily be the highest spillage level.

Exotic invasive plant cover, a dis-service, was found to have a positive quadratic relationship with storage volume (p-value < 0.001; Table 5) indicating that increasing the wetland storage would lead to an increase in the exotic plant cover and a decrease in native plant biodiversity. There are many invasive plants adapted to long hydroperiods in the Everglades region.

Wetland plant cover was positively corelated with time inundated and growing degree days (Table 5). This finding is consistent with other studies related to wetland plants (Foti et al., 2012; Sonnier et al., 2018). There is a maximum wetland depth (Maxdepth) at which macroinvertebrates are most abundant. Fish abundance decreased with longer delays between flooding periods and with later dates (autumn months) in the growing season (Table 5), similar to those observed by Gatto and Trexler (2019). Amphibian (frog) abundance was shown to decrease with TI (Table 5), which is consistent with the well-known phenomenon of rapid reproduction and tadpole development with initial flooding, and the negative impacts of increased wetland connectivity (DC; Table 3), that provides access for fish to prey on tadpoles. Mosquito abundance decreased with increase in fish abundance, likely because fish are predators of mosquitos (Table 5). Invertebrates and vertebrates (fish, macroinvertebrates, amphibians, and mosquitos) are important food sources for species of higher trophic levels, including birds and mammals and an increase in their abundance could sustain these other animals known to be abundant on the ranches (Swain et al., 2013). This benefit was not measured in this study, as the scale and number of wetlands studied were insufficient for adequate sampling, but it is reasonable to assume this co-benefit when biodiversity is included in the water storage PES program.

Models for two biodiversity indicators, exotic plant cover and amphibian abundance, included ranch effect variables, indicating that ranch management affects some of the biodiversity responses. The ranches used in the development of the eco-hydrological models were identified as either improved pasture, semi-native pasture, or a mixture of the two (Table 1). With the land use history and addition of fertilizer to the improved pastures, the wetlands in improved pastures tend to be more nutrient rich while wetlands in semi-native pastures are more oligotrophic (Boughton et al., 2019). All biodiversity indicators responded to changes in hydrology and could be predicted directly using hydrologic variables or indirectly (mosquitos) using ecological variables (fish) that were affected by hydrology. These models enabled the use of biodiversity in evaluating PES proposals.

3.2. Decision support system

Depending on the storage goals for the basin or sub-watershed, the state's affordability criteria and willingness to pay, and a rancher's tolerance of extended flooding in relation to forage production, participating ranches can provide varying degrees of water storage. Eco-hydrological models showed that variations in water storage and associated hydrological variables directly impact the biodiversity services from wetlands on the participating ranches (Table 5). In a review of other PES studies aiming to conserve biodiversity, Bullock et al. (2011) suggested that instead of accounting biodiversity benefits as a service, they be considered an inherent response to water management and included in designing the PES programs. The DSS presented here is based on the same line of thought. We recommend biodiversity benefits be used to select spillage levels as well as prioritize PES sites (ranches) given the limited funds available for the program.

3.2.1. Selecting the water storage sites

One of the founding principles of PES program is voluntary participation. Institution(s) paying for the service(s) have the right to determine the eligibility of the participants given the buyer-financed nature of the program. The participants (sellers) can accept or decline to provide the services. Like any other market-like program, PES programs also aim for maximizing cost efficiency and site selection has been shown to accomplish the same (Wünscher et al., 2008). Enhancement of "good" biodiversity was a "non-economic" but valued product of the PES program. We propose that it be used as a decision-making factor in selecting sites for implementation. However, in future, if biodiversity is bought by the state in lieu of financial compensation to the ranchers: 1) the eco-hydrological models could still be used to quantify the biodiversity services and associated trade-offs, and 2) the DSS could be used to prioritize the participating sites solely on the basis on the level of the biodiversity service. The DSS was used to rank multiple sites and prioritize them from biodiversity and water storage standpoints. Biodiversity is one of the most prominent indicators of ecological health of a system and paying for biodiversity services has recently gained momentum (Herzon et al., 2018). For example, in the Peninsular Florida ecoregion, state (e.g., Florida Forever) and federal (e.g., Everglades Headwaters National Wildlife Refuge) programs target land acquisition and easements for the protection of 88 federal and state listed species of greatest conservation need, many of which are wetland dependent (e.g., whooping crane, Everglades snail kite, wood stork). Agricultural expansion is the leading cause of biodiversity loss worldwide and therefore, the focus needs to be on agricultural practices to maintain a biodiverse environment. In theory, service buyers only need to pay for what they receive under a PES scheme. Any additional benefit is simply an outcome of design and implementation. In cases such as the payment for water storage program discussed here, where the additional benefit is biodiversity, a common dilemma faced by scientists and environmental managers alike is the basis for biodiversity services payment – Are actions enough or should the payments be based on results (Gibbons et al., 2011)?

The DSS takes an in-between approach by designating the best possible ecological outcome as the basis for PES fund allocation. The DSS bundles the biodiversity services with water storage through eco-hydrological models and assists in prioritizing the PES proposals based on their impact on biodiversity and importance weights assigned to various biodiversity indicators. The weights are decided by the user based on their interest. For example, if the user is a buyer that is an agency dedicated to native wildlife protection, their interests might lie in increased fish, and macroinvertebrate abundance and they would assign higher weights to the said biodiversity indicators. The DSS enables the ranking of PES proposals and allows the environmental managers to design and implement the payment for water storage services program such that the best possible ecological outcomes are an inherent component of the system.

Ranches in the Everglades and other landscapes are a combination of uplands and drained wetlands. Using agricultural lands to store excess on-site rain or off-site public water would lead to increased wetland hydration, enhanced baseflows and reduced peak surface flows to move towards restoring the hydrology of the Everglades and the two linked estuaries (Fig. 1). However, it does not necessarily guarantee the re-establishment of pre-existing ecological (biodiversity) health of the system given the complex interlinked nature of hydrology and ecology, and the fact that the basin still has a large network of regional canals and flood control structures essential to avoid flooding of agricultural and urban lands. The current "Business as Usual" approach to the NE-PES site selection is to choose a site that maximizes water storage in the basin making the best approach to rank the PES proposals based solely on the total storage offered by the landowners. However, such a mechanism defies the hydrological and ecological benefits of the "dispersed water management" approach.

Thirteen example PES proposals (sites) were compared using the DSS and assigned a priority ranking based on the predicted biodiversity and water storage responses. The rankings calculated using the three methods (WAM, CP, and PROMETHEE) were mostly in agreement. Two of the three methods consistently agreed for all weighting schemes; WAM consistently agreed with one or both of the remaining two, resulting in a more robust decision. For selection of sites from a

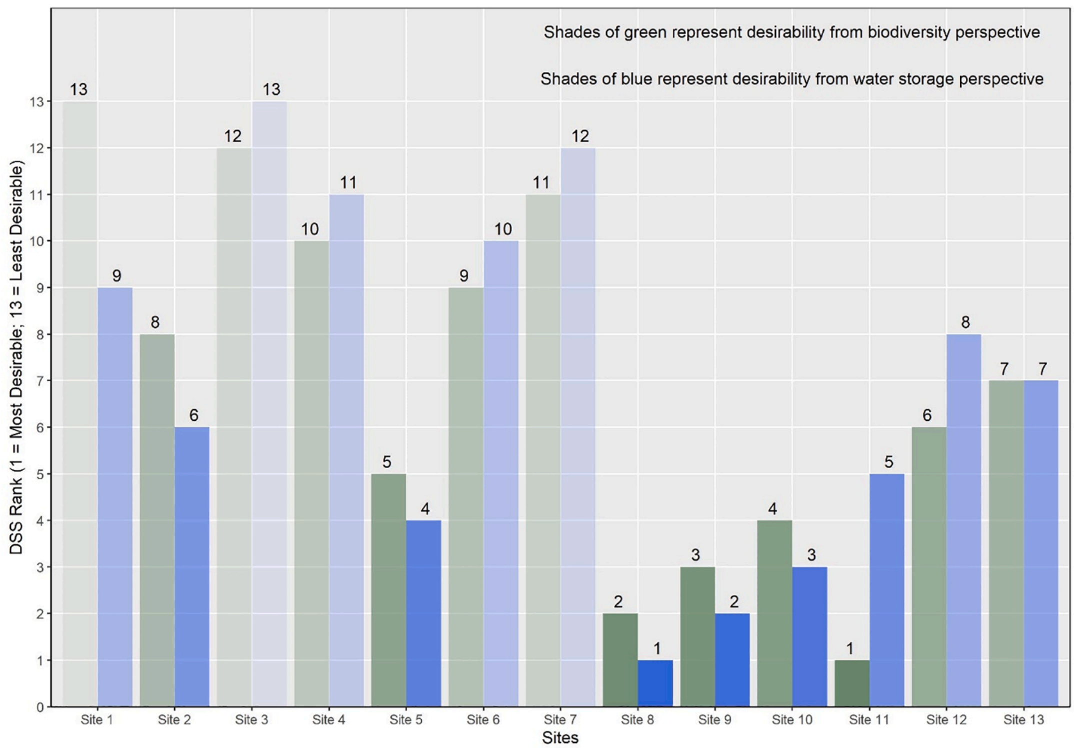

Fig. 3. The DSS ranking for 13 example proposals (sites) using the weighted average method (WAM).

biodiversity perspective, "desirable" biodiversity indicators (e.g., fish and wetland plant cover) were assigned the highest possible weight of 4 while the "undesirable" indicators (e.g., mosquito abundance) were assigned the lowest possible weight of 1 (Table 4). All three weighting techniques utilized by the DSS assigned the highest score and best rank (1) to Site 11 (Fig. 3) which coincided with a moderate increase in all biodiversity services and minimum dis-services. Given the weighing scheme, it is understandable that the option with a moderate increase in all biodiversity services would be chosen. For example, a site expected to provide maximum fish, but minimum amphibian abundance would be ranked lower than the one that would provide a moderate increase in both. Jessop et al. (2015) also showed that all biodiversity services do not maximize at a single level of hydrologic restoration. Since all "desirable" and "undesirable" biodiversity indicators were assigned equal weights of importance, there was no opportunity for compensatory trade-offs when selecting the most desirable site. However, buyers and stakeholders could assign different weights if they wished to analyze the trade-offs (e.g., increase in fish abundance while losing amphibians) and come to a consensus for ranking the sites.

Sites were not always ranked the same from the biodiversity and water storage perspectives (Fig. 3) indicating the need for a DSS to evaluate the trade-offs and help stakeholders make an informed decision. For example, Site 11 was ranked the best from the biodiversity standpoint, but it was ranked only fifth most desirable from the water storage perspective among the 13 example sites (Fig. 3). The MIKE-SHE/MIKE-11 model results showed that Site 8 (ranked the best) would lead to an 84% reduction in surface flows (compared to the baseline) followed by Site 9 (69%) which was ranked second most desirable from water storage standpoint. If the maximum weight i.e., 4 is assigned to water storage, the ranking simply follows the order of surface outflow reduction values. Although desirable, using biodiversity for ranking is inherently complex since biodiversity is a combination of multiple ecologic variables.

Site 8 provided maximum increase in fish abundance and wetland plant cover however, it was ranked second to Site 11 which was the most desirable because it provided higher than average increase in all "desirable" biodiversity indicators and the same level of increase in "undesirable" indicators as Site 8. With all biodiversity indicators being assigned equal weights, Site 8 was a less favorable choice from the biodiversity standpoint. If fish abundance and wetland plant cover were the only indicators chosen to quantify biodiversity response, Site 8 would have been ranked the best.

If water storage were the only criteria for awarding PES contracts, biodiversity enhancement would not be a guaranteed service. Essentially, buyers may have to settle for a lower amount of targeted service (water storage) to maximize biodiversity. Ecosystem service trade-offs have been suggested as an inevitable outcome when targeting more than one service through a program (Swain et al., 2013; Goldstein et al, 2012). Although sites were compared assuming equal importance would be given to all biodiversity indicators, in a case where there are multi-party buyers with different interest/preference (e.g., producers, state environmental agencies and/or NGOs, urban developers, tourism) the weights can be assigned accordingly, which is another use of the DSS discussed in section 3.2.3.

3.2.2. Selecting the level of service

The first use of the DSS is to rank PES proposals where the sellers (landowner/rancher) have already quantified the water storage they intend to offer i.e., their preferred spillage level at the discharge structures. However, it is possible that there exists a water storage alternative different from that proposed by the seller which the buyer wants because it can result in a better biodiversity response compared to what the seller is offering. The DSS could also be utilized to achieve the most desirable water storage level from biodiversity standpoint for each eligible participating site. If the most desirable water storage level from a biodiversity perspective is different from proposed, it would create an opportunity for negotiation – a compensatory trade-off between financial incentive and water storage. Eligibility versus the desirability of participants, in cases where multiple environmental services are bundled, has been discussed (Banerjee et al., 2013; Zanella et al., 2014). For example, Banerjee et al. (2013) discussed bundling floodplain restoration with additional services like habitat restoration. In our case,

Table 6
The DSS rankings based on biodiversity and water storage value for alternatives plausible at Ranch 2 using the weighted average method (WAM). Rank 7 represents the worst and 1 represents the best alternative in this example. The numbers in parentheses after the rank are the DSS scores for each water storage alternative. The numbers in superscript after DSS score are the ranks assigned by the DSS when the dis-services were excluded from the analyses. Dis-services include mosquito abundance and exotic plant cover.

Water Storage Alternative	Biodiversity Rank (DSS Score)	Water Storage Rank (DSS Score)
1 (Baseline)	7 (0.700) ^{7}	7 (0.461) ^{7}
2 (1 Board)	3 (0.704) ^{3}	3 (0.468) ^{3}
3 (2 Boards)	5 (0.702) ^{5}	4 (0.466) ^{4}
4 (3 Boards)	4 (0.704) ^{4}	5 (0.465) ^{5}
5 (4 Boards)	6 (0.702) ^{6}	6 (0.464) ^{6}
6 (7 Boards)	2 (0.738) ^{2}	2 (0.508) ^{2}
7 (8 Boards)	1 (1.379) ^{1}	1 (0.855) ^{1}

Table 7
The DSS rankings based on biodiversity and water storage value for alternatives plausible at Ranch 3 using the weighted average method (WAM). Rank 6 represents the worst and 1 represents the best alternative in this example. The numbers in parentheses after the rank are the DSS scores for each water storage alternative. The numbers in superscript after DSS score are the ranks assigned by the DSS when the dis-services were excluded from the analyses. Dis-services include mosquito abundance and exotic plant cover.

Water Storage Alternative	Biodiversity Rank (DSS Score)	Water Storage Rank (DSS Score)
1 (1 Board)	6 (0.503) ^{6}	4 (0.235) ^{3}
2 (2 Boards)	5 (0.535) ^{5}	5 (0.182) ^{5}
3 (3 Boards)	4 (0.583) ^{4}	6 (0.173) ^{6}
4 (4 Boards)	3 (0.608) ^{3}	3 (0.285) ^{4}
5 (5 Boards)	2 (0.624) ^{2}	2 (0.378) ^{2}
6 (6 Boards)	1 (0.635) ^{1}	1 (0.433) ^{1}

all ranches with varied spillage levels would be eligible to participate in the PES program however, the level of biodiversity services provided will be a factor in selecting the proposals. The DSS would fulfill the need of ranking in order of desirability, the available water storage (spillage) alternatives, based on trade-offs among water storage and biodiversity provisioning thus providing the buyer a holistic assessment of alternatives, essential for decision making.

After the DSS was used to select sites, the next step was to select the most desirable water storage service level by ranking a range of water storage (spillage) alternatives at each site. As an example, consider that Ranch 2 has been selected based on biodiversity response, using the DSS. There are seven different water storage alternatives plausible at Ranch 2 (0 (baseline), 1, 2, 3, 4, 7, and 8 boards; Table 6).

The biodiversity DSS scores, after assigning the highest weight (4) to all biodiversity service indicators and the lowest weight (1) to all biodiversity dis-service indicators (Table 4), ranged between 0.700 and 1.379 (Table 6). We have included two ranks based on whether or not the dis-services are considered (Table 6). The dis-services are ignored by most PES programs (Boughton et al., 2019). Results showed that there was little difference in the DSS rankings when excluding the dis-services from the analyses indicating that mosquito abundance and exotic plant cover had limited effect on the score and therefore ranking. This was to be expected because mosquito abundance did not correlate with any hydrologic variable therefore, any difference in the rankings with and without accounting for dis-services were mainly due to eliminating exotic plant cover and changes in other services.

An example of how DSS can facilitate negotiations between buyers and sellers follows. Assume that the seller (landowner) is offering Alternative 6 (7 boards: Biodiversity-focus DSS score = 0.738; Water storage-focus DSS score = 0.508). However, Alternative 7 (8 boards) was ranked highest from both biodiversity response and water storage perspectives (Biodiversity-focus DSS score = 1.379; Water storage-focus DSS score = 0.855). The hydrological model predicted a considerable difference in water storage service between Alternatives 6 and 7; Alternative 7 increased the storage by 70% in contrast to Alternative 6 which only provided 17% increase in storage compared to baseline (no structure at the ranch outlet). The DSS also illustrates the financial implications for the buyer and the seller as well as outcomes. For the participating ranches in the water storage PES, the payments ranged between $98 and $158 per acre-ft of water stored (https://fl.audubon.org/news/sfwmd-approves-eight-water-storage-projects-northern-everglades). The average payment was $134 per acre-ft.($16.5/ha-m) which was used in the DSS. The annual financial incentive for the seller (landowner) would be 31% higher in the case of Alternative 7 ($8,505/year) compared to Alternative 6 ($5,851/year) from a 476-ha site. In such cases where there is a disagreement between the water storage alternative offered by the seller and the most desirable one from biodiversity or water storage standpoint which the buyer wants, a negotiation between the seller and the buyer to come to a mutually agreeable solution will be required. One solution can be that the buyer absorbs the transaction costs to implement the PES in lieu of the seller implementing Alternative 7 which will result in increased economic losses due to flooding of additional pasture area. Transaction costs commonly comprise of negotiation, monitoring, and enforcement costs (Alston et al., 2013). For example, if the buyer (state) could absorb the cost of engineering/hydrologic consultants whom the seller hired to provide storage estimates to prepare the PES application to the buyer (state), then the seller might be more amenable to implementing the desired board height (Alternative 7). Furthermore, the cost of data collection and field-verification of complex hydrologic models such as MIKE-SHE/MIKE-11 are high and sharing or absorbing a part (or whole) of the expense by the state could be beneficial for the seller. Alternatively, the unit price of water storage service could be negotiated based on the level of biodiversity services; a site which provides additional biodiversity services may get a higher price for the water storage compared to a site which provides the same water storage but lower biodiversity service.

Given the sandy soils with a hydraulic conductivity of up to 216 cm per hour (Obreza and Collins, 2008) and shallow water table environment in south Florida, the term "water retention/storage" needs to be used with caution (Wu, 2014; Shukla et al., 2015). A considerable part of surface water retained due to discharge control structures, such as CRB structures in our study, can leave the system through subsurface pathways (Wu, 2014; Shukla et al., 2015; Shukla et al., unpublished data). Therefore, increase in spillage level (number of boards) at the outlet structure does not always translate to increase in water retention on the ranch. Holding more water on the ranch leads to a change in hydraulic gradient leading to surface water leaving the ranch through subsurface pathways outside of the ranch. For example, the MIKE-SHE/MIKE-11 model predicted negligible change in storage when the number of boards was raised from 1 (Alternative 2) to 4 (Alternative 5) at Ranch 2. The non-linear relationship between number of boards and surface storage indicated that increase in board height does not always result in increased storage and was demonstrated when the DSS ranked Alternative 4 (3 boards) (DSS score = 0.704; Table 6) marginally higher than Alternative 5 (4 boards) (DSS score = 0.702; Table 6) when weights were assigned from the perspective of increasing water storage and decreasing implementation cost (Table 4).

Results from the DSS can be used to facilitate negotiations among buyers and sellers. We present another example from Ranch 3 (Table 7). Six water storage alternatives (boards 1 through 6) were considered for selecting the most desirable storage level. Alternative 6 (6 boards) was ranked the highest from the biodiversity response perspective (Table 7) since it led to maximum fish, amphibian, and wetland plant abundance. The said biodiversity service response came at the cost of the lowest abundance of macroinvertebrates and an increased loss in forage. The latter is undesirable for the seller because the rancher has to buy additional cattle feed to compensate for the loss of forage or reduce cattle stocking density. These scenarios will require negotiation between the seller (landowner/rancher) and the buyer. For example, the buyer may agree to absorb the cost of forage loss for the seller by increasing the payment for water storage services. Alternative 6 was also ranked the highest from the water storage standpoint (Table 7). In this example, a buyer whose interest lies in biodiversity enhancement or a buyer whose interest is only water storage can agree to implement Alternative 6. However, this may not always occur and in most cases, the buyers of environmental services may be a coalition of an array of stakeholders with varied interests (Thompson and Friess, 2019) and trade-offs are often a necessity in negotiating agreements. We explain, in the following section, the use of the DSS in quantifying the trade-offs and coming to a consensus about site selection and service provision levels when multiple buyers with different preferences are involved.

3.2.3. Decision making for Multi-Party buyers

Environmental governance is a result of convergence of stakeholders from different backgrounds. Often, when acquiring environmental services, different constituents of environmental governance collaborate. The first step to make such inter-organizational collaborations successful is for the stakeholders to reach a consensus on which water storage alternative to buy. The DSS can help in such situations by incorporating the stakeholder preferences when ranking the available alternatives. Our DSS would provide a choice to all involved stakeholders to prioritize their preferences. To put this application into perspective, we provide a mock scenario where the buyer is a coalition of three environmental agencies A, B, and C with focus on water (e.g., Water Management Districts in FL), vegetation, and wildlife management, respectively. Agency A's objective would be to maximize water storage. The expenditure on exotic vegetation management in Florida ranges between $97 million and $127 million per year therefore Agency B's interest would be to maximize native vegetation while minimizing exotics. Exotic animal population in the Everglades basin is a growing concern (SFWMD, 2021c). It follows that Agency C's aim would be to maximize native fauna. The three stakeholders would assign different weights to the DSS parameters i.e., the biodiversity indicators, water storage payment, and cost of implementation, depending on their relative importance to them. The DSS would enable the analyses of synergies and trade-offs thereby aiding in reaching a consensus among buying coalition members with different interests which can otherwise be in conflict with each other. The onus would ultimately rest with the agencies to decide the acceptable solution, but the DSS would facilitate the discussion by providing metrics necessary to evaluate each water management alternative.

4. Conclusion

The PES proponents, given the complexity in its quantification, cautiously regard biodiversity as a standalone ecosystem service. This is especially true in our study where biodiversity was an unaccounted consequence of increasing water storage on private ranchlands (upland-wetland systems) subsequently resulting in wetland hydration and changes in "desirable" and "undesirable" biodiversity indicators. Instead of considering biodiversity as a direct service from ranchlands, we propose it to be used as a decision-making criterion for prioritizing PES sites and identifying optimum level of hydration. Eco-hydrological models were developed to quantify the effects of water storage on biodiversity. A DSS, utilizing the said models with a user-specified weighing scheme, was developed to analyze the trade-offs between water storage and biodiversity indicators for a variety of water storage alternatives. Contrary to the common belief, results showed that increase in wetland hydration level does not necessarily result in desirable biodiversity changes. Undesirable biodiversity indicators (e.g., exotic plants) also increase with enhanced water storage therefore the trade-offs between storage, and desirable and undesirable biodiversity indicators need to be quantified to make an informed decision.

Most PES programs are managed by layers of governmental and environmentally focused non-profit institutions. One of the contributions of our study was to develop a DSS tool which could be utilized by stakeholders with a spectrum of preferences (water, flora, fauna) and evaluate how their interests would be impacted as the proclivities of others in the coalition change. Above all, the DSS would provide a platform for all involved parties, sellers and buyers alike, to negotiate based on field-verified model predictions of storage and biodiversity indicators.

Combining hydrology and biodiversity for a PES is challenging because of inherent uncertainties and requires collaboration amongst physical, biological and social scientists and other stakeholders. The proposed DSS is only the first step in defining hydro-ecological linkages, quantifying how one impacts the other, and how multiple services and dis-services can be combined for a real-world PES program. The possibilities to expand the use of the DSS are many. For example, depending on the proximity to critical species habitat, new criteria could be added and assigned different weights. With an emphasis on rural prosperity, another factor that could be added is the socioeconomic equity; small ranches (e.g., 500 acres) could be given a higher weight to ensure their inclusion among the selected sites. Other factors could be proximity to critical groundwater recharge zones within the basin given the increased groundwater recharge benefit of water retention and increased carbon sequestration due to wetland hydration. Furthermore, if biodiversity is approved as a standalone ecosystem service with willing buyers, the DSS could be utilized to select desirable participants based on the existing level of water storage on their ranchlands. With simultaneous emphasis on mitigating and adapting to climate change, designing future PES programs are likely to become more complex due to the emerging need to consider more services and trade-offs. Although our study considers only a small subset of ecosystem services, it does provide a preview of how complex future PES designs are going to be.

Declaration of Competing Interest

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

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Appendices

Appendix A
Table A1

Table A1
Ecological species and their grouping for the development of eco-hydrological models.

Plant group species used for eco-hydrological models		
Forage Plants Group		
Species Name	Wetland Status	Origin
Andropogon glomeratus var. pumilus	FACW	native
Andropogon virginicus var. glaucus	FACU	native
Andropogon virginicus var. virginicus	FAC	native
Andropogon sp.	FACW	native
Andropogon virginicus		native
Aristida palustris	OBL	native
Aristida patula	FAC	native
Axonopus fissifolius	FACW	native
Axonopus furcatus	OBL	native
Cynodon dactylon	FACU	exotic
Dichanthelium erectifolium	OBL	native
Digitaria serotina	FAC	native
Echinochloa walteri	OBL	native
Hemarthria altissima	FACW	exotic
Hymenachne amplexicaulis	OBL	exotic
Panicum dichotomum	FACW	native
Panicum hemitomon	OBL	native
Panicum longifolium	FACW	native
Panicum repens	FACW	exotic
Panicum rigidulum	FACW	native
Panicum sp.	FACW	native
Paspalidium geminatum	OBL	native
Paspalum acuminatum	OBL	exotic
Paspalum conjugatum	FAC	native
Paspalum distichum	OBL	native
Paspalum notatum	FACU	exotic
Paspalum urvillei	FAC	exotic
Sacciolepis striata	OBL	native
Wetland Plants Group		
Species Name	**Wetland Status**	**Origin**
Pontederia cordata	OBL	native
Sagittaria graminea	OBL	native
Sagittaria lancifolia	OBL	native
Thalia geniculata	OBL	native
Exotic Plants Group		
Species Name	**Wetland Status**	**Origin**
Alternanthera philoxeroides	OBL	exotic
Cuphea carthagenensis	FACW	exotic
Desmodium incanum		exotic
Eichhornia crassipes	OBL	exotic
Eragrostis atrovirens	FAC	exotic
Ludwigia peruviana	OBL	exotic
Momardica charantia	FAC	exotic
Paspalum acuminatum	OBL	exotic
Salvinia minima	OBL	exotic
Solanum viarum		exotic
Urena lobata	FACU	exotic

Animal group species used for eco-hydrological models
Fish Species
Species Name
Elassoma sp.
Erimyzon succeta
Fundulus chrysotus
Gambusia holbrooki
Heterandia formosa
Heterandria formosa
Hoplosternum littorale
Jordanella floridae
Lepomis gulosus
Poecilia latipinna
unidentified fish
Frog Species
Species Name
Acris gryllus
Anaxyrus quercicus
Bufo terrestris
Gastrophryne carolinensis
Hyla cinerea
Hyla femoralis
Hyla squirella
Lithobates grylio
Lithobates sphenocephalus
Pseudacris ocularis
Macroinvertebrate Species
Species Name
Anax junius
caddis fly
Cybister fimbriolatus
damselfly
dobsonfly
dragonfly
fly
hemipteran bug
hemipteran bug
hydrophilid beetle
leech
Lethocerus uhleri
mayfly
Naucoridae
Notonectidae
Procambarus sp.
Pseudobranchus sp.
Ranatra sp.
small coleopteran species
small dytiscid beetle
small hydrophilid beetle
tipulid fly
Tramea carolina
unidentified beetle
unidentified insect
Mosquito Species
Species Name
Aedes sticticus
Anopheles bradleyi
Anopheles crucians
Anopheles sp
Anopheles walkeri
Culex declarator
Culex erraticus
Culex nigripalpus
Culex quinquefuscratus
Culex sp
Culex territans
larva
Mansonia tittilans
Psorophora ciliata
Psorophora columbiae
pupa
Uranotaenia sapphirina

Wetland status OBL is obligate wetland, FACW is facultative wetland, FAC is facultative, and FACU is facultative upland.

Appendix B

Table B1

Table B1
Sheets in the spreadsheet based DSS (Microsoft Excel) and their description and user navigability. MCDA = Multicriteria Decision Analysis; WAM = Weighted Average Method; CP = Compromise Programming method; PROMETHEE = Preference Ranking Organization Method for Enrichment Evaluation.

Excel Sheet	Description	Navigation Buttons
Introduction	Introduces the user to the DSS	Wetland Data, MCDA Matrix, Stakeholder Interface
Wetland Data	User enters all pertinent hydrological information for the alternatives to be analyzed	Introduction, Stakeholder Interface, Metadata
Eco-hydrological Estimates	Eco-hydrological models used to estimate biodiversity responses	None
MCDA Matrix	Displays the results of the eco-hydrological model estimations for each water storage alternative	Introduction, Stakeholder Interface
Stakeholder Interface	User defines importance weights for each criterion and the ranking method to use, views scores and ranks of alternatives (tables and graphs)	Introduction, MCDA Matrix, Calculate Results, Ranking Results
Ranking Results	Displays the results for three weighting schemes for a given ranking method in a tabular form	Introduction, Stakeholder Interface, MCDA Matrix, Clear Results
Matrix	Used to transform the MCDA Matrix that has alternatives in columns and the criteria in rows	–
WAM, CP, PROMETHEE	Ranking calculations are performed for the specific ranking methods	–
Metadata	Provides description of the hydrological variables used in the DSS	–

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