ABSTRACT. A two-year study evaluated the nitrogen (N) fluxes, processing, and treatment efficiency (TE) of a 14.85 ha stormwater detention area (SDA) receiving drainage from a vegetable farm in subtropical Florida. The TE was 62% and 89% in years 1 and 2, respectively. Seepage N losses are often ignored in estimating stormwater treatment. Approximately 11% and 20% of the incoming N left the SDA through seepage, reducing the TE to 51% and 67% in years 1 and 2, respectively, indicating the importance of subsurface N losses for downstream water quality. Rainfall variability controlled the timing and volume of the inflow drainage and surface water levels inside the SDA. Variable water levels controlled the aerobic and anoxic conditions inside the SDA, thus controlling the N processing and treatment. Coupled nitrification-denitrification, as a result of frequent wetting-drying cycles, was the main N treatment pathway during year 1. Drought conditions in year 2 led to 89% less surface outflow compared to year 1, resulting in water volume retention being the main process for N retention. The N TE could be increased from 68% to 86% if about two-thirds (63%) of aboveground biomass in the SDA area is harvested annually during the dry season. A payment for environmental services (PES) framework, with the state as buyer and the SDA owner as seller of N treatment services, was evaluated with a 20-year net present worth (NPW) of biomass harvesting for enhanced N treatment. The economic analysis included the benefit from composting the harvested biomass for on-farm use. A positive NPW ($835,000) indicated the economic feasibility of the project, predicting an annual benefit of $42,000 year⁻¹ for the 112 ha farm. Scale-up of the PES approach can offer additional N treatment and C sequestration services as well as increased farm productivity.

Keywords. Best management practices, Impoundments, Nutrient treatment, Payment for environmental services, Phytoremediation, Stormwater.
poses an urgent need for improved agricultural N management practices. Federal and state agencies have developed an extensive list of best management practices (BMPs), which include on-farm N management as well as edge-of-farm N treatment. On-farm management practices such as irrigation scheduling and fertilizer management are targeted at reducing nutrient leaching losses, while edge-of-farm practices are aimed at retaining nutrients lost from the farm. Edge-of-farm practices play a vital role because of the inevitable loss of N from croplands despite the best on-farm nutrient management (Shukla et al., 2010). For example, Hendricks and Shukla (2011) showed an elevated nitrate/nitrite-N concentration in leachate (76 mg L⁻¹) even after using the recommended fertilizer application rate for a vegetable farm in south Florida. On-farm practices have been evaluated for their effectiveness; however, limited data exist on nutrient treatment at the edge-of-farm, which is the last point of treatment before farm drainage enters a freshwater body.

This article focuses on edge-of-farm N treatment by agricultural stormwater detention areas (SDAs), which are assumed to reduce nutrient loads by targeting both water volume and nutrient concentrations (Shukla et al., 2010). Traditionally, SDAs were constructed as a flood protection measure in Florida and elsewhere, but they have now become a state-regulated, important water quality BMP. Commonly known as farm ponds or aboveground impoundments, SDAs represent almost 6% of the total farmed area globally (Downing et al., 2006). Despite being a significant feature and a priority among other BMPs, SDAs are not sufficiently inventoried (SFWMD, 2009). Furthermore, unlike most on-farm BMPs, constructing SDAs is expensive. The investment required to build an SDA ranges from a few hundred thousand dollars to more than a million dollars, depending on a variety of factors such as the area, number of inflow pumps, length of embankment, etc. (Shukla, 2014). Due to limited data on the nutrient treatment potential of SDAs, water management and other environmental agencies rely on results from well-studied detention/retention systems, such as constructed wetlands and urban stormwater detention areas. The N treatment processes of sedimentation, soil adsorption, plant uptake, and microbial transformation (e.g., denitrification) in other detention/retention systems are believed to be similar to those in SDAs; however, their extent differs given the differences in inflow type (pumped versus gravity fed in most constructed wetlands), hydraulic loading rate (significantly less compared to urban detention areas), and climatic conditions.

The irony of the situation is that SDAs are one of the most expensive BMPs but have limited evidence to prove their efficacy. There is a need to not only evaluate the N treatment efficiency of SDAs but also assess whether they are a cost-effective measure for treating agricultural drainage. Furthermore, it is important to find ways to enhance the N treatment efficiency of SDAs, especially in light of long-term nutrient inputs as well as future changes in climate. This article aims to quantify the N fluxes and N treatment efficiency of an SDA located in a vegetable farm in south Florida and identify cost-effective measures to enhance the N treatment efficiency of the SDA.

**METHODS AND MATERIALS**

**STUDY SITE**

The SDA evaluated in this study is located at the edge of a vegetable farm in the C-139 basin, a sub-watershed of the Everglades basin in south Florida (fig. 1). The SDA is 14.85
ha in area, was built to store the first 2.5 cm of surface water drainage from 112 ha of the farm, and has been in operation since 2000. The SDA was monitored for water and N fluxes for a two-year period (20 July 2009 to 20 July 2011).

The SDA structure consists of an embankment built from soil excavated from ditches on both sides. A borrow ditch surrounds the SDA on the outer perimeter and serves as a collector for the farm-scale drainage. After a rainfall event, the shallow water table rises close to the surface. To avoid flooding damage to the crops, the fields are drained through a network of drainage ditches. The drainage is conveyed to the outer borrow ditch from where it is pumped into the SDA by three diesel-powered axial-flow pumps (throw-out pumps), each designed for a maximum flow rate of 37.85 m³ min⁻¹. Pumps 1 and 2 are located on the western side of the SDA, and pump 3 is located on the east side (fig. 1). The outer borrow ditch on the western side receives farm drainage from the seepage-irrigated part of the farm, while the ditch on the eastern side receives drainage from the drip-irrigated part (fig. 1). The discharge site is located at the southern end of the SDA (fig. 1), where two corrugated aluminum culverts (1.22 m diameter) fitted with a sharp-crested rectangular weir are used as the discharge structure. The elevation of the weir is set at 5.52 m AMSL, NAVD 88 (i.e., height above mean sea level per the North American Vertical Datum established in 1991). Another borrow ditch is located around the interior boundary of the SDA. Over time, the inner borrow ditch gets partially filled and becomes discontinuous. Excluding the inner borrow ditch, the SDA is nearly level ground (5.49 m elevation) except for four depressions, three of which are jurisdictional wetlands. The discharge from the SDA is eventually released to the Everglades through two other SDAs and a stormwater treatment area (STA). The SDA soils are comprised of Myakka fine sand (sandy, siliceous, hyperthermic Aeric Haplaquods) and Basinger fine sand (siliceous, hyperthermic, Spodic Psammaquents). The SDA floor is covered by a variety of emergent vegetation types, including torpedograss ( Panicum repens), smartweed (Polygonum hydropiperoides), primrose willow ( Ludwigia peruviana), cattail ( Typha spp.), and Carolina willow ( Salix caroliniana). The wetlands inside the SDA and a part of the inner ditch on the eastern side are covered with water lettuce ( Pistia stratiotes).

**HYDROLOGIC MONITORING**

Surface water fluxes included pumped drainage, rainfall, evapotranspiration (ET), and discharge through the weir. The throw-out pumps at the site were locally manufactured and did not have any characteristic pump curves available. Therefore, all three pumps were calibrated for discharge rate as a function of impeller speed (rpm) and height of water above the impeller (h). Pump discharge for calibration purposes was measured using a propeller-type flowmeter with baffles installed at the end of the discharge pipe to ensure full pipe flow. The pumps were operated at different speeds to pump 10,000 gallons, and the average h was noted. Using this information, pumpage in gallons per minute (gpm) was estimated as a function of h and rpm. Variable h was measured using a vented pressure transducer (Campbell Scientific, Logan, Utah; accuracy ±1.5 mm, 5 min readings averaged every 15 min). The impeller speed (rpm) was measured using a proximity sensor (600 Series, Electro Sensors, Minnetonka, Minn.).

Weather parameters were measured at a weather station installed at the site for rainfall, relative humidity, air temperature, wind speed, and solar radiation. For rainfall, 5 s readings were summed to get 15 min totals, while for all other parameters, 5 s readings were averaged over a 15 min time span. The FAO-56 Penman-Monteith method (Allen et al., 1998) was used to estimate ET with the aforementioned weather parameters. Monthly crop coefficient (Kc) values were used, as reported for a wetland (87 km from the site) with similar vegetation (Wu and Shukla, 2013).

The discharge from the SDA was measured using an acoustic Doppler velocimeter (SonTek/YSI, San Diego, Calif.) installed at the end of one of the discharge culverts. Multiple instances of backflow from the adjacent SDA were observed and were considered as inflow while developing the water and N budgets. In addition to surface inflows and outflows, changes in surface storage were also estimated for developing the water and N budgets. A digital elevation model (DEM) was developed, and changes in surface storage were estimated using ArcGIS (ver. 10.1, ESRI, Redlands, Calif.).

Subsurface losses were estimated using the modified White method (White, 1932), as developed by Shukla et al. (2015). The modified method relies on two main assumptions: (1) nighttime ET in warm, subtropical south Florida is negligible only in the low-temperature season, and (2) there is no difference in seepage rates between day and night (Shukla et al., 2015).

**WATER QUALITY MONITORING**

Water quality samples at all inflow (pump) and discharge sites were collected using autosamplers (Teledyne ISCO, Lincoln, Neb.). A time-based discrete sampling strategy was used in year 1, while volume-based composite samples were collected in year 2. No significant differences were found in the N loads based on the sampling strategy (Shukla, 2014). Samples were analyzed for NH₄-N (ammonium), NOₓ-N (nitrate/nitrite), and TKN (total Kjeldahl nitrogen) at the Analytical Research Laboratory in Gainesville, Florida. Time series of nitrogen species concentrations were created using linear interpolation between two sampling points. Inflow and outflow nutrient loads were calculated using the flow volume time series in conjunction with nutrient concentration time series. For seepage load estimation, the 15 min time series of groundwater fluxes was multiplied with the 15 min time series of organic N.

**VEGETATION SAMPLING AND ANALYSIS**

An aerial survey of the SDA was conducted to identify major vegetation types and extent. A vegetation map of the SDA was prepared, and areas dominated by each vegetation type were calculated using ArcGIS (ver. 10.1, ESRI, Redlands, Calif.). Two replicates of all vegetation types identified during the survey were sampled for aboveground biomass in 1 m × 1 m quadrats during the dry season of 2010 and then dried, ground, and analyzed for TKN.
RESULTS AND DISCUSSION

WATER DYNAMICS

SDAs are designed to reduce both nutrient concentrations and flows; therefore, it is important to analyze the water dynamics in conjunction with the nutrient dynamics to better understand the nutrient load reduction processes inside SDAs. A significant difference (76%) was observed in pumped inflow between the two years because of the difference in magnitude and distribution of rainfall (fig. 2). Year 1 (2009-2010) received 25% more rainfall than year 2 (table 1). Furthermore, the January-May period of year 1 was categorized as the wettest spring in south Florida since 1932 (Shukla et al., 2017) after receiving 42% more rainfall than the historic average of 37.4 cm (Ali et al., 2000). The drainage requirements for a farm are a function of the magnitude and temporal distribution of rainfall. A bigger proportion of rainfall is drained from the farm during the dry season (November-May) compared to the wet season (June-October) because the dry season is the main growing season and the water table must be maintained at a low level to avoid damage to the crops in ground. Unusually high rainfall during the spring of year 1 resulted in the pumped inflow being almost four times that of year 2 (fig. 3). Most rainfall events during year 2 occurred during August-September 2010 and June-July 2011 (fig. 2); however, relatively dry conditions during the rest of the year led to reduced overall pumping in year 2 (fig. 2).

Differences between the pumped inflows led to significant differences in the surface outflows between years 1 and 2 (table 1). The surface outflow in year 2 (0.70 m) was 89% lower than in year 1 (6.64 m) due to the higher rainfall input and higher evapotranspiration losses in year 2.

![Figure 2. Daily rainfall, pumped inflow, and outflow at the SDA during the study period (July 2009 to July 2011). Negative flow represents backflow from the adjacent SDA located downstream (adapted from Shukla et al., 2017).]

Table 1. Water, total nitrogen (TN), and dissolved inorganic nitrogen (DIN) fluxes in year 1 (2009-2010) and year 2 (2010-2011).

<table>
<thead>
<tr>
<th>Component</th>
<th>Year 1</th>
<th>Year 2</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Water (m)</td>
<td>TN (kg)</td>
</tr>
<tr>
<td>Pump 1 inflow</td>
<td>7.73</td>
<td>3647.2</td>
</tr>
<tr>
<td>Pump 2 inflow</td>
<td>1.95</td>
<td>1453.4</td>
</tr>
<tr>
<td>Pump 3 inflow</td>
<td>1.66</td>
<td>755</td>
</tr>
<tr>
<td>Backflow into SDA</td>
<td>0.28</td>
<td>72.7</td>
</tr>
<tr>
<td>Rainfall</td>
<td>1.40</td>
<td>155.9</td>
</tr>
<tr>
<td>Dry atmospheric deposition(\delta)</td>
<td>13.02</td>
<td>6193.3</td>
</tr>
<tr>
<td>Total inflow(\beta)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Surface outflow</td>
<td>6.64</td>
<td>2359.1</td>
</tr>
<tr>
<td>Evapotranspiration</td>
<td>1.37</td>
<td>-</td>
</tr>
<tr>
<td>Seepage(\delta)</td>
<td>3.66</td>
<td>690.4</td>
</tr>
<tr>
<td>Change in surface storage</td>
<td>0.00</td>
<td>0.4</td>
</tr>
<tr>
<td>Surface retention(\delta)</td>
<td>6.38</td>
<td>3834.5</td>
</tr>
<tr>
<td>(excluding seepage losses)</td>
<td>49%</td>
<td>62%</td>
</tr>
<tr>
<td>Net retention efficiency(\delta)</td>
<td>21%</td>
<td>51%</td>
</tr>
</tbody>
</table>

\(\delta\) Dry atmospheric TN mass deposition = 0.7 \times TN deposition through rainfall (Poor et al., 2001).
\(\beta\) Total inflow = inflow from pumps 1, 2, and 3 + rainfall + dry atmospheric deposition + backflow.
\(\delta\) Seepage N loss estimated for the best-case scenario in which all DIN is treated and only dissolved organic N is lost from the SDA.
\(\delta\) Surface retention = total inflow – surface outflow; includes seepage and evapotranspiration losses.
\(\delta\) Surface retention efficiency = \([\text{total inflow} + \text{change in surface storage} - \text{surface outflow}] / (\text{total inflow} + \text{initial surface storage})\) \times 100.
\(\delta\) Net retention efficiency = \([\text{total inflow} + \text{change in surface storage} - \text{surface outflow} - \text{seepage loss}] / (\text{total inflow} + \text{initial surface storage})\) \times 100.
less than in year 1 (6.64 m). A similar effect of differential pumping was observed on subsurface losses, which were almost 50% less in year 2 compared to year 1 (table 1). These differences in water fluxes affected N fluxes and retention during years 1 and 2.

**NITROGEN DYNAMICS**

SDAs are designed to reduce drainage volumes originating from the farm as well as nutrient concentrations; therefore, concentration changes were analyzed in conjunction with loads to understand the N dynamics. A comparison between mean inflow and outflow concentrations of the N species showed the highest reduction in NO\textsubscript{x}-N concentrations in both year 1 (69%) and year 2 (91%; table 2), with the percent reduction in year 2 being numerically higher than in year 1. The main process for reducing NO\textsubscript{x}-N in detention/retention systems such as wetlands and reservoirs is denitrification (Saunders and Kalff, 2001). In addition to the incoming NO\textsubscript{x}-N through pumping and rainfall, nitrification of NH\textsubscript{4}-N (incoming as well as resident) was a result of ammonification (mineralization), a process by which organic N is biologically converted to NH\textsubscript{4}-N (Vymazal, 2007). During year 1, the SDA floor was inundated for a total of 202 days distributed throughout the year, compared to 64 days in year 2 (fig. 4). The inner borrow ditch, because of its low elevation, is a preferred pathway for the pumped drainage to reach the outflow location until it is filled. Due to its low average elevation (4.42 m) compared to the average groundwater table (5.36 m), the ditch intercepts groundwater and has water year-round, which facilitates denitrification. The inundation analysis combined with the higher mean hydraulic residence time in year 2 (30.1 h) compared to year 1 (11.3 h) shows that antecedent storage in the inner ditch and other connected depressions exceeded the pumped volume in year 2. This exceedance kept the drainage in the SDA, reducing the number of outflow events. Although the pumped drainage’s pathway to the discharge site was shorter, the higher residence time during year 2 led to higher denitrification and NO\textsubscript{x}-N treatment (91%; table 2) compared to year 1 (69%; table 2).


during both years, organic N passed through the SDA without significant change (year 1 = 19%, year 2 = 15%; table 2). A similar observation was made by Shukla et al. (2011) for drainage ditches in subtropical Florida. In shallow water table environments, most drainage is from subsurface rather than overland flow. This unique hydrology combined

### Table 2. Mean inflow and outflow concentrations of N species for year 1 (2009-2010) and year 2 (2010-2011).

<table>
<thead>
<tr>
<th>Nutrient Species</th>
<th>Pump 1</th>
<th>Pump 2 ( ^{[a]} )</th>
<th>Pump 3</th>
<th>Back Flow</th>
<th>Mean Inflow (mg L(^{-1}))</th>
<th>Mean Outflow (mg L(^{-1}))</th>
<th>Reduction ( ^{[b]} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>NH\textsubscript{4}-N</td>
<td>0.54</td>
<td>0.86</td>
<td>0.67</td>
<td>0.30</td>
<td>0.64</td>
<td>0.30</td>
<td>53%</td>
</tr>
<tr>
<td>NO\textsubscript{x}-N</td>
<td>0.74</td>
<td>1.97</td>
<td>0.62</td>
<td>0.11</td>
<td>0.88</td>
<td>0.27</td>
<td>69%</td>
</tr>
<tr>
<td>DIN</td>
<td>1.28</td>
<td>2.84</td>
<td>1.29</td>
<td>0.41</td>
<td>1.52</td>
<td>0.57</td>
<td>63%</td>
</tr>
<tr>
<td>TKN</td>
<td>2.38</td>
<td>2.98</td>
<td>2.30</td>
<td>1.44</td>
<td>2.43</td>
<td>1.74</td>
<td>28%</td>
</tr>
<tr>
<td>ON</td>
<td>1.85</td>
<td>2.11</td>
<td>1.64</td>
<td>1.14</td>
<td>1.79</td>
<td>1.44</td>
<td>19%</td>
</tr>
<tr>
<td>TN (SEM)</td>
<td>3.12 (0.10)</td>
<td>4.95 (0.30)</td>
<td>2.92 (0.07)</td>
<td>1.55 (0.91)</td>
<td>3.31</td>
<td>2.01 (0.03)</td>
<td>39%</td>
</tr>
<tr>
<td>NH\textsubscript{4}-N</td>
<td>0.53</td>
<td>-</td>
<td>1.01</td>
<td>0.55</td>
<td>0.69</td>
<td>0.35</td>
<td>49%</td>
</tr>
<tr>
<td>NO\textsubscript{x}-N</td>
<td>0.96</td>
<td>-</td>
<td>0.71</td>
<td>0.21</td>
<td>0.78</td>
<td>0.07</td>
<td>91%</td>
</tr>
<tr>
<td>DIN</td>
<td>1.42</td>
<td>-</td>
<td>1.62</td>
<td>0.62</td>
<td>1.37</td>
<td>0.42</td>
<td>69%</td>
</tr>
<tr>
<td>TKN</td>
<td>2.71</td>
<td>-</td>
<td>3.32</td>
<td>1.44</td>
<td>2.71</td>
<td>1.85</td>
<td>32%</td>
</tr>
<tr>
<td>ON</td>
<td>1.95</td>
<td>-</td>
<td>1.85</td>
<td>0.60</td>
<td>1.74</td>
<td>1.49</td>
<td>15%</td>
</tr>
<tr>
<td>TN (SEM)</td>
<td>3.67 (0.39)</td>
<td>-</td>
<td>4.03 (0.41)</td>
<td>1.65 (0.22)</td>
<td>3.49</td>
<td>1.94 (0.06)</td>
<td>44%</td>
</tr>
</tbody>
</table>

\( ^{[a]} \) Due to malfunctioning of the autosampler at pump 2, pump 1 concentrations were used for analysis for year 2. No significant difference was found between pump 1 and 2 concentrations in year 1. Both pumps are located in the same segment of the outer borrow ditch, which receives drainage from the seepage-irrigated part of the farm.

\( ^{[b]} \) Reduction = \[[\text{mean inflow concentration} - \text{mean outflow concentration}] / \text{mean inflow concentration}\] \times 100.
with the low potential for erosion in sandy soils because of the large particle size indicates that most of the incoming organic N was in dissolved form (Shukla, 2014). Furthermore, the fill and spill design criterion for the SDA greatly enhances sedimentation. If the incoming organic N was in particulate form, the retention should have been significantly higher than observed, indicating that it was indeed in dissolved state.

Positive correlation between N loading and N retention is well established for wetlands and lakes (Saunders and Kalff, 2001). A similar observation was made at the SDA, with year 1 retaining 2,222 kg more total nitrogen (TN) than year 2 as a result of receiving 4,345 kg more TN (table 1). Almost 50% (1,881 kg) and 38% (615 kg) of the surface TN load retained was in the form of DIN during years 1 and 2, respectively. During year 1, water retention (49%; table 1) was less than TN (62%; table 1), indicating that biogeochemical processes of nitrification-denitrification played a role in treating N beyond what was achieved through volume reduction. However, there was a close agreement between water retention (84%; table 1) and TN treatment (87%; table 1) in year 2, providing evidence for limited biogeochemical pathways for N treatment (fig. 5). This observation, coupled with the noteworthy differences in reduction in NO\textsubscript{3}-N (91%; table 2) and NH\textsubscript{4}-N (49%; table 2) concentrations in year 2, reinforced the supposition that extensive denitrification of the incoming NO\textsubscript{3}-N in the inner borrow ditch led to the majority of the N treatment.

Approximately 59% of the incoming drainage was retained by the SDA during the monitoring period. Assuming that the DIN retention was essentially a result of volume reduction and denitrification, the average denitrification rate was estimated to be 0.009 g m\textsuperscript{-2} d\textsuperscript{-1}. The average SDA denitrification rate was two orders of magnitude less than the average rate reported for constructed wetlands (0.5 g N m\textsuperscript{-2} d\textsuperscript{-1}; Vymazal, 2007), suggesting that measures are possible to enhance the N treatment potential of the SDA.

The N retentions of 62% and 87% in years 1 and 2, respectively, are based on the assumption that the N mass that did not leave the system through surface flow remained in the SDA (table 1). This is not likely because of significant seepage losses from the SDA (Shukla et al., 2015). Approximately half (45%) of the incoming TN was in the form of organic N (table 1). Given that organic N usually has a short-term retention potential in soil (McDowell, 2003), the subsurface movement of organic N from the SDA could be an important source of N to downstream waterbodies. Previous studies have suggested that during subsurface movement of N, a large part of the NO\textsubscript{3}-N is lost through denitrification under the anoxic conditions, thereby enhancing N treatment (Koron, 1992; Kliewer and Gilliam, 1995; Brauer et al., 2015). This is especially true for the warm, wet environment of the study area (Spalding and Exner, 1993). Denitrification is also a function of the organic matter content of the soil. For the sandy soils of south Florida, organic matter content less than 0.91% can be a denitrification-limiting factor (Tsai, 1989). The average organic matter content of surface (7.5%) and subsurface (3.2%) soil samples was greater than 0.91%. Furthermore, the groundwater fluxes also include the bank losses. Given the rapid rise in water levels inside the SDA during a pumping event and the simultaneous decrease in the outer borrow ditch, a favorable hydraulic gradient is created for the water to move out of the SDA and back to the outer borrow ditch. Drainage ditches in south Florida have been reported to have organic matter contents of almost 16% (Collins, 2005). The path of subsurface losses from the SDA is unknown and difficult to predict; however, given the facts above, we assume 100% treatment of DIN during underground transport. Under this scenario, the only N form leav-
ing the SDA through subsurface pathways would be dissolved organic nitrogen (DON). The TN treatment efficiency, taking the subsurface DON losses into account, was estimated to be 51% and 67% for years 1 and 2, respectively. The difference between treatment efficiencies, including and excluding subsurface N losses, shows that subsurface N losses need to be quantified to estimate the true treatment efficiency of the SDAs. Overall, the SDA was a net sink of TN, providing an average of 54% to 68% TN treatment, depending on whether or not groundwater N losses are considered as part of the N budget.

A relatively low net treatment efficiency, after considering seepage losses and a denitrification rate considerably lower than that reported for constructed wetlands, indicates that there is a potential to enhance the N treatment efficiency of the SDA. We propose harvesting the aboveground biomass to increase the N treatment efficacy of SDAs. Furthermore, we recommend developing a payment for environmental services (PES) program, a mutually beneficial venture for SDA owners and the state.

**FUTURE RESEARCH: BIOMASS HARVESTING FOR LONG-TERM N TREATMENT ENHANCEMENT**

Phytoremediation of impaired waters has been discussed for a long time in the research community (Huet et al., 2005; Vymazal, 2007; Dhote and Dixit, 2009; Lu et al., 2010). However, opinions are divided regarding the efficiency of plant nutrient uptake (Lee et al., 2009), the effect of biomass harvesting on water quality (Martin et al., 2003), and the economic viability of biomass harvesting (Jakubowski et al., 2010). Commercialization of biomass harvesting from terrestrial as well as wetland systems has been investigated to a large extent with regard to energy production (Moreira, 2006; Wang et al., 2011; Akhtari et al., 2014; Quinn et al., 2014) and carbon sequestration (Nair et al., 2009; Nath et al., 2015). However, for increasing the efficiency of nutrient treatment systems, the commercialization of biomass harvesting and use is still a long way off because of the limited information available on its effectiveness and economic viability.

To explore the potential of aboveground biomass harvesting for enhancing N treatment efficiency of the SDA, vegetation samples were collected and analyzed for TKN (table 3). Water lettuce was found to have the highest aboveground tissue TKN concentration, followed by primrose willow, torpedograss and smartweed, Carolina willow, and cattail. Although highest in concentration, water lettuce had much less biomass N than the other vegetation types within the SDA. Given their dominance, torpedograss and smartweed were identified as the leading source of aboveground tissue N (table 3). The estimated total aboveground biomass N was 1,079.3 kg (72.7 kg ha⁻¹).

Not all the aboveground vegetation can be harvested, given the limitations arising from saturated ground conditions, high vegetation density, wildlife, etc. Keeping the aforementioned limitations in mind, the best time to harvest would be the dry season, when water levels are at their lowest. Furthermore, the dry season marks the end of the growing period, when leaf shedding begins, followed by the decay of leaf litter and release of sequestered N. Harvesting at the end of the dry season would ensure that a minimum of biomass N is lost as a result of leaf litter decay. A spatial analysis of the observed water levels showed that only 6.8 ha of the SDA area would be unsaturated when the water level inside the SDA is lowest and hence easily harvestable (fig. 6). The harvestable area was predominantly covered with torpedograss and smartweed. In addition to this area, wetlands dominated by water lettuce (2.6 ha) could also be easily harvested because of their proximity to the SDA embankment. Based on these facts and assumptions, the N assimilated in the aboveground tissue in the harvestable area (including wetlands) was quantified to be 748 kg.

The SDA retained 5,447 kg of N during the two-year monitoring period (table 1). Assuming one harvest per year, an additional 1,496 kg of N could be potentially retained, enhancing the two-year N treatment efficiency of the SDA to 86% from 68%. The estimate of N treatment enhancement is based on the assumption that harvesting the biomass would not have a

<table>
<thead>
<tr>
<th>Vegetation Type</th>
<th>TKN Concentration (mg kg⁻¹)</th>
<th>Area of Dominance (ha)</th>
<th>Aboveground Biomass TKN (kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Torpedograss and smartweed</td>
<td>9,208.5</td>
<td>7.1</td>
<td>539.2</td>
</tr>
<tr>
<td>Primrose willow</td>
<td>10,781.8</td>
<td>1.4</td>
<td>82.5</td>
</tr>
<tr>
<td>Water lettuce</td>
<td>17,279.4</td>
<td>2.6</td>
<td>228.1</td>
</tr>
<tr>
<td>Carolina willow</td>
<td>8,852.1</td>
<td>2.0</td>
<td>172.5</td>
</tr>
<tr>
<td>Cattail</td>
<td>8,724.1</td>
<td>1.0</td>
<td>57.0</td>
</tr>
</tbody>
</table>

[a] TKN = total Kjeldahl nitrogen.
[b] Torpedograss and smartweed grew together.
negative impact on plant growth. A total of 3,606 kg of DIN entered the SDA during the monitoring period. The areal density of the incoming DIN load (0.02 kg m\(^{-2}\)) was an order of magnitude greater than the N content of the aboveground tissue of torpedograss and smartweed (0.008 kg m\(^{-2}\)) and water lettuce (0.009 kg m\(^{-2}\)), indicating that the incoming load would be sufficient to sustain vegetation growth inside the SDA despite annual harvesting (table 3).

**Economic Feasibility of Biomass Harvesting**

Getting agricultural landowners to agree to implement biomass harvesting as a practice for enhancing N treatment efficiency of existing SDAs would be a challenge, given the significant investment they have already made in constructing detention systems. Under such circumstances, a payment for environmental services (PES) program offers a promising solution by translating the environmental services (water quality enhancement) to financial incentives. The term PES can be broadly used for a voluntary transaction of a well-defined environmental service that has a buyer and seller of the service (Engel et al., 2008; Shukla, 2014).

The first pilot PES program in Florida, the Florida Ranchlands Environmental Services Project (FRESP), was launched in 2006 to provide nutrient treatment and water storage services in south Florida. The goal of the program was to design a system in which willing ranch owners would provide water and nutrient retention services beyond the regulatory requirements in exchange for payments from state agencies (FRESP, 2016). The program was intended to reduce excessive nutrient and water discharges to Lake Okeechobee, estuaries, and the iconic Florida Everglades. The FRESP was adopted by the state through the Northern Everglades Payment for Environmental Services (NEPES) program. A similar PES program for the water quality benefits provided by SDAs by means of biomass harvesting would involve the state paying growers for the additional N retained by SDAs. Such a program would potentially increase the N treatment efficacy of existing SDAs without being a financial burden on the producers.

The economic feasibility of enhancing N treatment efficacy by harvesting aboveground biomass on an annual basis was evaluated using the parameters shown in table 4. The 20-year net present worth (NPW) of the project was estimated to be $835,000, assuming that the cost of N treatment in wetlands in south Florida is $103 kg\(^{-1}\) (SWFWMD, 2016).

A discount rate of 5% was used for the analysis, as is common practice for evaluating public projects (Shukla, 2014). Harvesting biomass to increase N treatment in the SDA would provide payments to the grower of almost $42,000 annually. Furthermore, if the composted biomass were applied as an organic amendment to the same farm that contributes drainage to the SDA, it could increase the field capacity and plant-available water by 32% and 30%, respectively, resulting in increased retention of rainfall and irrigation water as well as reduced nutrient leaching losses (Pandey and Shukla, 2006; Shukla and Pandey, 2008). In essence, applying organic soil amendments could potentially increase the crop yield without additional synthetic fertilizer while reducing the irrigation inputs. If considered as a financial incentive to the grower, the increased crop yield would further augment the NPW of the project. Harvesting biomass could easily be turned into an economic avenue for recovering the N lost from the farm while simultaneously reducing impacts of agriculture on downstream waterbodies.

**Conclusion**

The SDA retained 62% and 87% of incoming N in years 1 and 2, respectively, assuming that what did not leave the SDA through surface outflow was retained. However, 31% of the inflow volume was lost through seepage. Under the best possible scenario in which all DIN is retained prior to reaching the downstream water body through denitrification during subsurface transport, the adjusted N treatment efficiency of the SDA will be 51% and 67% for years 1 and 2, respectively. The difference between treatment efficiencies was a result of seasonal differences in rainfall, inflows, and inundation dynamics, which affect the extent of denitrification, the main avenue for N treatment. The treatment efficiency of the SDA could be enhanced by controlled biomass harvesting. Annual harvesting of the biomass could potentially increase the N treatment efficiency from 68% to 86%.

The cost of harvesting, if borne by the state through cost sharing or an innovative payment for environmental services (PES) program, could create an incentive for private landowners to adopt such improvements. The 20-year net present worth of a program involving biomass harvesting and composting would be $835,000, or about $42,000 per year. Applying the composted biomass as an organic amendment to cropland would make the movement of N between farms and SDAs a two-way process while simultaneously increasing carbon sequestration and providing the additional potential service of climate change mitigation. Excessive nutrients are the primary source of impairment of approximately 163,000 km of rivers and streams and 1,416,000 ha of lakes and reservoirs (EPA, 2016). The state of the nation’s waterbodies calls for “out of the box”, win-win solutions for treating the unavoidable nutrient losses at end-of-farm locations, which are the last point of treatment before farm drainage enters freshwater bodies. Use of SDAs combined with incentivized harvest and utilization of the SDA biomass can retain the nutrients lost from production fields, improve farm productivity, and help restore the ecological integrity of impaired waters.

**Table 4. Factors for cost/benefit analysis of harvesting SDA biomass.**

<table>
<thead>
<tr>
<th>Category</th>
<th>Unit Cost/Benefit</th>
<th>Total Cost/Benefit per Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Harvesting</td>
<td>$450 acre(^{-1})</td>
<td>$10,491</td>
</tr>
<tr>
<td>Transport of biomass to composting facility</td>
<td>$8 mile(^{-1})</td>
<td>$200(^{[b]})</td>
</tr>
<tr>
<td>Compost produced</td>
<td>$18 tonne(^{-1}) (c)</td>
<td>$755</td>
</tr>
<tr>
<td>N treatment</td>
<td>$103 kg(^{-1}) N treated(^{[d]})</td>
<td>$77,366</td>
</tr>
</tbody>
</table>

\(^{[a]}\) Includes costs of machinery and labor per Shukla (2014) and personal communication with the farmer.

\(^{[b]}\) Calculated as the difference between the market price of compost and the cost of composting, assuming a composting facility is available within 25 miles of the SDA.

\(^{[c]}\) Assuming that 40% of the biomass weight is lost during composting.

\(^{[d]}\) SWFWMD (2016).
ACKNOWLEDGEMENT

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REFERENCES


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