Western Dredging Association

Journal of Dredging

Volume 20, No. 1, September 2022

Official Journal of the Western Dredging Association

(A Non-Profit Professional Organization)

The late Tim Welp conducting field work at the Seven Mile Island Innovation Laboratory in 2020.

Produced and printed by the Western Dredging Association (WEDA)

ISSN 2150-9409

CONTENTS

Journal of Dredging Editorial Board ii

Western Dredging Association Board of Directors ii

Editor's Note iii

Sediment Distribution Pipe: Modeling Tool and Field Application
by Colton T. Beardsley, Timothy L. Welp, Brian D. Harris, Brian C. McFall, Zachary J. Tyler, and Gaurav Savant 1

Large Scale, 4-Component, Settling Slurry Tests for Validation of Pipeline Friction Loss and Pump Head Reduction Models
by R. Visintainer, G. McCall II, A. Sellgren and V. Matoušek 16

Impact of Strategic, Unconfined, Dredged Material Placement on Turbidity Within a Shallow Back Bay System: Observations from Seven Mile Island Innovation Laboratory, NJ
Kelsey Fall, David Perkey, Lenore Tedesco, and Monica Chasten 38

A Multi-Decadal Assessment of Dredged Sediment Beneficial Use Projects Part 1: Ecological Outcomes
by Jacob F. Berkowitz, Nathan R. Beane, Nia R. Hurst, Jacob F. Jung, and Kevin D. Philley 50

A Multi-Decadal Assessment of Dredged Sediment Beneficial Use Projects Part 2: Ecosystem Functions, Goods, and Services
by Jacob F. Berkowitz, Nia R. Hurst, Nathan R. Beane, Kevin D. Philley, and Jacob F. Jung 72

Aims and Scope of the Journal 89

Notes for Contributors 89

EDITOR'S NOTE

I dedicate this issue to my late friend and colleague, Mr. Tim Welp. Tim passed away unexpectedly in June 2021. Tim's technical contributions to the dredging community are numerous and will withstand the test of time. His gregarious nature and genuine interest in others were his trademarks; he will not soon be forgotten. Several papers in this issue are from his colleagues, who share their own memories. Additional papers from colleagues will likely appear over the next few issues. The breadth and importance of these articles are a testament to the many tenacles of Tim's contributions to our industry.

This issue of WEDA's Journal of Dredging includes five manuscripts covering a wide range of technical topics related to dredging and sediment management. Two manuscripts describe results from field studies conducted at the Seven Mile Island Innovation Laboratory (SMIIL) in southern New Jersey. SMIIL is a unique collaboration between Federal, State, academic, and private partners with similar interests in protecting and restoring New Jersey's important marshes. Their bold, forward-looking approach sets an outstanding example of leadership through partnership. Hopefully, many others follow this blueprint.

The first SMIIL paper, from colleagues of Tim Welp, describes a mathematical model of a sediment distribution pipe. Those who knew Tim, know that sediment distribution pipes were his latest passion. He read about the concept in John Huston's 1971 book "Hydraulic Dredging" and thought it deserved further study. It is great to see his efforts come to fruition. The second SMIIL paper (third manuscript in this issue) describes water quality impacts associated with unconfined placement of hydraulically dredged sediment on Sturgeon and Gull Islands in southern New Jersey. Using unconfined placement to nourish subsiding upland and mudflat habitat could be an economical approach for many coastal areas.

The second manuscript describes ground-breaking research on energy loss associated with slurry flow in pipes. This is an important topic for dredging companies, both in terms of dredging system design and identifying potential areas to improve efficiency. This is also an outstanding and well-written paper that you will certainly enjoy.

The last two manuscripts revisit several beneficial use sites that received dredged sediment over 40 years ago. They evaluate how well these projects meet the objectives set forth at the time of construction. I worked on similar projects as a young engineer (yes, about 40 years ago!) and was flabbergasted when a biologist colleague told me that we would not know if the projects were successful for at least 20 years. Thanks to Dr. Berkowitz and his team, we now know that these projects were quite successful. There are also some important lessons for ensuring success of future projects.

I hope that you enjoy this issue as much as I have pulling it together. Many thanks to our dedicated authors for their excellent manuscripts. I hope that you will consider submitting one yourself! Please contact me if you have any questions about the submission and review process for the Journal of Dredging.

Don Hayes
Editor, WEDA Journal of Dredging
September 2022

SEDIMENT DISTRIBUTION PIPE: MODELING TOOL AND FIELD APPLICATION

Colton T. Beardsley[1], Timothy L. Welp[1], Brian D. Harris[1],
Brian C. McFall[1], Zachary J. Tyler[1], Gaurav Savant[1]

ABSTRACT

Thin layer placement of dredged material on coastal wetlands has proven to be a beneficial technique for wetland nourishment and flood risk reduction. The sediment distribution pipe system offers a novel method of achieving thin layer placement by discharging sediment at multiple locations along the pipe length. Here, the implementation of this method is described, along with a mathematical model for predicting system behavior. A simplistic model of the system is achieved through use of hydrodynamic equations which include empirical coefficients to account for effects of viscosity. The model is calibrated by optimizing these empirical coefficients to match the measurements taken in the field application, and the results are shown to provide reasonable estimates of system behavior. In-pipe slurry velocity and pressure are also calculated and provided in the modeling tool.

Keywords: Wetland nourishment, Beneficial Use of Dredged Material (BUDM), Natural and Nature Based Features (NNBF), coastal wetlands, thin layer placement

INTRODUCTION

Coastal wetlands are a critical natural or nature-based feature (NNBF) that provide a suite of ecosystem benefits and can provide flood risk reduction capability (Narayan et al. 2016). Unfortunately, coastal wetlands are particularly vulnerable to sea level rise (Mitchell et al. 2017). The advantages of beneficially using dredged sediment to nourish wetlands have been documented in several studies (Ray 2007) and typical wetland plants (e.g., *Sporobolus alterniflorus*) have the capacity for rapid recovery after a thin layer of dredged sediment [< 1 ft. (0.3 m)] is placed in the marsh environment (Berkowitz et al. 2019).

Historically, thin layer placement in the marsh environment uses a discharge pipe to spray (Figure 1a) or spill sediment slurry on the marsh. Heavy machinery is required to move the discharge pipe, which can damage the susceptible marsh surface and reduce dredging production rates. The Sediment Distribution Pipe (SDP) is an innovative technique that places sediment along a length of pipe (Figure 1b). The SDP can be elevated on cribbing with holes in the bottom of the pipe that allows the placed sediment slurry to flow in multiple directions or the SDP can be placed on the marsh surface, discharged to the side, and the pipe itself can act as a containment measure. The potential benefits of the SDP include efficiently placing sediment over a larger area, reducing containment costs, and reducing the risk of construction-related damage to the wetland. Any discharge pipe can be easily converted into a SDP by cutting holes into the pipe, but fundamental hydraulic questions about the slurry “throw distance” discharged from the pipe and the influence of the holes on the pressure and velocity are important to understand to minimize clogging by natural and artificial debris (*e.g.*, shells, seaweed, garbage). This manuscript describes a tool that can easily be used to calculate the throw distance and the pressure and velocity in the pipe to test different hole configurations in an SDP.

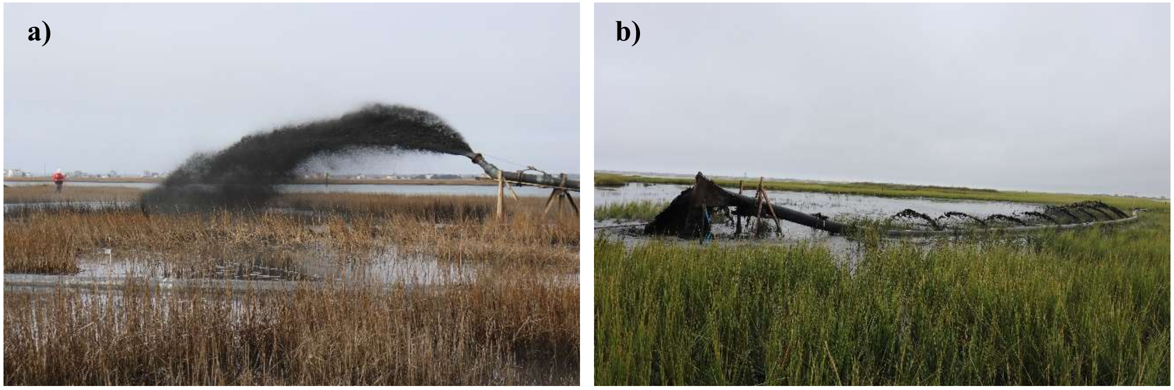

Figure 1. (a) Spray nozzle placing sediment on the marsh and (b) SDP placing sediment over a larger area.

The manuscript uses U.S. Customary units because all USACE dredging contracts are written in this form and the developed program detailed in this manuscript is currently only available in these units. However, SI units are shown in parentheses in the text for general information.

MODELING METHODS

Modeling of the slurry flow through the pipe and out of the discharge holes is accomplished through the hydrodynamic formulations of mass and momentum conservation. The computational model is implemented in Python as a one-dimensional algorithm to iteratively solve the pipe flow based on the user specifications. Pipe flow conditions of pressure and velocity at the upstream end of the pipe are assumed. The method then progresses downstream, considering the features along the user-defined pipe design to solve for flow conditions throughout the pipe. For sections of the pipe without any discharge holes, the only phenomena that requires consideration is a pressure loss due to friction. This pressure loss effect is modeled by Equation 1, also known as the Darcy-Welsbach Equation. This method formulates the pressure loss, Δp, as a function of the length of the pipe sections, L, mean flow velocity, V, flow density, ρ, and Darcy friction factor, f_D.

$$\Delta p = \frac{1}{2D} \rho f_d L V^2 \tag{1}$$

The user specifies the value for the Darcy friction factor or it can be determined by a numerical solution to the Colebrook-White equation, Equation 2. This equation defines the friction factor implicitly as a function of the Reynolds number, Re, pipe inside diameter, D, and characteristic roughness height, ϵ. This equation is solved iteratively with Newton's method for the friction factor at each pipe section.

$$f_d^{-\frac{1}{2}} = -2 \log_{10} \left(\frac{\epsilon}{3.7D} + \frac{2.51}{Re} f_d^{-\frac{1}{2}} \right) \tag{2}$$

The influence of a hole cut into the pipe is an important yet complicated phenomena in the sediment distribution pipe system. The modeling of this feature is accomplished by considering the flow of mass and momentum through a control volume, Figure 2. The presented model implements this feature by considering the effects of the hole as a reduction of mass flow rate and momentum from the internal pipe flow by approximating the velocity of the slurry discharge through a hole using the principle of constant stagnation pressure along a streamline in ideal flows. Because the flows modeled are heavily influenced by viscous effects, a discharge coefficient, ξ, that is the ratio of the resulting velocity to that of an ideal

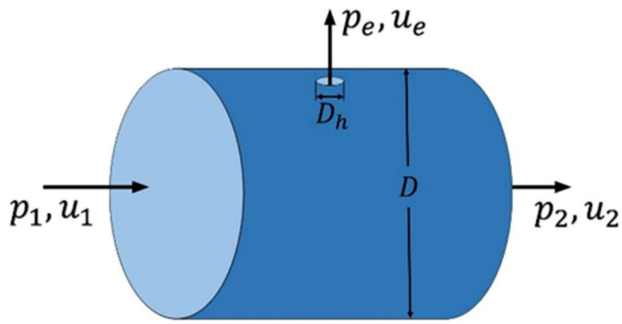

Figure 2. Diagram of a representative control volume considered when determining the influence of a hole cut into the pipe.

flow model, is introduced. The resulting expression for the exit velocity through the hole, u_e, is given in Equation 3 as a function of the discharge coefficient, upstream pressure in pipe, p_1, upstream velocity in the pipe, u_1, and fluid (slurry) density, ρ. The downstream velocity, u_2, is then determined by balancing the mass flow through the control volume.

$$u_e = \xi \left(u_1 + \frac{2p_1}{\rho} \right)^{\frac{1}{2}} \tag{3}$$

The presence of the hole causes an increased resistance in the pipe flow, which can be represented by a pressure loss to the internal flow. This pressure loss formulation is based upon a drag coefficient type model where the drag force due to the hole is applied to the pipe cross section to yield a net pressure reduction in the pipe proportional to a resistance coefficient, κ. Using this and Bernoulli's principle to evaluate the changing velocity's effect on the internal pressure, the following expression for the downstream pressure is developed:

$$p_2 = p_1 + \frac{1}{2}\rho(u_1^2 - u_2^2) - \frac{\kappa}{2}\rho u_1^2 \left(\frac{D_h}{D}\right)^2 \tag{4}$$

These modeling equations allow a user to begin with upstream conditions and solve the pipe for the downstream solution, but, in many cases, users would like to constrain their problem with downstream and upstream conditions. For example, a dredge operator likely knows the pressure in the discharge pipe at the dredge and that the pressure at the end of pipe (EOP) is atmospheric. The application gives the user a choice of boundary conditions. The downstream conditions can be enforced by iteratively solving for the upstream conditions that result in the specified downstream conditions using a secant method. This algorithm works the same as Newton's method but replaces the derivative term with a finite difference formulation to avoid deriving analytical derivatives of lengthy functions.

The final feature of the application is the ability to predict the throw distance of the slurry jets. The throw distance is calculated by modeling the fluid jet using kinematic equations of a projectile under the influence of air drag that is proportional to the square of the velocity magnitude and acting in a direction opposing the velocity vector. The resulting differential equations that describe this motion are non-linear ordinary differential equations, given in Equations 5 and 6. Here, C_d is a drag coefficient, ρ_a is the air density, and ρ_l is the density of the slurry mixture. The dot accent above directional variables refer to the time derivative of that variable.

$$\ddot{x} = -C_d \frac{\rho_a}{2D_h \, \rho_l} \dot{x}\sqrt{\dot{x}^2 + \dot{y}^2} \tag{5}$$

$$\ddot{y} = -C_d \frac{\rho_a}{2D_h \, \rho_l} \dot{y}\sqrt{\dot{x}^2 + \dot{y}^2} - g \tag{6}$$

By defining a state vector $\vec{X}$, that includes the position and velocity of the fluid projectile, this set of two, second-order equations can be written into a set of four, first-order equations allowing a numerical integration with respect to time.

$$\vec{X} \coloneqq \begin{bmatrix} x \\ y \\ \dot{x} \\ \dot{y} \end{bmatrix} \xrightarrow{yields} \dot{\vec{X}} = \begin{bmatrix} \dot{x} \\ \dot{y} \\ -C_d \frac{\rho_a}{2D_h \, \rho_l} \dot{x}\sqrt{\dot{x}^2 + \dot{y}^2} \\ -C_d \frac{\rho_a}{2D_h \, \rho_l} \dot{y}\sqrt{\dot{x}^2 + \dot{y}^2} - g \end{bmatrix} \tag{7}$$

An Adams-Bashforth method of time integration was chosen as it is an explicit integration method and can be implemented easily, while also providing a second-order temporal accuracy (Peinado et. al., 2010). This time integrator is a type of linear multistep algorithm. The time step size is another parameter that the user can control. The stopping condition for the integration is the y-value reaching a negative value. The x-value at this final time is then taken as the throw distance as measured from the pipe center. Because of the discrete nature of the time integration, the y-value is unlikely to be zero at any particular time step and the error incurred in using a nonzero y-position is well within an acceptable range.

As of the current implementation, there is no recommendation for appropriate values for the modeling constants ξ, C_d, and κ. These values must be evaluated by physical model experiments and these experiments have yet to be conducted. The calibrated values for these constants determined in the field application of this manuscript may provide a first order approximation for future projects until additional guidance is provided. Future research will include more robust experimental and computational studies that will yield insight to the formulations given here and the values of modeling coefficients.

MODELING TOOL OVERVIEW

The Sediment Distribution Pipe Modeling Application (SDPMA) is designed to allow users to apply the described equations to generate an arbitrary pipe design within the graphical user interface (GUI), specify necessary inputs, and calculate a prediction for the flow through the pipe and the throw distance through each of the holes. The SDPMA interface is shown in Figure 3. A diagram is included in the interface so users can reference input parameters and ensure consistent definition of input variables. The interface also contains a graphics window that updates as the user designs their distribution pipe, modifies a design, generates a solution, or alters the output control settings. The application features can be grouped into four main categories: basic inputs, hole placement, solution and output control, and a file storage method.

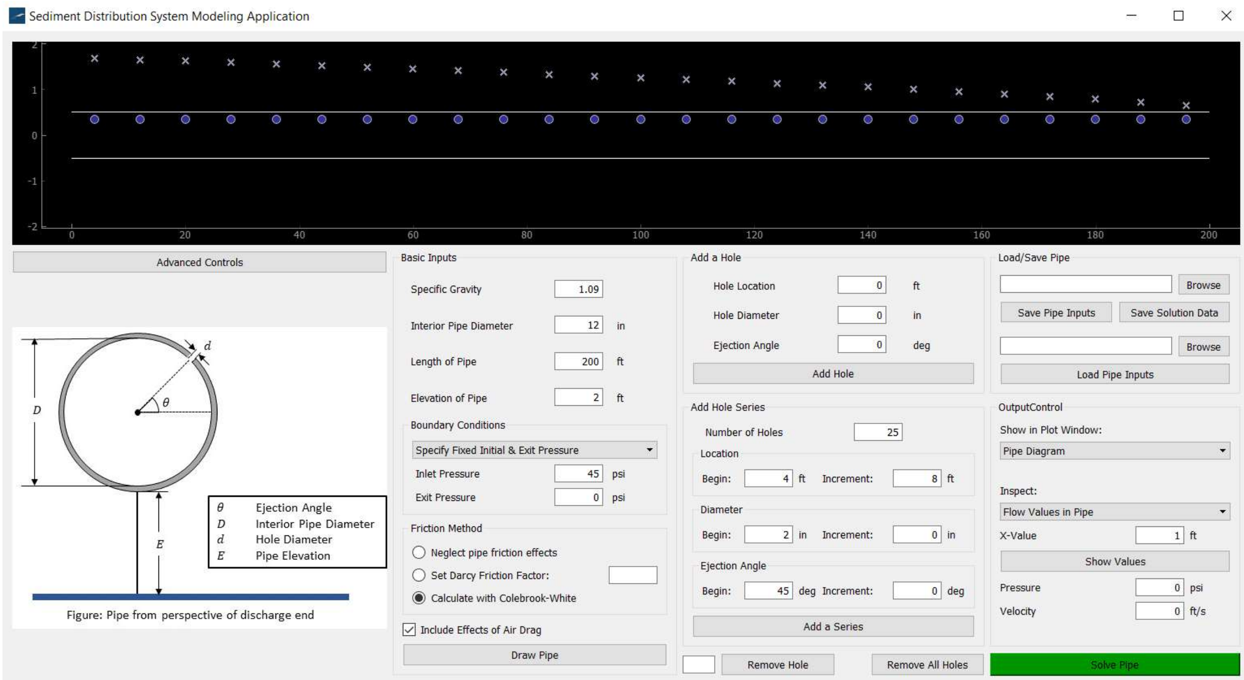

Figure 3. Screenshot of the user interface for the SDPMA.

The user begins by specifying the basic inputs pertaining to the pipe design, along with modeling options available. In addition to these inputs, the user may specify more numerical and model controls through an Advance Controls pop-up menu. The user then places the holes using the dedicated modules, with the option to place a single hole at a time, or to place a series of holes that vary incrementally in size, distance, or angular position along the cross-section. A representation of the pipe and holes are updated continually in the graphics window.

The user then must select the "Solve Pipe" button (green) to generate a solution. This updates the graphics window containing the pipe diagram to include positions of the sediment placement for each hole. The user can also choose to replace this pipe diagram with a plot of the pressure or velocity as a function of the length along the pipe. The user can save the setup of their problem to a comma-separated values (CSV) file, which can be uploaded and modified later. Also, the user can save the solution data along the pipe in CSV file format for plotting or manipulating in another program. Another use for these set-up files is for the rapid generation of a pipe design using another code or program with the ability to load-in and solve the pipe with SDPMA.

Here, an example of a sediment distribution pipe is considered with a length of 200 ft. (61 m) and interior diameter of 12 in. (0.3 m), lying 2 ft. (0.6 m) above the surface of the wetland. The specific gravity of 1.09 for the fluid (slurry) was used. The Colebrook-White equation was utilized to solve for the Darcy friction factor to calculate the friction losses in the pipe. Pressure at the upstream end of the pipe is set to 45 psi, and the downstream value is set to 0 psi because the EOP is open to atmospheric pressure. Twenty-five holes are added in a series every 8 ft. (2.4 m) beginning 4 ft. (1.2 m) from the pipe's inlet end with a diameter of 2 in. (0.05 m) and cut at a 45-degree angle to the cross-section. A discharge coefficient (ξ) of

0.05, resistance coefficient (κ) of 1.3, and air drag coefficient (C_d) of 0.8 were used. All numerical controls are left to their default values, as well as are the remaining physical parameters within the Advanced Controls window. After solving the pipe, Figure 4 shows the results for velocity along the length of the pipe.

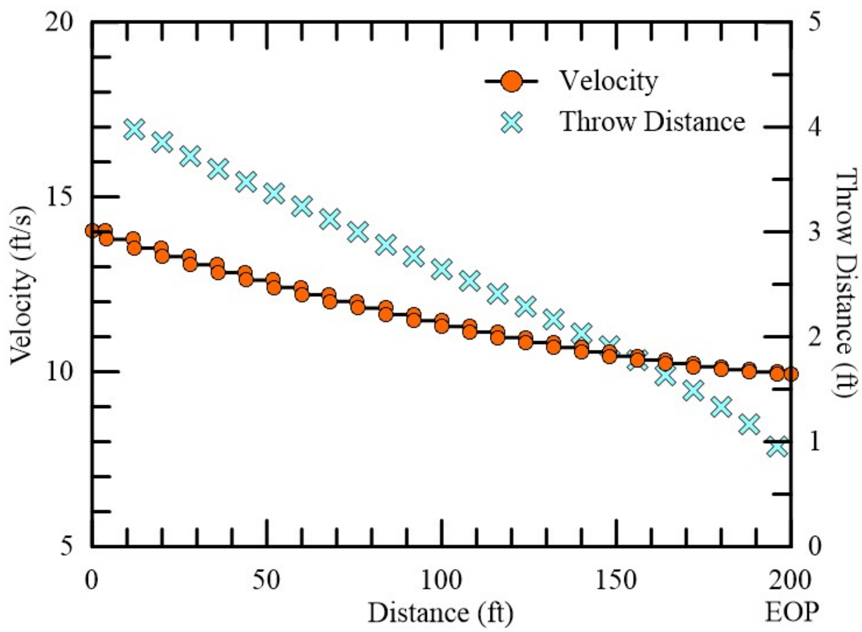

Figure 4. Velocity and throw distance as a function of the length along the pipe as measured from the upstream end for the example scenario.

At each hole location, the flow in the pipe experiences a reduction in velocity, which coincides with a reduction in pressure. Pressure at the end of the pipe reaches its specified boundary condition and the velocity throughout the pipe is solved iteratively. While the reduction in pressure over each subsequent hole experiences slight variation, the change in velocity over the holes changes significantly. This is because the velocity out of the hole is determined by the upstream pressure, so that when the upstream pressure is reduced, the velocity out of the hole is very small, meaning that the velocity within the pipe is maintained. This trend is seen in the plot of throw distances for each of the holes in the pipe in Figure 4.

The value of modeling coefficients in this application can improve the accuracy of the sediment distribution pipe system's behavior prediction. Therefore, maintaining the design used in the preceding example, the sensitivity of the pipe flow predictions to the resistance coefficient and the discharge coefficient were investigated. Since the discharge coefficient is a ratio of the exit velocity out of the hole to the ideal flow solution, a constant factor of two between each of the five different discharge coefficients was selected. These discharge coefficients are each used in the solution for the pipe with four different resistance coefficients separated by an increment of 0.8. The velocity distribution was selected to represent the solution differences as the boundary values of pressure are specified and the velocity value through the pipe may be of more importance as the velocity influences the potential for sediment to deposit within the pipe and cause build up that adversely impacts the system effectiveness. The resulting solutions are plotted in Figure 5.

A distinctive trend between the reduction in discharge coefficient resulting in lower initial velocities and higher exit velocities is evident for much the same reason as a reduction in hole diameter has these effects.

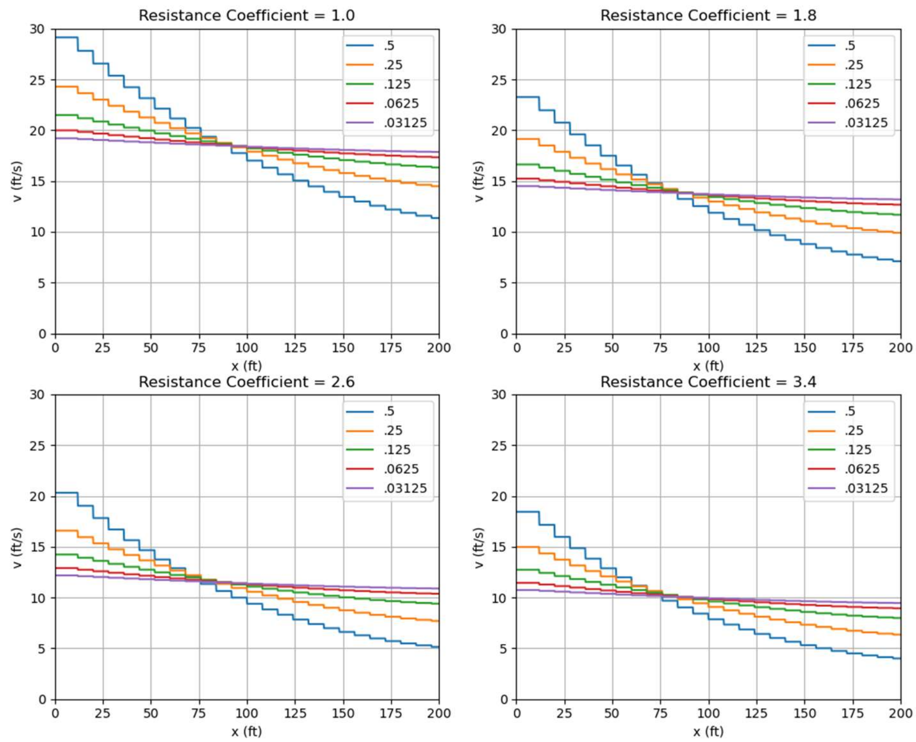

Figure 5. Velocity data for a variety of discharge coefficients (ξ) and for four different resistance coefficients (κ) used when modeling the example pipe design.

Furthermore, this effect seems slightly more prevalent when the resistance coefficient is small. The decrease in resistance coefficient results in a significant increase in initial pipe velocity because the reduction in pressure scales with the velocity squared. Therefore, to get the same pressure reduction across the entire length of the pipe, satisfying the specified boundary conditions, the velocity values in the pipe must be smaller. It should be noted that this does not account for changes in pump performance against different pressure heads.

FIELD APPLICATION

Nourishment Background

The SDP was used to place dredged sediment on a 13.5-acre island called Sturgeon Island that is adjacent to the New Jersey Intracoastal Waterway (NJIWW). The site is located within a 4200-acre tidal marsh complex that has limited freshwater input (Ferland 1990, Berkowitz et al. 2018, Harris et al. 2021). Prior to nourishment, the average elevation of the placement site was 1.5ft. (0.5 m) NAVD88 [SD ±0.5 ft.(0.15 m)] and was primarily vegetated by *Sporobolus alterniflorus* (Harris 2020). The southern portion of the island is dominated by *Phramites australis* and shrubby vegetation that occupies the higher elevations resulting from historic dredged material placement from maintenance of the NJIWW.

The SDP was applied to the Sturgeon Island site in three distinct layouts over two dredging maintenance events of the NJIWW in March and September 2020. The dredging for both events was performed by the cutterhead dredges, *FULLERTON* and *MONTGOMERY*, that are owned and operated by Barnegat Bay Dredging Company using a 14 in. (0.36 m) and 12 in. (0.3 m) discharge pipelines, respectively. For the March 2020 event, the 14 in. pipe was brought onto the northern portion of the island and split via a wyevalve into a 12 in. (0.3 m) High Density Polyethylene (HDPE) pipe to the west that terminated with a spreader plate or spray nozzle and the 12 in. HDPE SDP to the east, Figure 6. The last 70 ft. (21 m) of the SDP was elevated with wooden scaffolding 3.6 ft. (1.1 m) above the wetland surface and dredged sediment flowed through five holes cut into the bottom of the pipe, Figure 7a. A 14 in. (0.36 m) water filled pipe and hay bale/wooden dam was employed along the eastern boundary of the island to retain the dredged sediment on the island. After 4 days of dredging, 3,250 CY (2475 m^3) of material was dredged.

During the September 2020 maintenance event, a 14 in. (0.36 m) pipe was brought onto the western portion of the island. It was reduced to a 12 in. (0.3 m) HDPE pipe and oriented to discharge towards the northern end of the island. The layout consisted of only one pipe laid directly on the wetland surface. Holes were cut into the pipe to create an SDP that allowed material to flow towards the east, Figure 7b. Several hole configurations were tested in this pipe layout with round hole diameters ranging from 2 in. to 4 in. (0.05 m to 0.1 m) cut into the pipe to discharge slurry 45° from horizontal. The holes tended to get blocked with debris (e.g., shells, sea grassweed) and were enlarged to reduce blockage. The final hole configuration consisted of nine 3 in. (0.08 m) holes and three 4 in. (0.1 m) holes as shown in Table 1. The pipe laid in this orientation for 17 days and was then repositioned along the northwestern edge of the island for two days to test placing sediment on the marsh edge to build a marsh edge protection feature, Figure 7c. This manuscript focuses on the first two application scenarios that nourished the wetland, and the SDPMA is calibrated using the second configuration, Figure 7b.

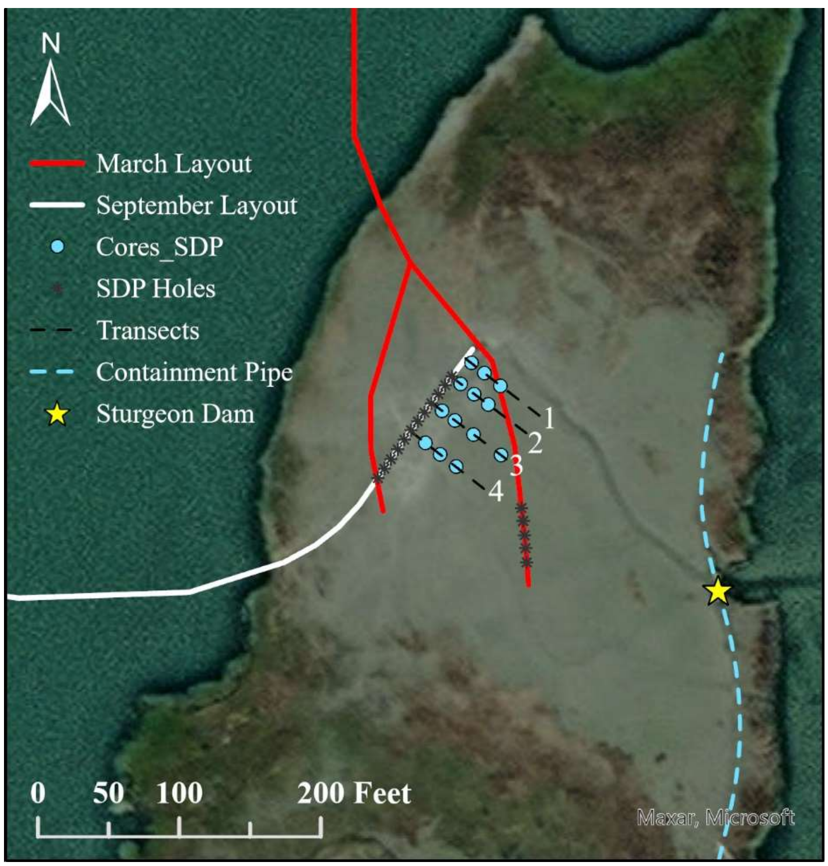

Figure 6. Sturgeon Island discharge pipe configurations during the spring and fall dredge maintenance events, hole locations on the SDP, and coring locations during the July 2021 field expedition.

Figure 7. SDP application scenarios: (a) elevated pipe, (b) placed on the marsh surface, and (c) placed on the marsh edge to build an edge protection feature.

Table 1. Hole configuration during the Fall 2020 dredging event.

Hole #	Distance from EOP (ft.)	Hole Diameter (in.)
1	112	3
2	104	3
3	96	3
4	88	3
5	80	3
6	72	3
7	64	3
8	56	3
9	48	3
10	40	4
11	32	4
12	24	4

To investigate the impact the SDP had on the soil stratigraphy, sediment samples were collected along four transects moving away from the September 2020 terrestrial placement, Figure 6. During a field investigation in July 2021, a Russian Peat Corer was used to determine the depth of dredged sediment and samples were collected to determine the grain size distribution. This type of corer was used due to the presence of a root mat and its ability to cause less compaction during collection than typical coring methods. Figure 8 highlights four transects of wetland surface elevation at varying times [i.e., March 2020 (pre-placement, July 2020 (in-between placements), and July 2021 (post-placement)], as well as the depth of dredged sediment and sediment type from the cores. The sediment type was divided into two classifications based on the United Soil Classification System (USCS): coarse-grained (>0.075 mm) and fine-grained (<0.075 mm).

Elevation gain was measured in the areas nourished by the SDP. Localized indentions in the wetland surface were noted 6 ft. to 10 ft. (1.8 m to 3 m) from the SDP which corresponded to the measured throw distances from the SDP. On Transect 1, a 0.5 ft. (0.15 m) sand lens was measured 5 ft. (1.5 m) from the outfall, but farther from the SDP, the sand was overlain by fine-grained sediment. This differed from the other transects and was most likely attributed to this transect being closer to the EOP than the holed section of the SDP. A 1 ft. (0.3 m) thick sand layer was recorded across Transect 2 from 6 ft. to 20 ft. (1.8 m to 6.1 m) away from the SDP and decreased to a thickness of 0.3 ft. (0.1 m) at 32 ft. (9.7 m) from the SDP. Transect 3 was nearly in the center of the SDP holes and exhibited the most pronounced sand bulb with a thinner 0.3 ft. (0.1 m)

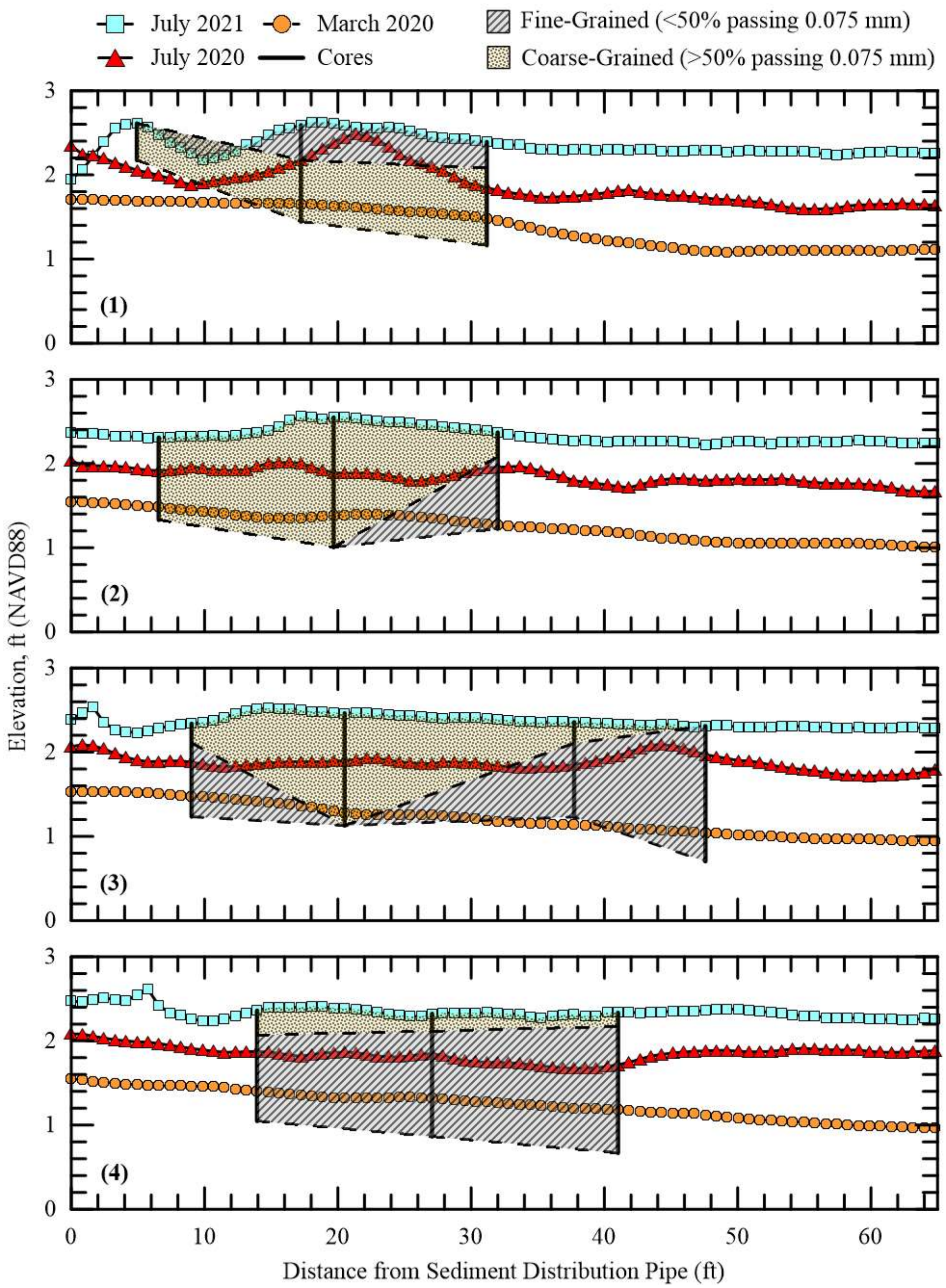

Figure 8. Transects of sediment type based on USCS and elevation profiles moving downstream of the SDP outfall. Transect numbers correspond to numbering shown in Figure 6.

layer of sand at 9 ft. (2.7 m) that increased to a max thickness of 0.5 ft. (0.15 m) at 21 ft. (6.4 m), but then decreased back to 0.3 ft. (0.1 m) at 38 ft. (11.6 m) before transitioning to all fine-grained material at 48 ft. (14.6 m). Lastly, Transect 4 exhibited the lowest amount of coarse-grained material compared to the other transects with a maximum thickness of 0.3 ft. (0.1 m) at 14 ft. (4.3 m) that stayed relatively consistent as distance from the SDP increased. It is possible the coarse-grained layer may be thicker closer to the SDP outfall (EOP), but a closer core was not collected.

Ultimately, the SDP nourished a large portion of the island, raising the surface elevation more than 1 ft. (0.3 m). Laying the SDP on the marsh surface and discharging to the side did not create any discernable features even though the cores indicated an increased amount of coarse material closer to the SDP. It is hypothesized that the area closer to the SDP will become more pronounced due the coarse-grained material not being as compressible as the predominately fine-grained material farther from the SDP.

Application of the SDPMA

The SDPMA is applied to the final September 2020 SDP configuration with hole locations noted in Table 1. Measuring the throw distances in the field was challenging because the slurry did not impact the ground at an exact point location but had an impact area of approximately 2-3 ft. (0.6-0.9 m) in diameter. Additionally, if seaweed was partially obstructing the discharge holes, it was removed before measuring the throw distance. This was common for the smaller holes tested. To measure the throw distance, the maximum and minimum distances to the impact area from the SDP were measured and the average of these values was used to calibrate the SDPMA. The maximum and minimum measurements are shown in Figure 10 with the bounding bars. While the throw distances were measured, the velocity was measured upstream of the SDP with a Doppler flowmeter. The mean measured velocity was 10.9 ft/s (3.3 m/s) with a standard deviation of 0.30 ft/s (0.1 m/s). The problem was constrained by applying an upstream velocity condition and a downstream pressure condition of zero since it was open to atmospheric pressure. Since throw distances are not very sensitive to the air drag coefficient, a constant value of 0.5 was assumed. The values of the discharge coefficient (ξ) and the in-pipe resistance coefficient (κ) are then calibrated to best match the throw distance measurements.

The goal of the calibration is to find the modeling coefficients that minimize the mean squared error (MSE) in the throw distance predictions as compared to the field measurements. The MSE is chosen as opposed to the mean absolute error as the MSE minimization tends to yield results that match all data points well, while the mean absolute error minimization may allow for some data points to be matched very poorly if the majority are matched very closely. This calibration is accomplished by a uniform sampling of the modeling coefficients. Here, the model was run for combinations of seventeen different values of each modeling coefficient for a total of 289 runs. The color contour shown in Figure 9 displays how the MSE changes with changes in these parameters.

Based on this analysis, a discharge coefficient (ξ) of 0.12 and an in-pipe resistance coefficient (κ) of 0.86 are optimal as the resulting MSE is 0.39 ft^2 (0.036 m^2). This approach is not able to give very precise modeling coefficient values without incurring very large computational costs, but it is easy to implement and can yield reasonable results. Once the model predictions are plotted alongside the average throw distances with error bounds that reflect the minimum and maximum throw distances for each of the holes, as seen in Figure 10, it is shown that the model yields reasonable results. The average error for the throw distances is 0.56 ft. (0.17 m).

Based on the comparison, it is evident that the assumption of the model is that all holes of equal size behave similarly (i.e., the modeling coefficients for all holes are consistent). In the field, however, many variables cause the holes to have variations, whether that is in the way they are cut, or in the presence of debris blockages and other similar factors. The goal of the tool is to provide field engineers with an estimate of throw distance with errors less than 100%. Of the several SDP configurations tested in the field, the throw distances were in the 3 to 8 ft. (1 to 2.5 m) range. A 100% error applied to such small throw distances is still accurate enough for most practical applications, particularly wetland nourishment. By this metric, the

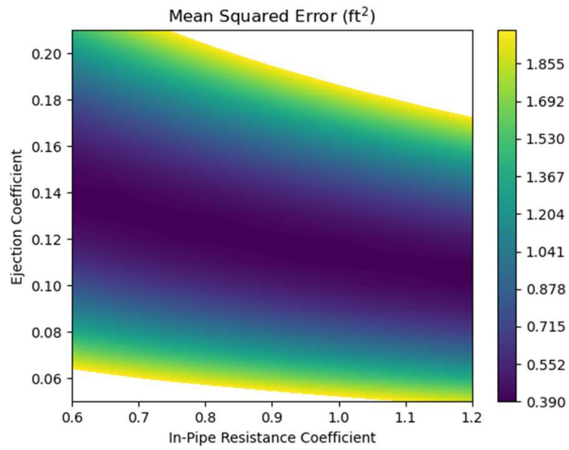

Figure 9: A color contour showing the mean squared error as a function of the two modeling coefficients considered in calibration.

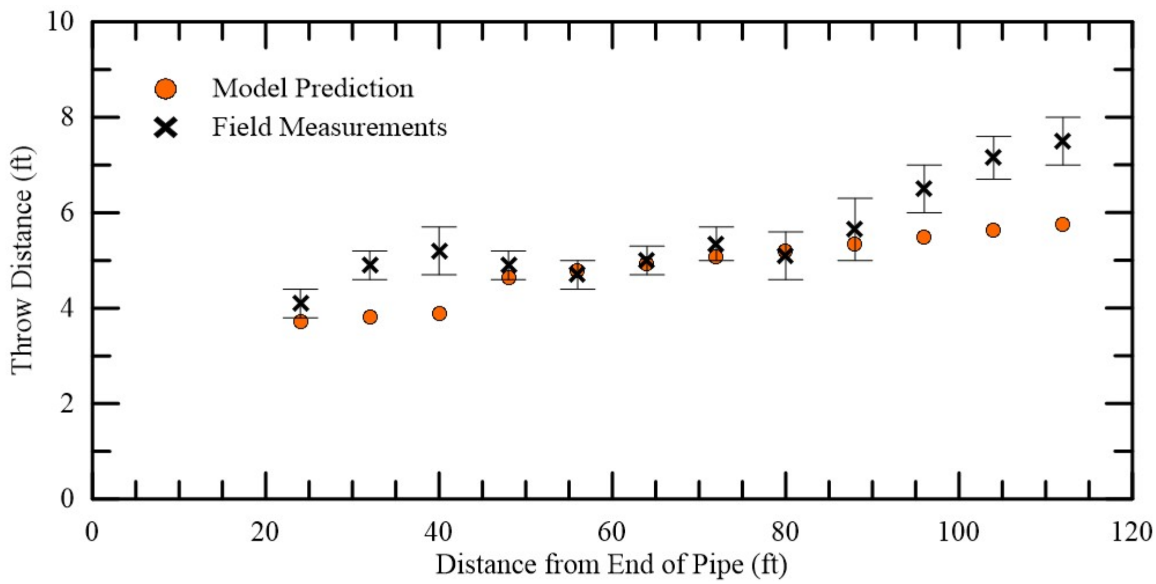

Figure 10: A plot comparing the measured throw distances for each hole to the predictions by the SDPMA after calibration.

model provides good estimates as to the throw distances once the modeling coefficients are calibrated. Much more data is required to make a conclusion as to the model accuracy, but the current model proves to be useful for initial estimates of system behavior.

An additional benefit of the modeling application is that having calibrated the modeling coefficients to match the measure throw distances, the user now has estimates for the internal flow characteristics everywhere within the pipe. This gives users access to information about the SDP system that is otherwise difficult or expensive to measure directly. The in-pipe flow characteristics are shown in the plots in Figure 11. The resulting pressure from this analysis was not constrained in our calibration.

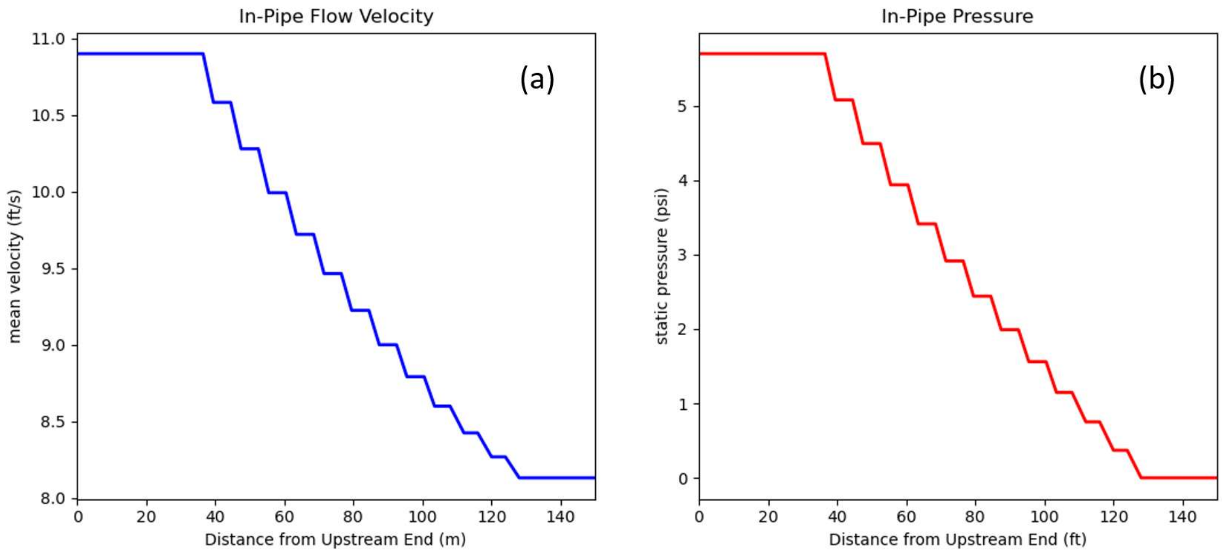

Figure 11: Plots of the average velocity (left) and the static pressure (right) for the flow within the pipe as a function of distance from the inlet end. Results are based on the Sturgeon Island SDP with model coefficients calibrated to match throw distance measurements.

CONCLUSIONS

The manuscript explains the mathematical formulations used by the Sediment Distribution Pipe Modeling Application (SDPMA) and how to best predict dredged slurry flow characteristics in the SDP. The empirical discharge and in-pipe resistance coefficients were calibrated to a field case study in coastal New Jersey. Although an extensive calibration process was detailed in this manuscript, the coefficients can also be manually tuned in the SDPMA or the noted calibrated coefficients can be initially applied to future modeling studies until additional calibration can be completed. The primary findings of this study are summarized below:

- The SDPMA predictive software tool is explained and an example case using the Python graphical user interface (GUI) is presented.
- The software tool can rapidly estimate the slurry velocity and pressure in the sediment distribution pipe section with the discharge holes and the throw distance for varying hole sizes and locations allowing for an iterative design.
- The software tool was calibrated to field data for a New Jersey thin layer placement site and was able to accurately simulate in-pipe flow characteristics like exit velocity and throw distance.

The work presented herein serves as a proof of concept of this modeling tool. Future SDP research will include conducting physical modeling to identify conditions suited for sediment sorting through the hole and computational fluid dynamics modeling of the slurry in the pipe that will allow for further SDPMA refinement.

REFERENCES

Berkowitz J. F., Vanzomeren, C. M., Piercy, C. D. and White, J. R. (2018) *Evaluation of coastal wetland soil properties in a degrading marsh*. Estuarine, Coastal and Shelf Science, Vol. 212, pp. 311–7.

Berkowitz, J. F., Piercy, C. D., Welp, T. L. and Van Zomeren, C. M. (2019). *Thin Layer Placement: Technical Definition for US Army Corps of Engineers Applications*. ERDC TN-19-1: U.S. Army Engineer Research and Development Center, U.S. Army Corps of Engineers, Vicksburg MS.

De Nevers, N. (1970). *Fluid Mechanics*. Addison-Wesley Publishing Company, Boston, MA.

Ferland M.A. (1990). *Holocene depositional history of the Southern New Jersey barrier and backbarrier regions*. CERC-90-2, USACE-WES Coastal Engineering Research Center, Vicksburg, MS.

Keady, G. (1998). *Colebrook-White Formula for Pipe Flows*. Journal of Hydraulic Engineering, ASCE, Vol. 124, No. 1, pp. 96-97

Harris, B. D. (2020). *Geotechnical Properties, Design, and Performance of Natural Infrastructure*. Louisiana State University Dissertations, Civil and Environmental Engineering Department, Baton Rouge, LA.

Harris, B., Day. D., Jafari, N., Cadigan, J., Bailey, S., and Tyler, Z. (2021). *Establishment of Soil Strength in a Nourished Wetland Using Thin Layer Placement of Dredged Material*. PloS one, Vol. 16, No. 5, pp. e0251420.

Mitchell, M., Herman, J., Bilkovic, D.M. and Hershner, C. (2017). *Marsh persistence under sea-level rise is controlled by multiple, geologically variable stressors*. Ecosystem Health and Sustainability, Vol. 3, No. 10, pp.1379888.

Narayan, S., Beck, M.W., Reguero, B.G., Losada, I.J., Van Wesenbeeck, B., Pontee, N., Sanchirico, J.N., Ingram, J.C., Lange, G.M. and Burks-Copes, K.A. (2016). *The effectiveness, costs and coastal protection benefits of natural and nature-based defences*. PloS one, Vol. 11, No. 5, pp. e0154735.

Peinado, Ibanez, Arias, Hernandez (2010). *Adams-Bashforth and Adams-Moulton methods for solving differential Riccati equations.* Computers and Mathematics with Applications, Vol. 60, No. 11, pp. 3032-3045.

Ray, G. L. (2007). *Thin layer placement of dredged material on coastal wetlands: A review of the technical and scientific literature*. ERDC/EL TN-07-1: U.S. Army Engineer Research and Development Center, U.S. Army Corps of Engineers, Vicksburg, MS.

ACKNOWLEDGEMENTS

This manuscript would not have been possible without the tireless efforts of the late Tim Welp. His continual drive to innovate dredging and placement operations led to his championing of the sediment distribution pipe research. His infectious, child-like enthusiasm for dredging research, as seen in Figure 12, will be greatly missed. Development of the SDPMA was supported by the USACE Dredging Operations Technical Support (DOTS) Program and the field campaign was supported by the Dredging Operations and Environmental Research (DOER) Program. The assistance from The Wetlands Institute, U.S. Army Corps of Engineers Philadelphia District (NAP), and New Jersey Department of Environmental Protection (NJDEP) is graciously acknowledged. The Chief of Engineers has granted permission to publish this work.

Figure 12. The late Tim Welp testing risers on the Sediment Distribution Pipe.

LARGE SCALE, 4-COMPONENT, SETTLING SLURRY TESTS FOR VALIDATION OF PIPELINE FRICTION LOSS AND PUMP HEAD REDUCTION MODELS

R. Visintainer[1], G. McCall II[2], A. Sellgren[3] and V. Matoušek[4]

ABSTRACT

A 4-component model for settling slurry pipe flow has been previously described by Wilson and Sellgren (2001) to predict pipeline friction loss over a range of slurry compositions: from fine to coarse particle size, narrow to broad particle size distribution, and low to high solids concentration. Further development of the model was undertaken by Visintainer et al. (2017a, 2021) based on a comprehensive set of laboratory tests in 203 mm (8 in) and 103 mm (4 in) pipelines. This was later adapted to the modelling of settling slurry performance reductions (derates) for centrifugal pumps (Visintainer 2017b). The goal of the present work is to validate the applicability of the 4-component model to larger pipeline sizes, typical for dredging installations. To that end, a second comprehensive test program, similar to those previously run, has been conducted in a 489 mm (20 in) pipe loop with a correspondingly larger pump. In all, 24 tests were performed with particle sizes ranging from <40 μm to 25 mm, d_{50} particle sizes from <40 μm to 11 mm and delivered solids concentrations from 4% to 40% by volume. Particle size distributions varied from very narrow to very broad, with d_{85}/d_{50} ratios ranging from 1.5 to 320. The resulting data have been compared to the current formulation of the 4-component pipeline friction loss and pump head reduction models to assess the applicability of these models for use with larger pipelines and pumps.

Keywords: Settling slurry, Pipeline friction models, Pipeline pressure gradient, Pump solids effect, Head reduction factor, 4-component model, Slurry testing.

INTRODUCTION

The solid-liquid flow of slurry in a pipeline exhibits various regimes according to the mixture velocity, particle size distribution, carrier fluid properties and pipeline geometry. Various schemes have been developed to classify and model these (Shook et al., 2002, Wilson et al., 2006). A 4-component model was originally proposed and further developed by Wilson and Sellgren to predict settling slurry pipeline friction loss over a range of slurry compositions and flow regimes: from fine to coarse particle size, narrow to broad size distribution, and low to high solids concentration (Sellgren et al., 2014). In the 4-component methodology, pipeline friction losses are calculated by a weighted average approach of the standard Newtonian pipe flow model, combined with three established settling slurry models for pseudo-homogenous, heterogeneous and fully stratified flow. The solids are partitioned into four volume fractions

[1] VP Engineering and R&D, GIW Industries Inc., 5000 Wrightsboro Rd., Grovetown, Georgia 30813, USA, T: 706-434-0526, Email: Robert.Visintainer@ksb.com.
[2] Hydraulic Lab Manager, GIW Industries Inc., 5000 Wrightsboro Rd., Grovetown, Georgia 30813, USA, T: 706-434-2286, Email: George.McCall@ksb.com.
[3] Professor Emeritus, Luleå University of Technology, Luleå, Sweden. Email: anders.sellgren@telia.com.
[4] Full Professor, Czech Technical University in Prague, Prague, Czech Republic. Email: v.matousek@fsv.cvut.cz

or "components" and each is assigned to one of the four sub-models based on the characteristic particle sizes which bound their range of application:

- **The "Carrier Fluid" fraction, *Xf*** includes all particles <40 µm. These solids are assumed to "combine" with the liquid, altering its density and dynamic viscosity.
- **The "Pseudo-homogeneous" fraction, *Xp*** includes all particles >40 µm, up to a limit of 200 µm.
- **The "Heterogeneous" fraction, *Xh*** includes all particles larger than the upper limit of *Xp* and smaller than the lower limit of *Xs*.
- **The "Stratified" Fraction, *Xs*** includes all particles larger than 0.015 times the pipeline diameter.

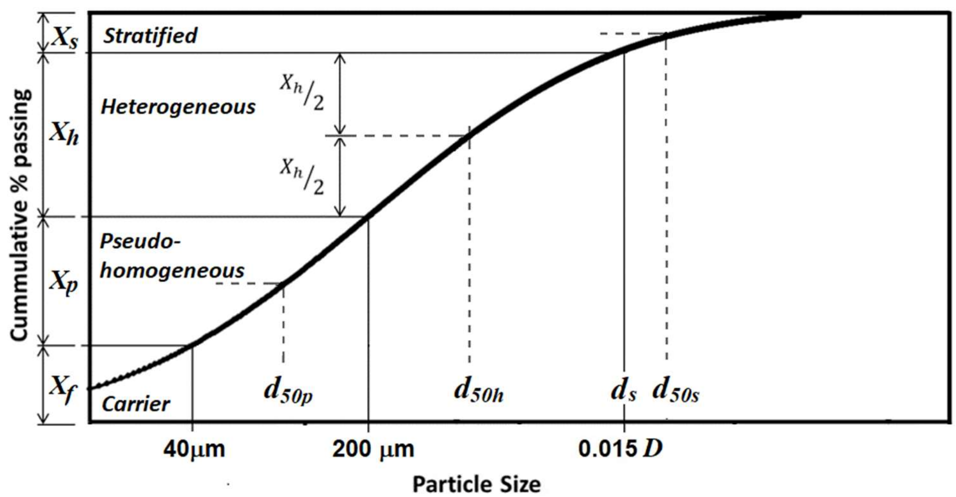

Figure 1. Representation of the 4-component fractions and their particle sizes.

The model was further developed by Visintainer et al. (2017a), based on a comprehensive, settling slurry test program performed in 203 mm (8 in) and 103 mm (4 in) pipelines, with particle sizes ranging from <40 µm to 12.5 mm, and delivered solids concentrations from 4% to 38% by volume. Analysis of the data suggested the introduction of two new empirical parameters to account for the interactions between components. The revised model provided good agreement with the experimental data over the entire range of the test program.

Based on the same series of tests, which were driven by GIW 8x10 LSA-32 and 3x4 LCC-12 slurry pumps respectively, the 4-component concept was adapted by Visintainer et al. (2017b) to the calculation of centrifugal pump performance reductions (derates) for head and efficiency. By individually accounting for the performance losses contributed by each solids component, improved accuracy over existing mono-sized models was obtained, especially for broad and bi-modal solids size distributions. In this case, empirical parameters to account for the interactions between components were not required.

In Visintainer et al. (2021), the 103 mm pipe test program was further extended to higher density solids, consisting of magnetite at a specific gravity S_S = 4.75. Further refinements were proposed, including a variable particle size boundary for the *Xf* fraction depending on S_S, and an expansion of the empirical parameter *A*" used with the pseudo-homogeneous *Xp* fraction. With regard to pump derates, a strong velocity dependence became apparent, not seen in previous testing with 2.65 S_S solids. This was addressed by introducing the three empirical parameters A'', B'' and C'' already used in the pipe friction model, thus establishing a closer correspondence between the pipe friction and pump derate models.

However, it is recognized that industrial pipeline systems, such as those used in dredging, are designed and built at scales considerably larger than the equipment used in the above-mentioned test programs. Furthermore, key variables used in the new empirical parameters are strong functions of pipeline diameter, such as the velocity at the limit of stationary deposition and the threshold velocity at full suspension (i.e. the pseudo-homogeneous velocity). Without a similar, comprehensive data set in some larger system, extrapolation of the models beyond the tested pipeline and slurry pump sizes becomes increasingly uncertain. Therefore, a series of tests was undertaken in the GIW Hydraulic Lab using a newly constructed 489 mm (20 in) pipeline and GIW 24x24 TBC-57 pump.

Detailed descriptions of the 4-component model are given in the references, including an online web page address at which a summary of the latest formulation may be accessed.

DESCRIPTION OF THE EXPERIMENTS

A total of 24 tests were conducted according to a similar methodology, and using a similar range of slurry compositions, as the previous 103 mm (4 in) and 203 mm (8 in) tests, albeit on a larger scale. The pump, sump and sampling system are shown in Figure 2, and a portion of the pipe loop in Figure 3.

The system was driven by a GIW 24x24 TBC-57 pump with 610 mm (24 in) suction, 610 mm (24 in) discharge and 1435 mm (56.5 in) impeller. The system volume, including sump, was approximately 60 m^3 (2120 ft^3). Slurry was returned to the top of the sump to maximize the active slurry volume, and thus increase the inventory of available solids and reduce particle degradation during testing. Flow into the sump was positioned below the water line to avoid introducing air into the slurry.

Figure 2. Showing (a) pump, (b) sump and (c) sampling system for the 489 mm (20 in) pipe loop.

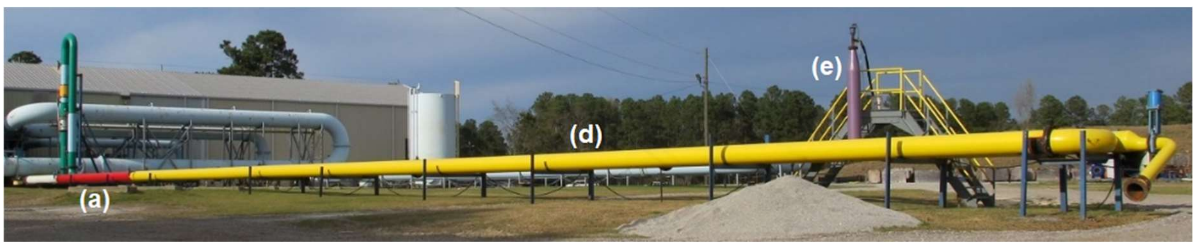

Figure 3. 489 mm system showing: (a) first half of measurement section (red), (b) second half of measurement section - partly hidden, (c) inverted U-loop for SG measurement with magnetic and ultrasonic flow meters, (d) entrance section (yellow), (e) de-aeration stack.

An entrance section of 100 pipe diameters was provided upstream of the friction loss measurement section, which was 50 pipe diameters in length. An additional pressure tap was provided midway in the friction loss section, so that the pressure gradient over the entire section could be compared to the gradients over the first and second half. This was done to identify variations in flow dynamics along the length of the measurement section which might be caused by insufficient entrance length or the development of flow instabilities.

Slurries were "constructed" by combining four individually sourced silica and granite-based products, each with a particle size distribution falling substantially within one of the four component particle size limits, as follows:

- Representing the "Carrier Fluid" fraction ***Xf*****:** A silica "rock flour" with approximately 88% passing 40 µm.
- Representing the "Pseudo-homogeneous" fraction ***Xp*****:** A silica sand product with approximately 90% falling between 40 µm and 0.2 mm.
- Representing the "Heterogeneous" fraction ***Xh*****:** Granite screenings and gravel from a local quarry, with approximately 85% falling between 0.2 mm and 7.3 mm.
- Representing the "Fully Stratified" fraction ***Xs*****:** A screened granite product with approximately 95% larger than 7.3 mm and a top size of 25 mm.

During testing, these fractions were combined in various "blends" from the narrowly graded individual products, through various combinations of two or three or all four together. Volumetric concentrations of the components were selected to maximize the correlations with previous tests. The target test matrix is shown in Table 1. Each consecutive integer in the test matrix indicates a fresh loading of material. Approximately 475 tonnes of solids were used during these experiments.

Table 1. Test matrix showing target percentages for volumetric concentration.

Test	1a	1b	2	3a	3b	4a	4b	5a	5b	5c	5d	6a	6b	6c	6d	6e	7a	7b	7c	7d	7e	8a	8b	8c
Xf																	10	10	10	10	10			
Xp								5	5	10	10	5	10	10	10	10					5	5	7.5	10
Xh				10	10	20	20	10	10	20	20					10				15	15	10	15	20
Xs	5	10	20		5		10		5	5	10			5	10	10		5	10	10	10	5	7.5	10
Total	5	10	20	10	15	20	30	15	20	35	40	5	10	15	20	30	10	15	20	35	40	20	30	40

The actual concentrations and fraction contents achieved during testing varied from the target values, in some cases significantly, due to uncertainties in the loading procedure and the fact that adjustments were not made during running, to minimize run time and solids degradation. Furthermore, even though steps were taken to limit particle degradation, such as loading at velocities below the limit of stationary deposition, particle degradation was significantly higher in the 489 mm pipe loop than seen in the previous 203 mm and 103 mm loop tests. This was due to several factors:

- The loading time was longer due to the larger system volume.
- The relative length of the 489 mm system was shorter and its relative sump volume smaller, compared to previous tests. This was a practical consideration driven by the available space and driver power and resulted in more frequent recirculation of solids.
- The operating velocities during testing were higher, due to the higher deposition velocities seen in the 489 mm pipe.
- The system was less stable during operation, especially when the coarser *Xs* solids were the dominant fraction, requiring longer run time intervals to stabilize operation between data points, and longer averaging intervals for data collection.

In any case, the actual measured concentrations and particle fraction contents based on samples taken during testing were used in the model calculations and analysis, rather than the target values. These are shown in Table 2. Note that all solids concentration values cited in this paper, both measured and modelled, represent delivered (not in-situ) volumetric concentration, as determined by the inverted U-loop.

Table 2. Actual measured fraction content and volumetric concentration.

Test	1a	1b	2	3a	3b	4a	4b	5a	5b	5c	5d	6a	6b	6c	6d	6e	7a	7b	7c	7d	7e	8a	8b	8c
Xf	.07	.07	.12	.04	.11	.12	.15	.04	.07	.10	.12	.05	.04	.05	.10	.10	.74	.55	.49	.25	.25	.14	.11	.17
Xp	.06	.04	.03	.10	.17	.12	.13	.35	.24	.36	.32	.82	.83	.42	.33	.18	.17	.14	.12	.14	.23	.12	.11	.15
Xh	.18	.15	.16	.73	.31	.65	.33	.50	.38	.37	.29	.10	.10	.10	.10	.25	.04	.06	.07	.38	.28	.51	.58	.57
Xs	.69	.75	.69	.12	.41	.12	.39	.11	.32	.17	.27	.03	.03	.43	.48	.47	.05	.25	.31	.24	.24	.23	.19	.12
%Cv	5.7	10.5	21.7	9.4	13.9	18.4	26.6	16.8	21.2	34.4	39.1	3.9	7.4	11.0	15.6	22.3	11.5	13.3	18.2	32.3	34.9	21.9	32.8	40.4

Once loaded, the pipeline velocity was increased to approximately 10 m/s (32.8 ft/s), which represented the maximum practical velocity obtainable in this experimental setup, but also a velocity which is greater than typical operating velocities in industrial slurry systems of this size. All tests were run from the maximum velocity downward until stationary deposition was observed, and sometimes slightly beyond. In some cases, the slurry was bedded out in the pipeline and an additional (often coarser) fraction added for another test. In this way, 2 to 5 different tests were accomplished within a single loading cycle. No cycle was taken through more than 5 loadings before all solids were flushed and the next cycle of tests started with fresh material.

The data collected at each velocity included slurry flowrate (by magnetic induction meter), delivered slurry density (by inverted U-loop as described in Wilson et al., 2006), slurry temperature, pipeline loss section pressure drops (as described above), pump suction, discharge and differential pressures, pump rotational speed and pump shaft power (by strain gauge torque bar transducer). The data were sampled at 100 Hz and averaged over an interval of 10 to 20 seconds, depending on the stability of the system. Whenever possible, data were taken during steady state operation, the exception being at velocities near deposition where the system was inherently unstable. Unstable points near deposition were taken for reference only and were not used in the evaluation of the model. As mentioned above, the system was also sometimes unstable when large fractions of *Xs* were present, especially when *Xh* and *Xp* solids were largely absent. In these cases, the instabilities were usually cyclic and longer data collection times were used to capture a total average value.

A clear section of piping was installed just downstream of the measurement section to allow observation of the flow dynamics and to identify the velocity at the limit of stationary deposition. Unfortunately, this clear section fractured partway through the test program and was replaced with a piece of steel pipe. However, an alternate indication of deposition was provided by measurement of the temperature differential between two actively heated probes on the top and bottom of the pipe. The onset of a stationary deposit altered the heat transfer rate at the bottom of the pipe and created a sudden, observable change in the magnitude of the temperature differential on the order of 0.10 to 0.15 ºC (0.18 to 2.7 ºF). This alternate method compared well with the visual observations available during the first part of the test program and was subsequently used for determination of stationary deposition throughout the remainder of the program.

As with previous tests, slurry samples for measurement of the particle size distribution (PSD) were taken using a device specifically designed for these experiments. During sampling, a pneumatic linkage was used to lift the sump inflow pipe above the water line by about 0.5 m (1.65 ft). A trapezoidal slurry sampler was then passed through the flow stream, cutting through the entire cross section of the 489 mm (20 in) inflow. Sample cutter dimensions and velocities are given in Table 3. A picture of the device can be seen in Figure 4, in position for sampling, but with the inflow pipe not yet raised. It can also be seen in Figure 2, positioned directly beneath the raised inflow pipe. A slurry sample was taken at a pipeline velocity of approximately 7.5 m/s (24.6 ft/s) during each of the 24 tests. This velocity was selected on the criteria of being high enough to ensure good mixing of the various particle sizes, but also representative of the interesting range of velocities for industrial applications.

Table 3. Sample cutter details.

Pipe system	opening mm (in)	Sampler velocity m/s (ft/s)	Slurry velocity m/s (ft/s)	Ave. sample volume litres (gallons)
489 mm	76 x 760	~ 2.0	~ 7.5	~ 22
(20 in)	(3 x 30)	(~ 6.6)	(~ 24.6)	(~ 5.8)

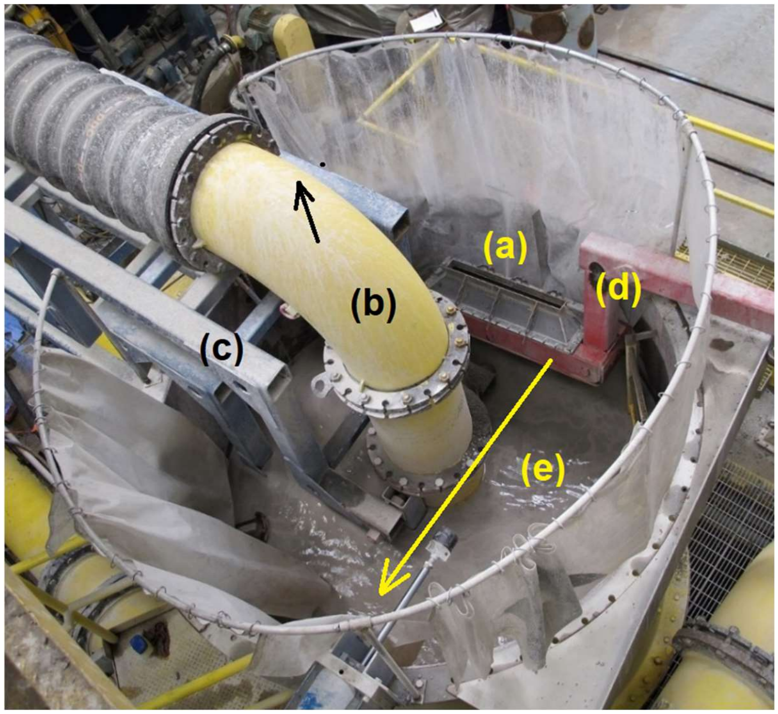

Figure 4. Slurry sampling device showing: (a) sample cutter, (b) return pipe to sump, (c) linkage for lifting return pipe above sump level, (d) linkage for passing sample cutter beneath raised return pipe, (e) slurry surface in sump. Arrows show movement of parts during sample cutting.

The methodology for processing the PSDs was as follows: All >6.3 mm (1/4 in) particles were first cut from the entire sample by a manual wet screening, dried, sieved by size, and weighed. The remaining <6.3 mm solids were dried and weighed in aggregate, then well mixed and split by multiple passes through a sample splitter to produce a representative sub-sample weighing 0.3 to 0.6 kg (0.66 to 1.32 lb). This sub-sample was then wet-sieved by size down to 40 μm, the individual size fractions dried and weighed, and the total weight for each size fraction calculated based on the ratio of the sub-sample to the total <6.3 mm sample weight. Finally, the <40 μm solids, which were collected in the wet sieve runoff, were captured in a filter press, dried and weighed, and the total sample weight of <40 μm solids calculated as above.

Several clear water tests were performed throughout the test program to determine pipe wall roughness, an important parameter for the carrier fluid pressure gradient calculation. Due to the polishing effect of the slurries, this was consistently found to be 0.002 mm (0.00008 in). The clear water tests were also used to establish a baseline water performance for the pump from which the slurry derates would be determined.

ANALYSIS OF THE DATA

Due to the nature of closed loop testing, slurry concentration will change slightly with velocity, and slurry temperature will change with time. Therefore, in comparing the measured data to the predictions of the 4-component models, calculations were made using the individual measured values of flow velocity, slurry concentration, and temperature at each data point.

Pipeline pressure gradient

Figure 5 shows the dimensionless pressure gradient data (i_m), expressed in meters (or feet) of standard water per meter (or foot) of pipe for all 24 tests from the 489 mm loop, where standard water is defined as having a density of 1000 kg/m^3 (62.43 lb/ft^3). This provides a qualitative representation of the scope of the data. Each test is labelled with the dominant particle fractions present and the average total solids concentration during testing at velocities above deposition. The measured PSDs obtained during sampling for these tests are shown in Figure 6. Figure 7 displays the relative error between the measured data and the 4-component model calculation for all data points above stationary deposition. The % error is defined as follows:

$$\%error = 100 \cdot \frac{\left(i_{m,CALCULATED} - i_{m,MEASURED}\right)}{i_{m,MEASURED}} \tag{1}$$

The spread of error for the 489 mm tests exceeded that seen in previous tests, with an average error of 8.0%, compared to 5.4% and 5.5% for the 203 mm and 103 mm tests respectively, as seen in Figure 8. This may be due to the less stable operation of this larger, closed loop system, as mentioned above. However, since the magnitude of the friction losses are smaller in the larger pipe, the average absolute error in the pressure gradient was similar to previous tests, in the range of 0.010 m water/m pipe. Note that the two *Xs+Xf* tests were omitted from Figure 7 and from the error analysis, as there was doubt concerning their validity. This is covered in more detail in Figure 17.

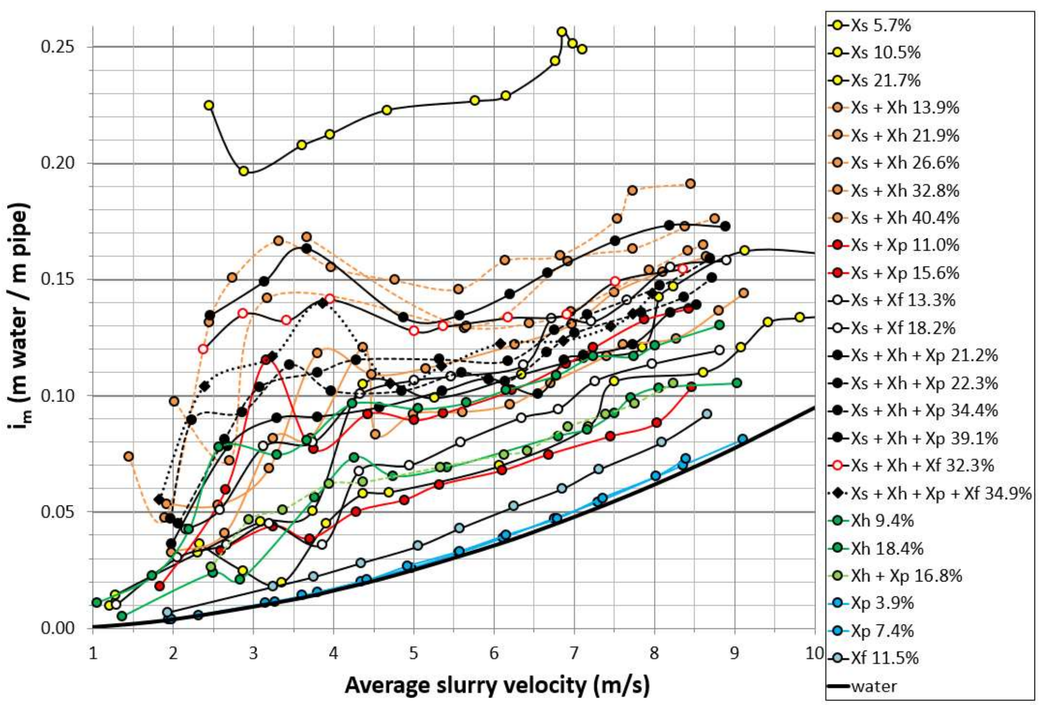

Figure 5. Dimensionless pressure gradient data measured in the 489mm pipe loop.

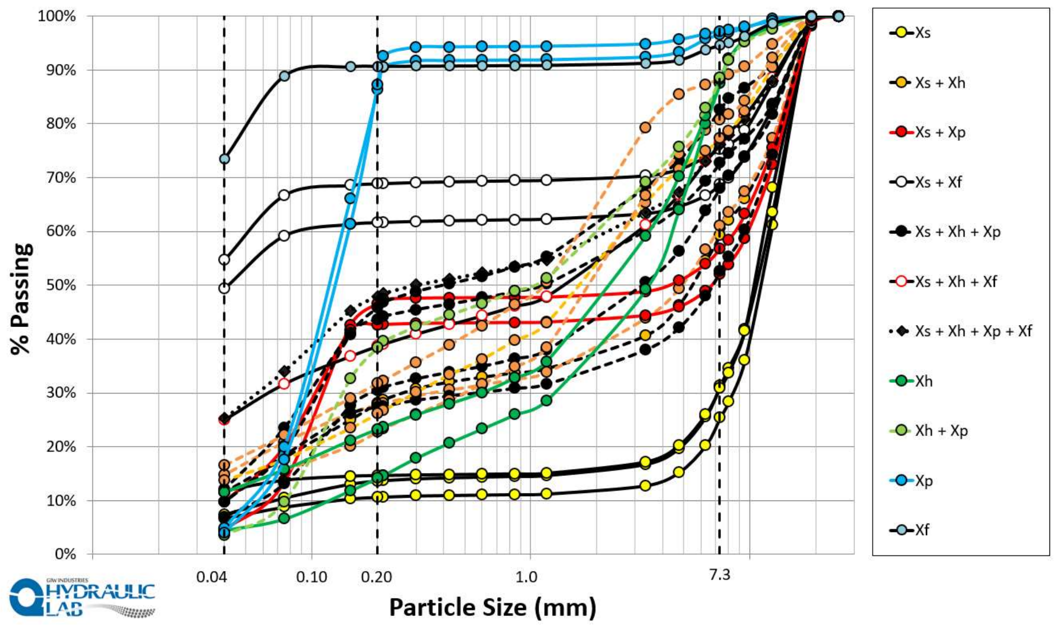

Figure 6. Particle size distributions from test samples in the 489 mm loop.

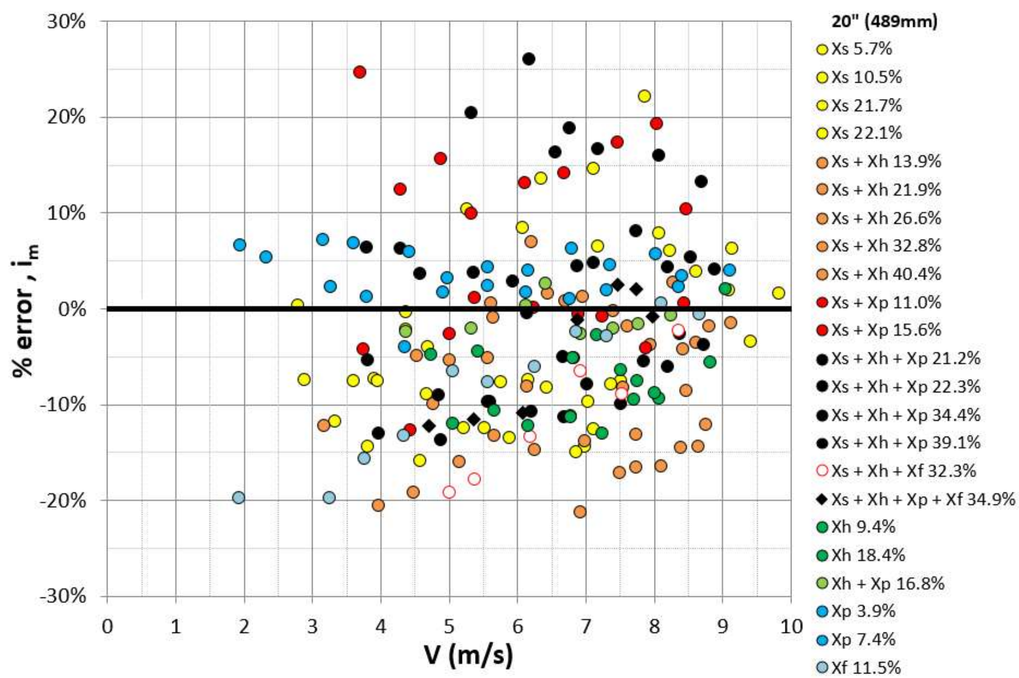

Figure 7. Pressure gradient % error (calculated vs. actual) for the 489 mm pipe.

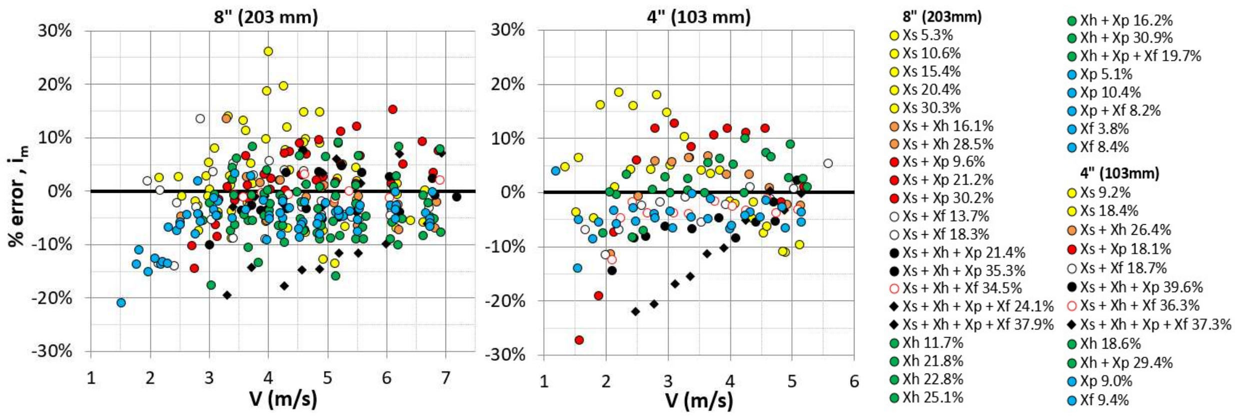

Figure 8. Pressure gradient % error for the previous 203- and 103-mm pipe tests.

The following Figures 9 to 14 provide comparisons of individual test data from the 103 mm, 203 mm, and 489 mm tests with 2.65 S_S solids. Each figure attempts to group tests of similar composition and delivered concentration, however, it should be noted that the correspondence is not exact. Actual component fractions and concentrations are given on each plot, together with the measured water, measured slurry and calculated 4-component dimensionless pressure gradients. As before, the 4-component calculation is made based on the actual conditions measured for each data point.

Figures 9 and 10 show slurries where the single *Xs* or *Xh* components are dominant, resulting in relatively narrow particle size distributions. Volumetric concentrations are near 20%. Note the very visible effect of particle size and pipe size on measured pressure gradient.

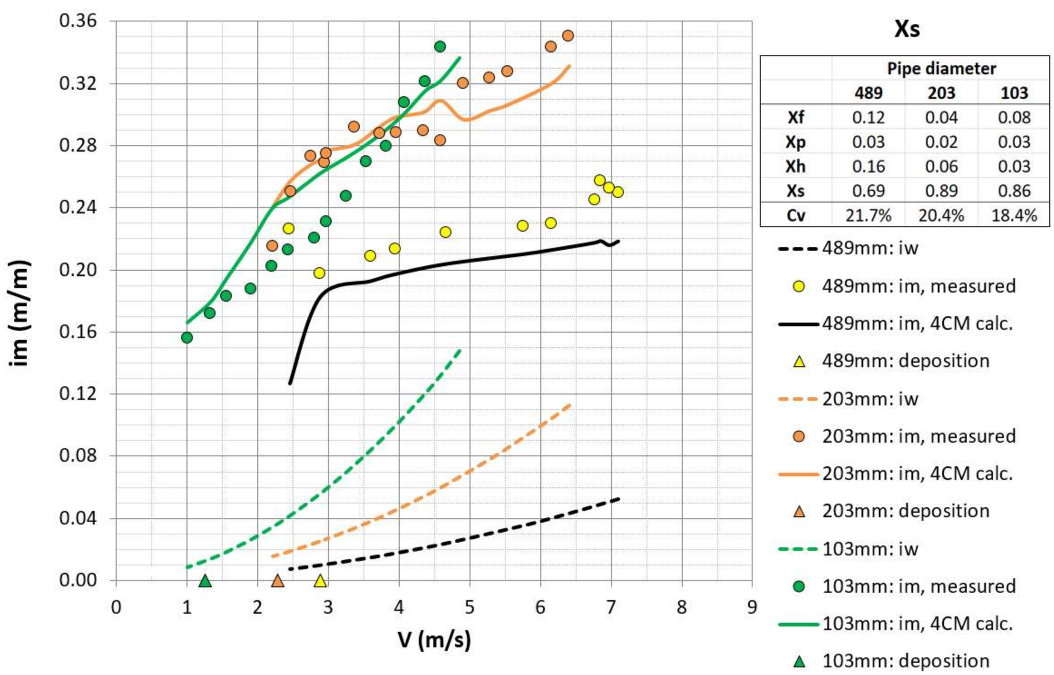

Figure 9. Measured vs. calculated pressure gradients for 489-, 203- and 103-mm pipe loop tests. *Xs* solids dominant, C_v = 18% to 22%

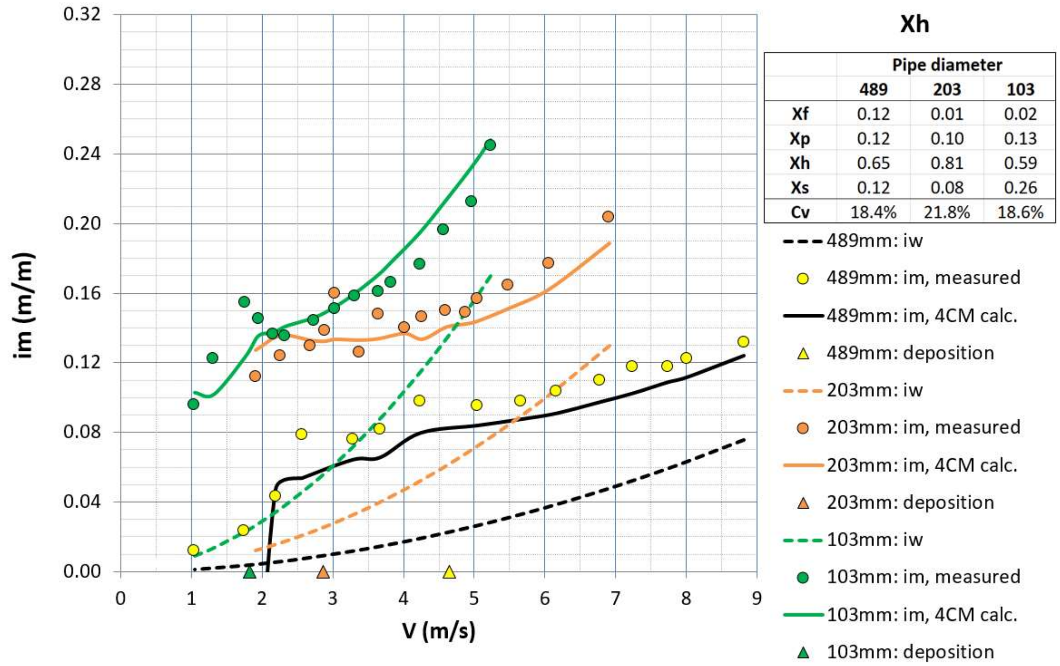

Figure 10. Measured vs. calculated pressure gradients for 489-, 203- and 103-mm pipe loop tests. *Xh* solids dominant, C_v = 18% to 22%

Figures 11 and 12 show mixtures of stratified *Xs* solids combined with heterogeneous *Xh* and pseudo-homogeneous *Xp* solids respectively. Note that the *Xs* + *Xp* mixtures (Figure 12) represent bimodal PSDs, having a relatively small fraction of *Xh* surrounded by larger fractions of *Xs* and *Xp*.

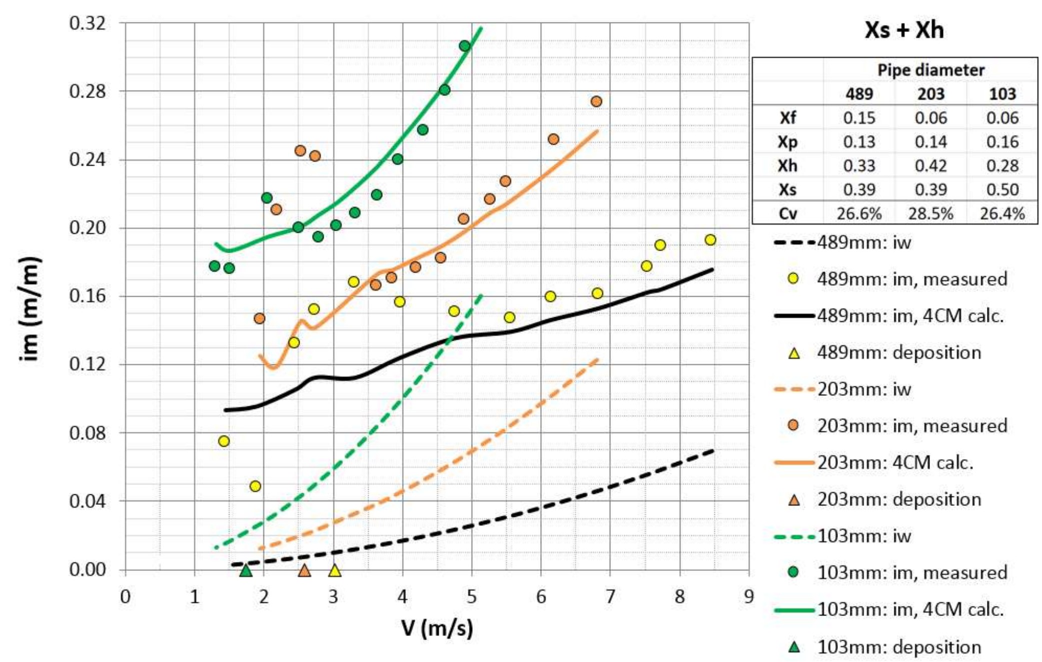

Figure 11. Measured vs. calculated pressure gradients for 489-, 203- and 103-mm pipe loop tests. *Xs* and *Xh* solids dominant, C_v = 26% to 29%

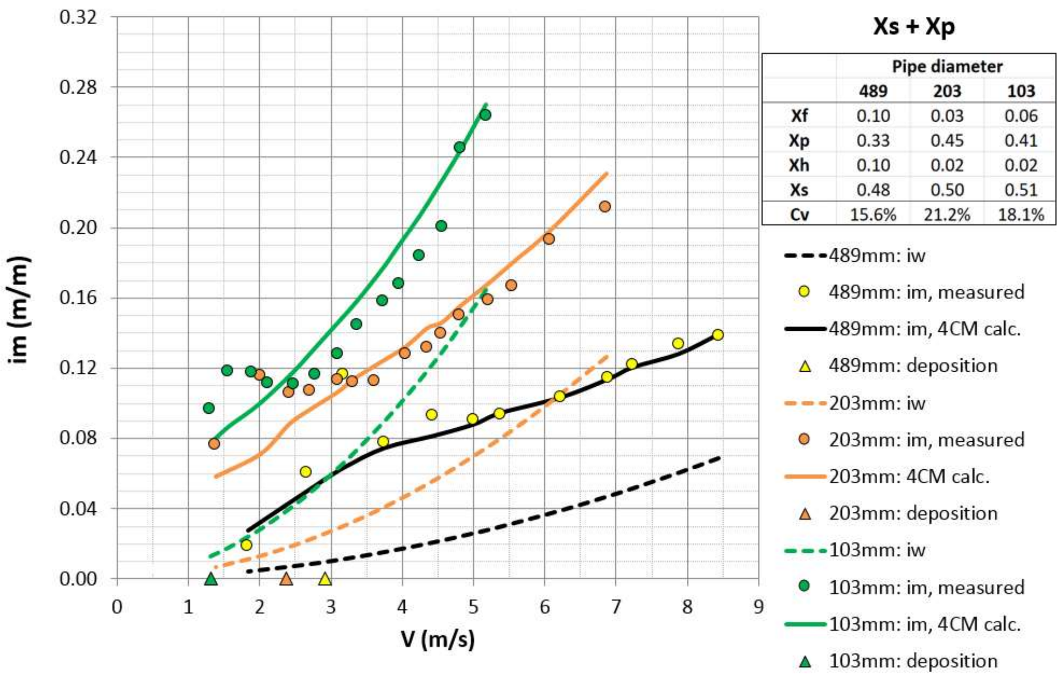

Figure 12. Measured vs. calculated pressure gradients for 489-, 203- and 103-mm pipe loop tests. *Xs* and *Xp* solids dominant, C_v = 16% to 21%

Figures 13 and 14 show mixtures of three and four components respectively at volumetric concentrations near 35%. These tests represent concentrated slurries with broad particle size distributions. Note that the pressure gradients shown in these figures are comparable to the lower concentration, narrow PSD distributions seen in Figures 9 and 10.

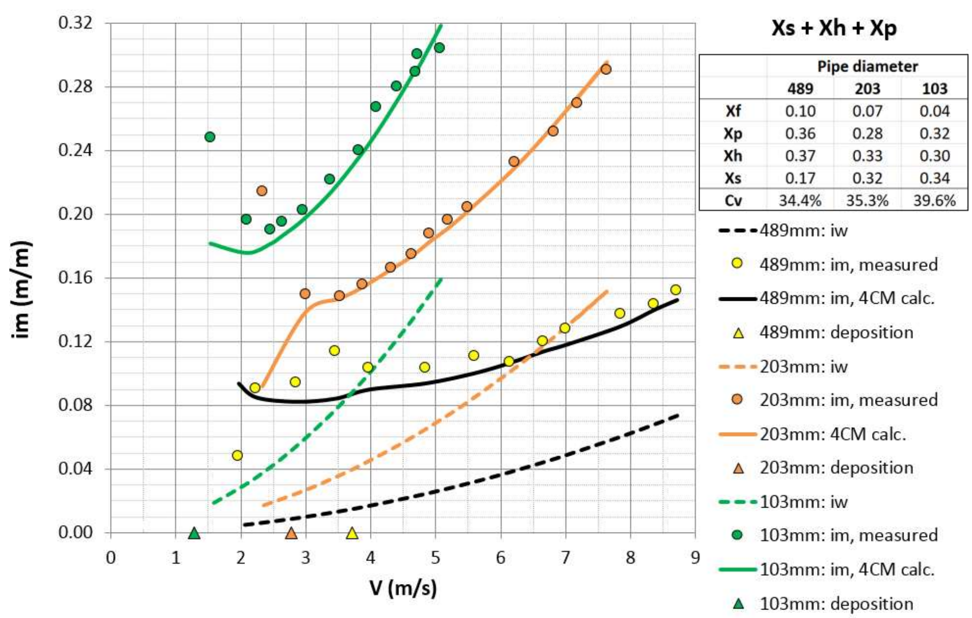

Figure 13. Measured vs. calculated pressure gradients for 489-, 203- and 103-mm pipe loop tests. *Xs*, *Xh* and *Xp* solids dominant. C_v = 34% to 40%

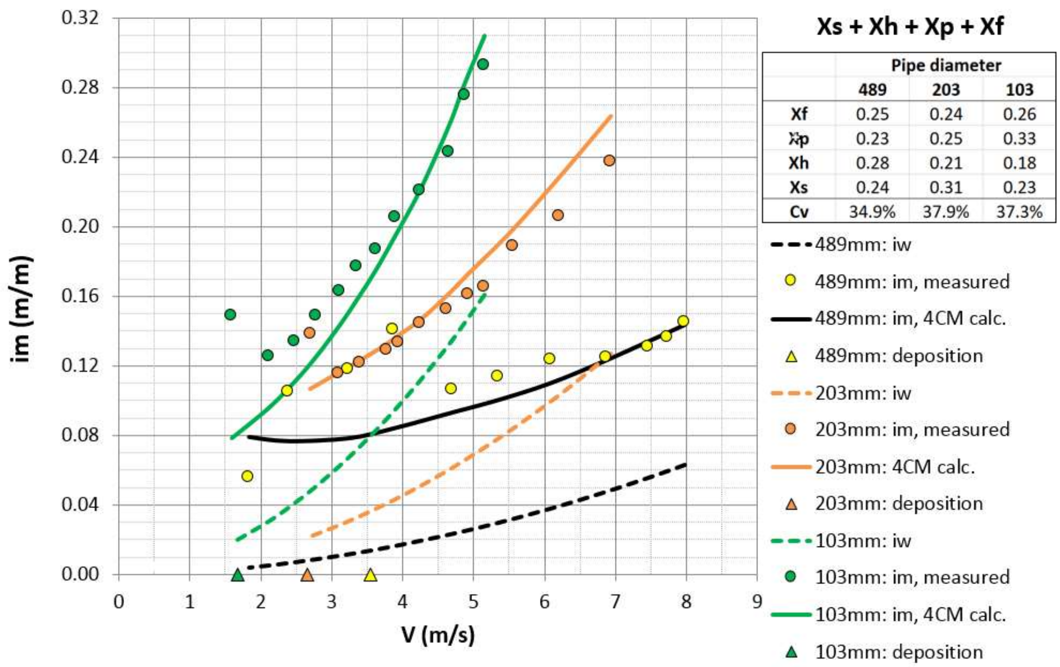

Figure 14. Measured vs. calculated pressure gradients for 489-, 203- and 103-mm pipe loop tests. All solids components represented, C_v = 35% to 38%

Figure 15 compares three tests in the 489 mm pipe loop where the delivered volume concentration of the stratified *Xs* solids is fixed near 10%, while the other fraction concentrations vary, as does the overall solids concentration. In all three tests, the flow contains essentially the same volume of *Xs* grains. In test 6e, the concentration of the heterogeneous *Xh* solids is approximately one half of the *Xs* fraction, and there is also

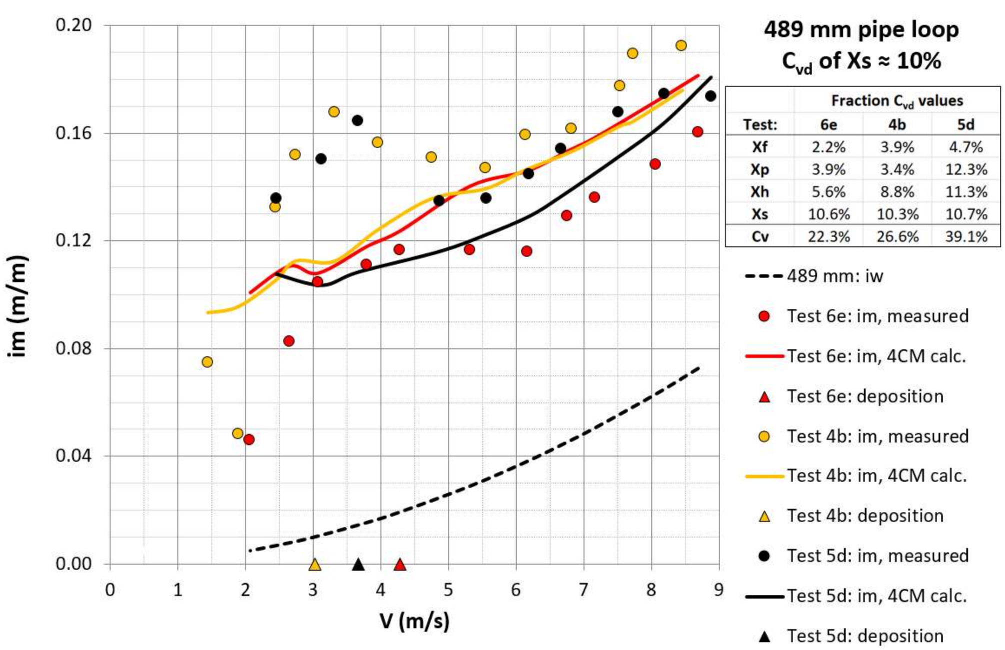

Figure 15. Various 489 mm pipe loop tests where the volume concentration of *Xs* fraction ≈ 10%

a small fraction of the pseudo-homogeneous *Xp* solids. Test 4b differs from 6e primarily by the higher concentration of *Xh* solids, which has increased by 3.2%. This test displays a slightly higher pressure gradient as shown by the measurements plotted in the figure. In test 5d, the concentration of *Xh* solids is further increased by another 2.5%, however, the pseudo-homogeneous *Xp* solids are also increased by 8.9%. Although the overall solids concentration for this test is increased by 12.5% over 29b, the pressure gradient is essentially unchanged, as the additional friction caused by the higher solids concentration is offset by the friction reducing effects of the *Xp* fraction on the *Xh* and *Xs* solids. Although the correspondence is not exact, the 4-component model successfully predicts this trend.

Figure 16 compares three other tests in the 489 mm pipe loop where the delivered volume concentration of the heterogeneous *Xh* fraction solids is fixed near 11%, while the other fraction concentrations and overall solids concentration vary. In test 4a, the concentration of the stratified *Xs* solids and pseudo-homogeneous *Xp* solids are both very low. Test 8a differs from 4a primarily by the higher concentration of *Xs* solids, which is more than doubled from 2.2% to 5%. This produces a higher pressure gradient as confirmed by the measurements. In test 5d, the concentration of *Xs* solids is again more than doubled, being now almost 5 times higher than in the first test 4a. However, the concentration of *Xp* solids has also been increased from 2.7% to 12.3%. Due to friction-reducing effect of the *Xp* fraction, the increase in the pressure gradient between tests 5d and 8a is similar to that seen between 8a and 4a, despite the considerably higher concentration of *Xs* solids and significantly higher overall solids concentration, which increased from 21.9% to 39.1%. The 4-component model successfully predicts this trend.

Figure 17 shows results from the *Xs*+*Xf* mixture tests 7b and 7c. These mixtures contained stratified *Xs* solids in an *Xf* fraction carrier, with minimal pseudo-homogeneous *Xp* and heterogeneous *Xh* solids in between. In previous 203 and 103 mm tests, the *Xf* solids provided friction reducing support for the *Xs* solids in agreement with the predictions of the 4-component model. However, in the 489 mm tests, these mixtures exhibited a higher than predicted loss, with no apparent benefit from the *Xf* solids. In fact, test 1b containing more *Xs* solids that 7c, and having minimal *Xf* solids, showed lower pressure gradients than test

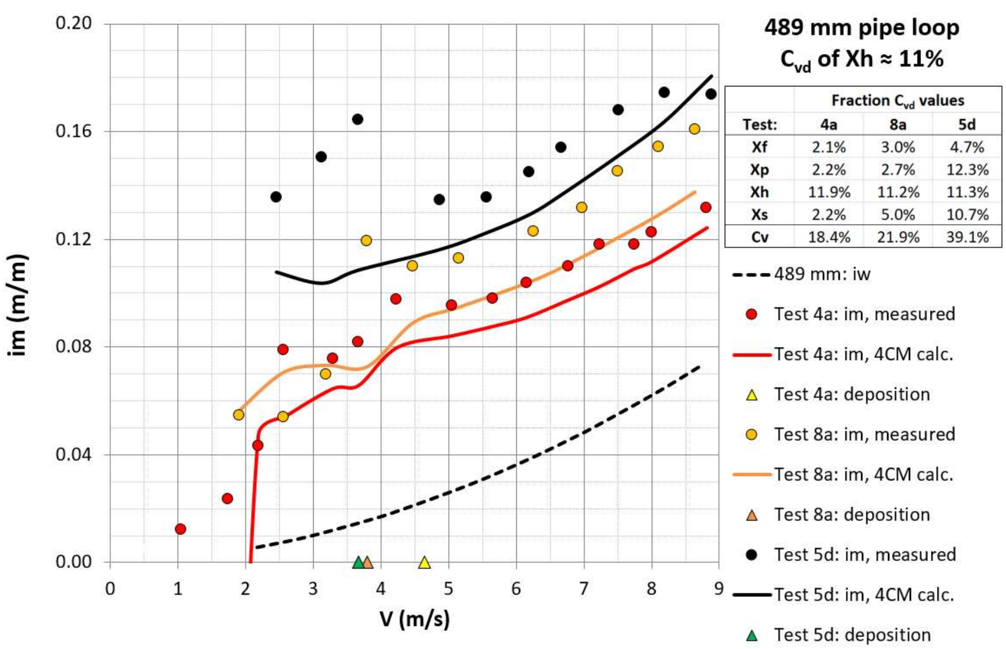

Figure 16. Various 489 mm pipe loop tests where the volume concentration of *Xh* fraction ≈ 11%

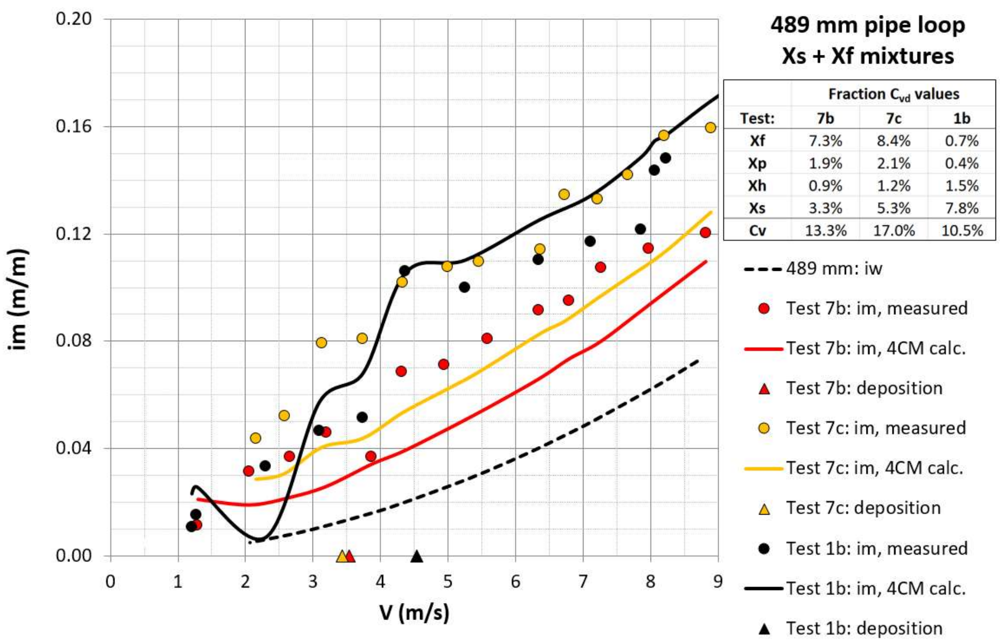

Figure 17. Various 489 mm pipe loop test mixtures of coarse *Xs* in fine *Xf*, with few *Xp* or *Xh* solids.

7c. It is suspected that the measurements taken for these two tests were faulty, although this could not be proven after the fact. Resolution of this questions awaits future repeat testing. In the meantime, the results of these two tests have been discounted as unreliable.

Pump Head Reduction

Flow control during the tests was accomplished by varying pump speed. Speeds ranged from 300 to 410 RPM at the higher pipeline velocities, depending on slurry composition, and decreased to the range of 75 to 200 RPM at the lower velocities. Figures 18 and 19 show the pump speeds and %BEPQ flowrates respectively for the collected data as a function of pipeline velocity, where %BEPQ = percentage of pump flowrate relative to the best efficiency flowrate.

Figure 20 shows the pump head reduction data for all 24 tests from the 489 mm loop and provides a qualitative representation of the scope of the data. Each test is labelled with the dominant particle fractions present and the average total solids concentration during testing at velocities above deposition. The measured PSDs obtained during sampling are the same as those shown previously in Figure 6.

The pump head reduction factor r_h is defined as follows:

$$r_h = \frac{\left(H_w - H_m\right)}{H_w} \tag{2}$$

where H_w is the pump head on water and H_m is the pump head on the slurry mixture.

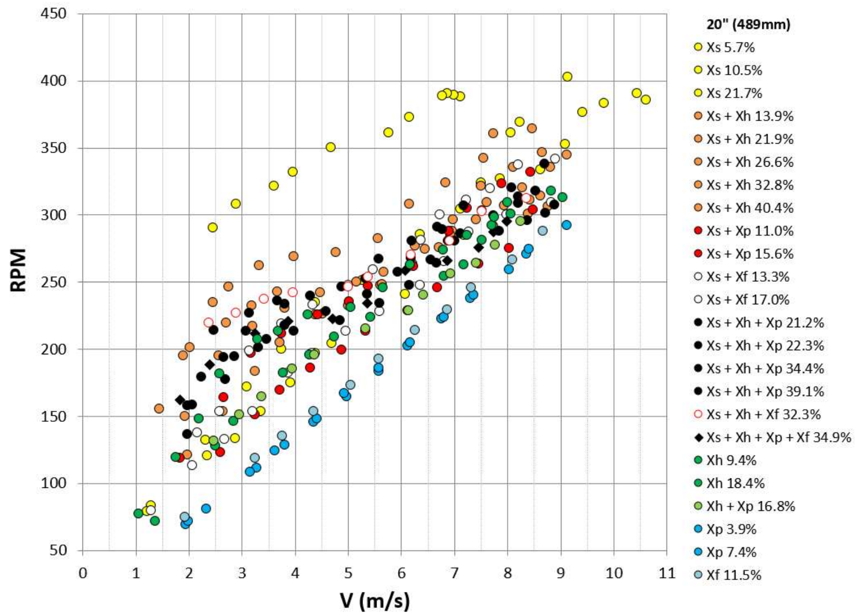

Figure 18. Pump speed measured during testing for the 24x24 TBC 57 pump.

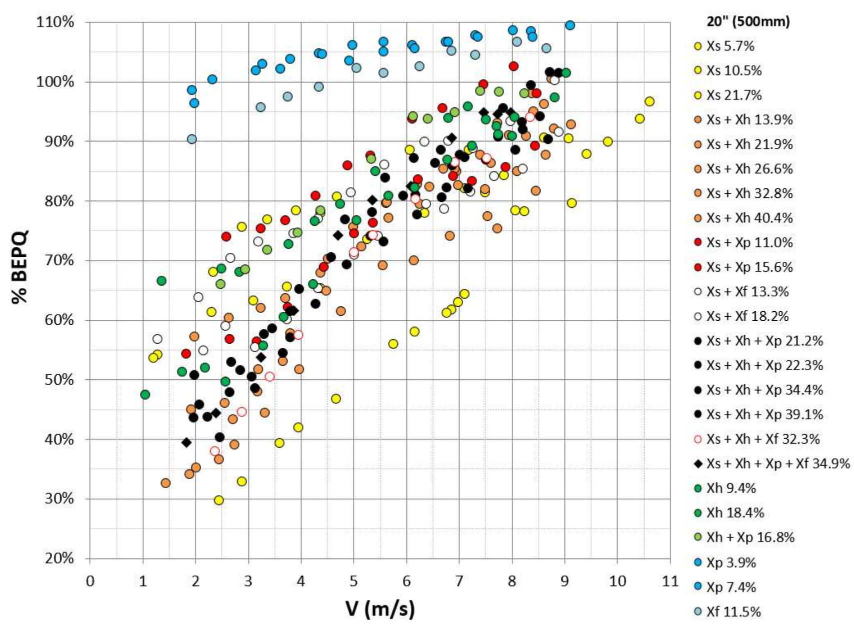

Figure 19. Pump %BEPQ measured during testing for the 24x24 TBC 57 pump.

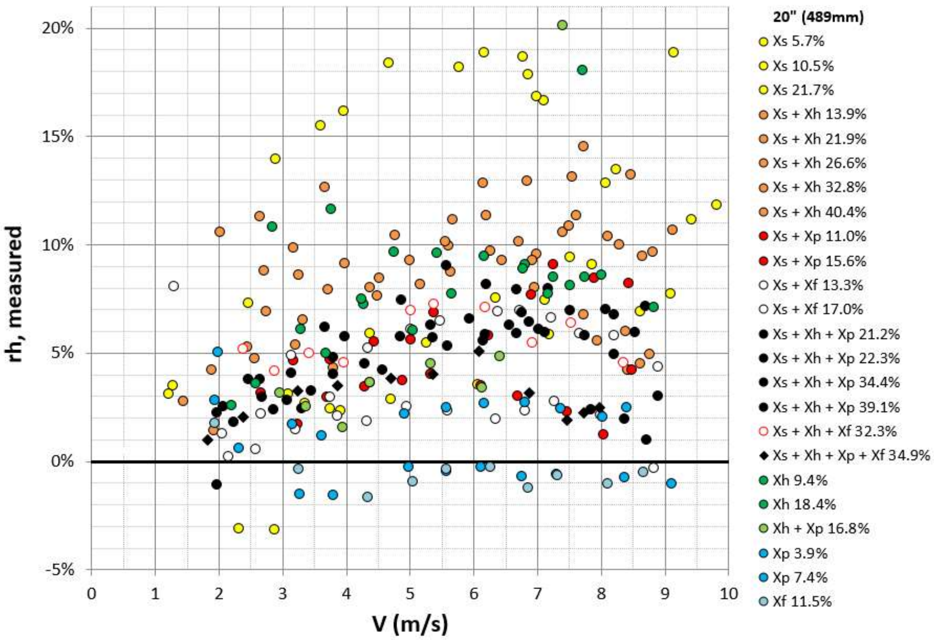

Figure 20. Head reductions measured during testing for the 24x24 TBC 57 pump.

The resulting error between measured and calculated values is expressed as a percentage, but this percentage represents the absolute difference in the head reduction factor, rather than a relative error, as follows:

$$\%error = 100 \cdot \left(r_{h,CALCULATED} - r_{h,MEASURED} \right) \quad (3)$$

A +1% error therefore indicates a calculated head reduction factor which is one percentage point (i.e. 0.01) greater than the measured value. This provides a better context for the analysis, since the head reduction factor is already expressed as a percentage and the absolute error is a better measure of model success in actual application. Using a ratio error calculation for r_h would magnify the perceived error where the derate effects are small and skew a proper understanding of the results.

As mentioned above, tests conducted on higher density 4.75 S_S solids showed a velocity (flowrate) dependence not previously seen with 2.65 S_S solids in the 103 mm and 203 mm pipe tests. This resulted in the introduction of the three empirical parameters A'', B'' and C'' already used in the pipe friction model. While a similar velocity dependence is not seen in the present 20" tests, there was a significant impact of using the empirical parameters to account for the action of finer solids on reducing the pump performance derate contribution of the coarser solids.

Figure 21 shows the error analysis for the 24x24 TBC 57 tests using the 4-component model. In the left hand plot, the empirical parameters A'', B'' and C'' have been omitted. The errors are largely unaffected by changes in flowrate or %BEPQ and fall mostly within a band of 8 head reduction % points, with an average value near zero. This spread of error is larger than seen in the previous tests with smaller pumps, which were generally within a range of 5 head reduction % points. In addition, there is a clear difference between tests where stratified *Xs* solids predominate (yellow shaded points), compared to those containing broader 3- and 4-component mixtures of *Xs* solids together with heterogeneous *Xh* and pseudo-homogeneous *Xp* fractions (solids black points). The model under-predicts losses for the former and over-predicts for the latter, both by about 5%. A similar spread in error is seen between these slurry types if the mono-sized method defined in ANSI/HI 12.1-12.6-2016 is applied, although in that case the results are shifted downward, and head reductions for the single component *Xs* slurries are under-predicted by up to 10%. This observation indicates that some degree of friction reducing support is being provided to the coarse solids by the finer solids. Reintroducing the empirical parameters as proposed in Visintainer et al. (2021) gives the error result seen in the right side plot of Figure 21, with the largest improvements seen for the mixtures containing coarse solids, both with and without fines. Even so, slightly larger errors are seen in these tests than in the previous tests with smaller pumps. Whether this is a byproduct of the difficulties encountered with the test system stability in this large pipe size, or is due to some other dynamic, is not known.

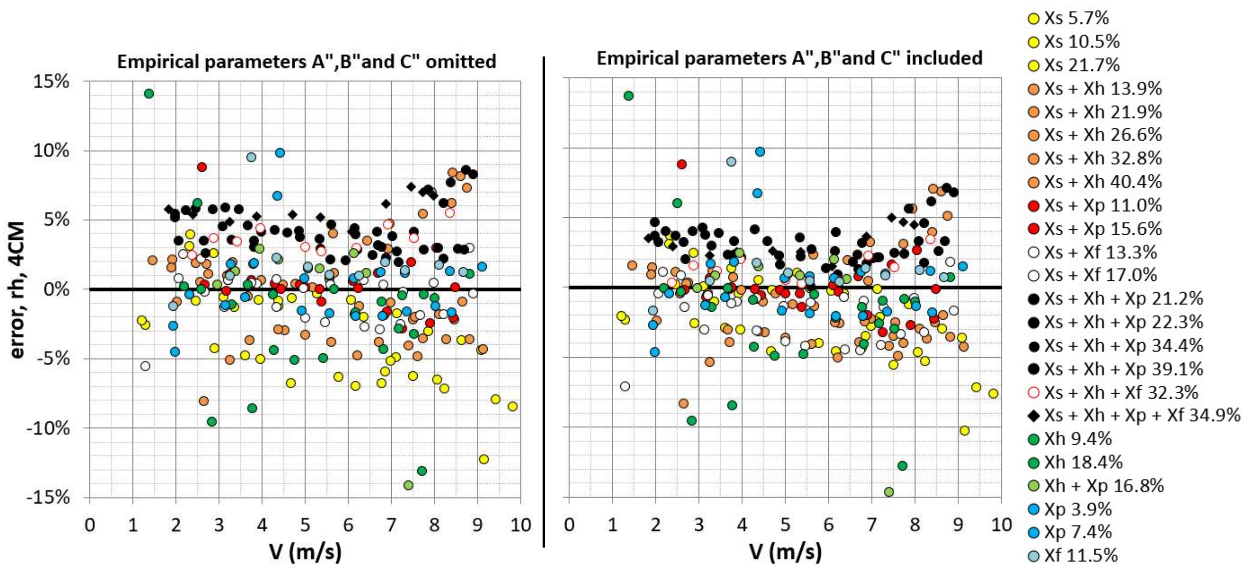

Figure 21. Head reduction %error for the TBC 57 pump.
Empirical parameters *A*", *B*" and *C*" omitted at left, included at right.

The following Figures 22 to 25 provide comparisons of individual test data from the 103 mm, 203 mm, and recent 489 mm tests. Each figure attempts to group tests of similar slurry composition and volumetric concentration, however, it should be noted that the correspondence is not exact. The actual component fractions and volumetric concentrations are given for each test, together with the measured water, measured slurry and calculated 4-component pump head performance according to the new formulation. As before, the 4-component calculation is made using the actual conditions measured for each data point. Note that the total head will vary according to the system resistance, which tends to decrease with pipeline size.

Figure 22 shows slurry mixtures where the stratified *Xs* solids are dominant at a total concentration of approximately 20% by volume. In Figure 23, both *Xs* and heterogeneous *Xh* solids are present, as well as a smaller portion of pseudo-homogeneous *Xp* solids. The pipeline pressure gradient and pump head reduction are both correspondingly lower than in Figure 22, even though the total volumetric concentration of these mixtures is higher at around 27% by volume.

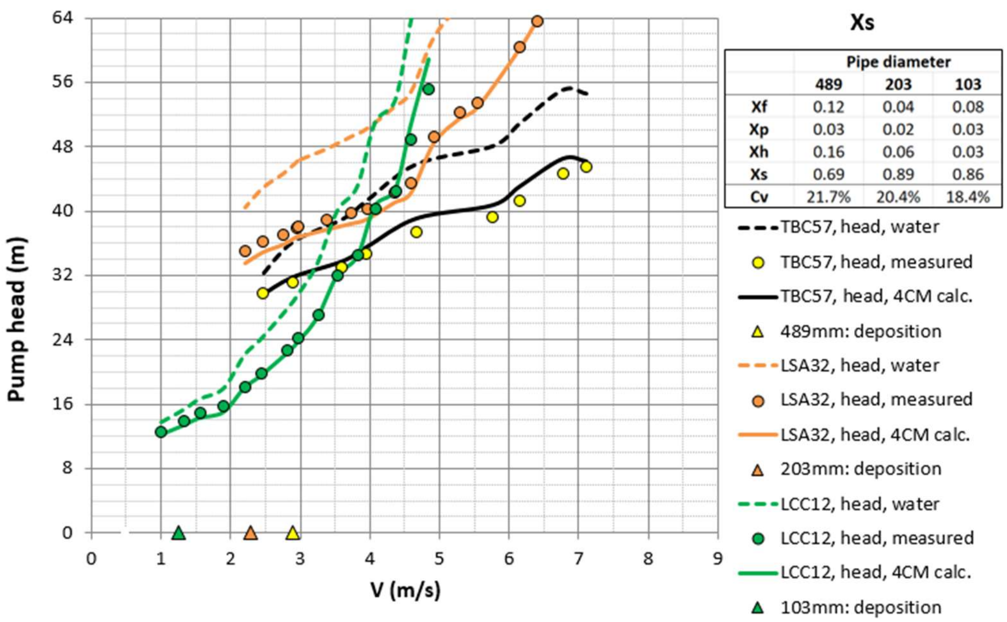

Figure 22. Measured vs. calculated pump head reduction for 489-, 203- and 103-mm pipe loop tests. *Xs* solids dominant, C_v = approx. 20%

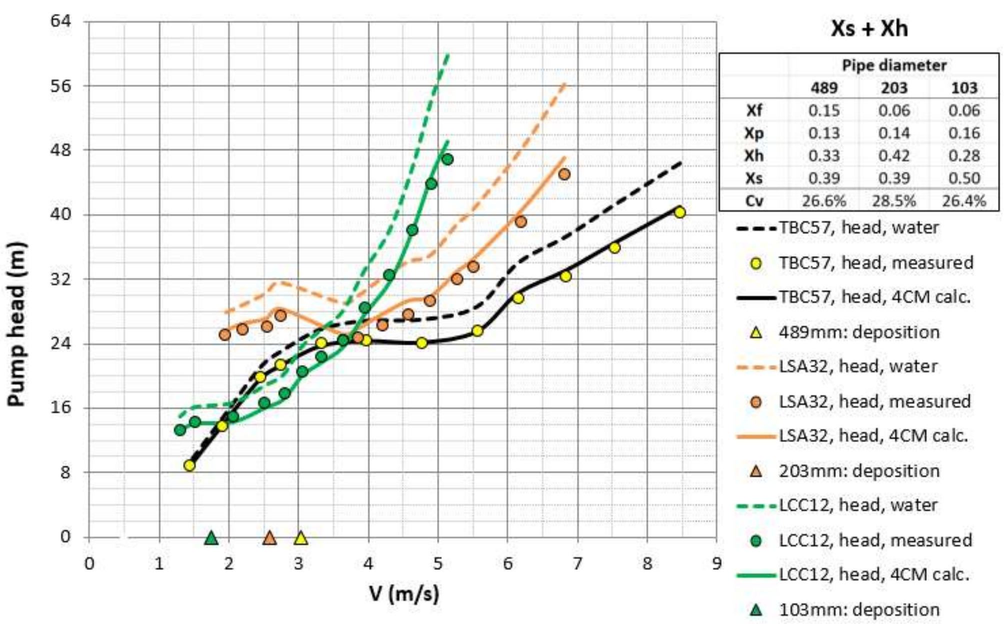

Figure 23. Measured vs. calculated pump head reduction for 489-, 203- and 103-mm pipe loop tests. *Xs* and *Xh* solids dominant, C_v = approx. 27%

Figure 24 shows mixtures of stratified *Xs* solids with pseudo-homogeneous *Xp* solids. These represents bimodal PSDs, having a relatively small fraction of heterogeneous *Xh* solids bounded by larger fractions of *Xs* and *Xp*. Note that while these mixtures average around 18% in volumetric concentration, they have a similar volumetric concentration of *Xs* solids to the slurries in Figure 23, about 10%, and a correspondingly similar pipeline friction loss. However, the mixtures of Figure 24, with a higher proportion of *Xp* solids and lower overall concentration show a significantly lower pump head reduction.

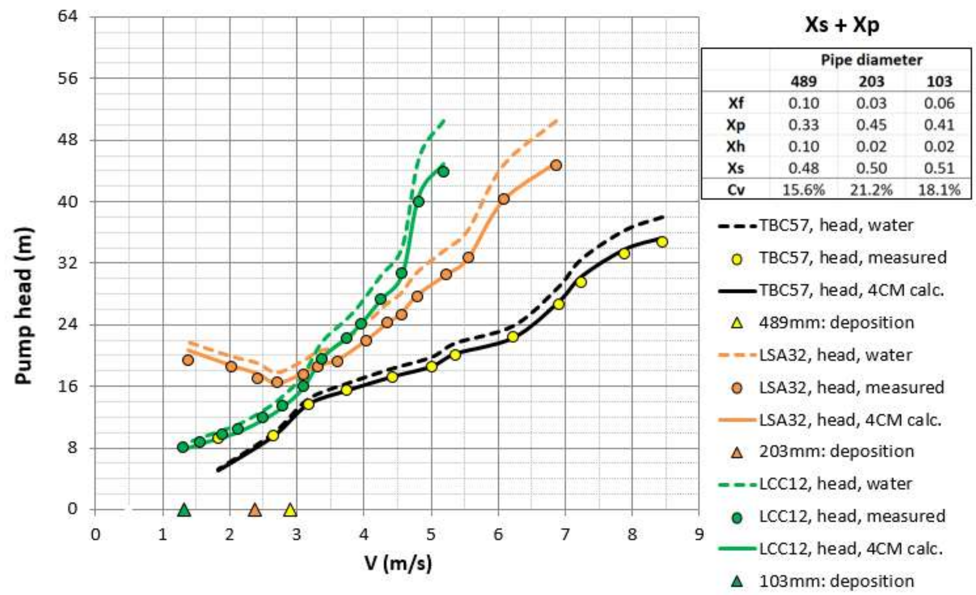

Figure 24. Measured vs. calculated pump head reduction for 489-, 203- and 103-mm pipe loop tests. *Xs* and *Xp* solids dominant, C_v = approx. 18%

Figure 25 shows mixtures with all 4 components present in roughly equal quantities. Although the delivered concentration is high, the broad particle size distribution allows the mixture to act very much like an equivalent fluid as far as pump performance is concerned, with very little difference between the pump head on water or slurry.

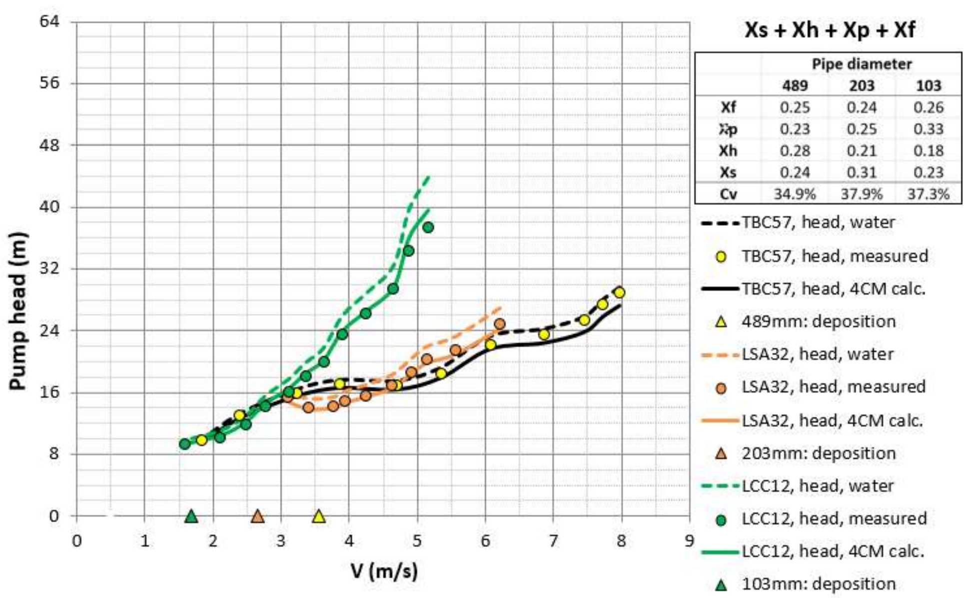

Figure 25. Measured vs. calculated pump head reduction for 489-, 203- and 103-mm pipe loop tests. All components present, C_v = approx. 36%

In all the above cases, the revised 4-component model for centrifugal pump head reduction corresponds well to the measured pump performance.

CONCLUSIONS

A comprehensive data set spanning multiple settling slurry flow regimes has been collected to validate the previously described 4-component models for pipeline friction loss and centrifugal slurry pump head reduction (derate) in pipelines up to 489 mm.

Tests carried out in the 489 mm pipeline experienced higher particle degradation and less operating stability than previous similar tests in smaller pipelines.

Two bi-modal tests of stratified Xs solids in Xf-fraction carrier exhibited unexpectedly high pipeline friction losses. It could not be determined if this was the result of a faulty measurement, or an unexplained behaviour of the mixture in the larger pipe size. Additional tests will be needed to resolve this question.

Excluding the two tests mentioned above, the average error between measured and calculated values for pipeline friction loss in the 489 mm pipe loop tests was 8.0%, compared to 5.4% and 5.5% for the previous 203 mm and 103 mm tests. However, since the magnitude of the losses are smaller in the larger pipe, the average absolute error was similar to previous tests, in the range of 0.010 m water/m pipe.

Test containing coarse (stratified) Xs fraction solids showed a higher deviation between measured results and model predictions for pump head reduction than seen in previous tests with smaller pumps.

Unlike previous tests on smaller pumps handling 2.65 S_S solids, but in agreement with more recent tests on smaller pumps handling 4.75 S_S solids, the introduction of the 4-component empirical parameters A'', B'' and C'' that account for interactions between the various particle size fractions were required for best accuracy of the pump head reduction model. Without these parameters, head reductions for mixtures containing stratified Xs solids with few fines were under-predicted, while those with a larger content of fine solids were over-predicted.

The current formulation of the 4-component model provides reasonable predictions for both pipeline friction loss and centrifugal pump head reductions across the range of tested pipelines (103 to 489 mm).

REFERENCES

ANSI/HI 12.1-12.6-2016, *American National Standard for Rotodynamic Centrifugal Slurry Pumps*, Hydraulic Institute, Parsippany, NJ, USA, 50-53.

Sellgren, A., Visintainer, R., Furlan, J., Matousek, V. (2014). "Pump and pipeline performance when pumping slurries with different particle gradings". *Proceedings of 19th International Conference on Hydrotransport,* Colorado, USA.

Shook, C.A., Gillies, R.G., Sanders, R.S. (2002). "Pipeline Hydrotransport with Applications in the Oil Sand Industry", *SRC Publication No. 11508-1E02*, SK, Canada.

Visintainer, R., Furlan, J., McCall, G., Sellgren, A., Matousek V. (2017a). "Comprehensive loop testing of a broadly graded (4-component) slurry", *Proceedings of 20th International Conference on Hydrotransport*, Melbourne, Australia.

Visintainer, R., Sellgren, A., Furlan, J., McCall, G. (2017b). "Centrifugal pump performance deratings for a broadly graded (4-component) slurry, *Proceedings of 18th International Conference on Transportation and Sedimentation of Solid Particles*, Prague, Czech Republic.

Visintainer, R., Sellgren, A., Matousek V., McCall, G (2021). "Testing and modelling of diverse iron ore slurries for pipeline friction and pump head derate", *Proceedings of Iron Ore Conference 2021*, Perth, Australia.

Wilson, K. C., Sellgren, A. (2001). "Hydraulic transport of solids", *Pump Handbook*, 3rd edition, pp. 9.321-0.349, McGraw Hill, New York.

Wilson, K.C., Addie, G.R., Sellgren, A., Clift, R. (2006). *Slurry Transport Using Centrifugal Pumps*, 3rd edition, Springer, New York.

CITATION

Visintainer, R., McCall II, G., Sellgren, A. and Matoušek, V. "Large Scale, 4-Component, Settling Slurry Tests for Validation of Pipeline Friction Loss and Pump Head Derate Models", *WEDA Journal of Dredging*, Western Dredging Association, Temecula, CA, USA.

DATA AVAILABILITY

The dataset generated during the described testing is proprietary to GIW Industries. Inc. The details of the 4-component model are available online at: https://bit.ly/4component.

ACKNOWLEDGEMENTS

The authors would like to acknowledge the contribution of GIW Industries Inc. and the GIW Hydraulic Lab engineers and staff in providing the supporting data for this study.

IMPACT OF STRATEGIC, UNCONFINED, DREDGED MATERIAL PLACEMENT ON TURBIDITY WITHIN A SHALLOW BACK BAY SYSTEM: OBSERVATIONS FROM SEVEN MILE ISLAND INNOVATION LABORATORY, NJ

Kelsey Fall[1], David Perkey[1], Lenore Tedesco[2], and Monica Chasten[3]

ABSTRACT

Near-marsh, shallow-water strategic placement of sediment dredged from navigation channels is a promising method for increasing marsh accretion rates and providing erosion protection to marsh edges. A significant challenge for unconfined sediment placement in near-marsh, shallow water areas is the concern related to the degree of and persistence of turbidity, both during and following placement. The objective of this study was to document turbidity during and following unconfined sediment placement near and on a marsh. Observations were collected during and following a strategic placement activity within the Seven Mile Island Innovation Laboratory (SMIIL), a living laboratory launched by the US Army Corps of Engineers, in conjunction with the State of New Jersey and The Wetlands Institute (TWI) to evaluate beneficial use of dredged material management practices in coastal New Jersey. In 2020, a field effort utilizing roving turbidity surveys conducted during and following placement was combined with a two-month deployment of a near-bed sensor to characterize turbidity associated with unconfined placement near and on the southern end of Gull Island, a low marsh island within the SMIIL system. Roving turbidity surveys found the resulting turbidity plume was localized, only extending about 20 m offshore and 100 m along shore, and that during calm conditions (wind speeds <5 m/s), plume direction and intensity were driven by tidal circulation. Monitoring showed that near-bed turbidities during active placement were greater than typical background conditions but were often less than those observed during high wind or storm events. During post placement surveying the week following the completion of dredging, turbidity in the region was observed to be similar to levels documented in an earlier study of the region, prior to any placement.

Keywords: Dredging; Nearshore, shallow water placement; water quality monitoring; beneficial use; regional sediment management.

INTRODUCTION

Marsh-dominated back bay systems are a dominant setting in some of the most densely populated portions of the Atlantic and Gulf coasts. The value of these natural features for shoreline protection of back bay and coastal communities has been well documented with tidal wetlands being some of the most valuable (Arkema et al., 2013, Peteet et al., 2018). Unfortunately, many east coast tidal wetlands are showing evidence of decline due to the combined effects of coastal development, climate change, and accelerating sea level rise rates which are resulting in modified sediment transport patterns (Bertness et al., 2002, Church and White, 2012, Hartig et al., 2002). In the mid-Atlantic region, these relative sea level rise (SLR) rates

---

[1] Engineer Research and Development Center, Coastal and Hydraulics Laboratory, US Army Corps of Engineers, Vicksburg, MS 39180.

[2] The Wetlands Institute, Stone Harbor, NJ, 08247

[3] Operations Division, Philadelphia District, US Army Corps of Engineers, Philadelphia, PA 19107

are higher than surrounding coastal regions (Miller et al., 2012). It was estimated that sea level rose at a rate of 0.2-0.3 inches/year from 2007-2020 (Sweet et al., 2022) and it is projected to rise between 0.3 and 1.3 ft between 2010 and 2030 (Kopp et. al., 2019), further exacerbating marsh vulnerability. With marsh resilience intricately tied to a host of accretionary and erosional processes that are altered under anthropogenic activity and accelerating sea level rise, the long-term fate of marshes is receiving significant attention.

Increases in rates of SLR and frequency of coastal flooding in recent decades have helped bring about a change in perception regarding dredged sediment in coastal areas. The availability of suspended mineral sediment has been identified as one of the major factors influencing wetland development, geomorphology, and response to relative SLR (Kirwan and Murray, 2007; Day et al., 2011, Mudd, 2011, Fagherazzi et al., 2012, Ganju et al., 2017). Many marshes and mudflats will require anthropogenic support to enhance resilience against SLR. A critical component for addressing SLR is increasing mineral sediment input to marshes. Instead of viewing dredged sediments as a waste to dispose of, many coastal managers are seeking to emphasize Regional Sediment Management (RSM, https://rsm.usace.army.mil/) practices that utilize clean dredged material as resources for beneficial use across various coastal environments for both storm protection and environmental restoration.

Near-marsh, shallow-water strategic placement of sediment dredged from navigation channels is a promising method for increasing marsh accretion rates. Here, strategic placement is defined based off the description in Gailani et al., 2019, as the process in which dredge material is either directly placed to form a permanent or semi-permanent feature or in which dredged material is placed in a manner in which sediments can be redistributed by natural forces. Both aim to place material such that results will assist in reducing flood risks, support navigation or encourage and strengthen environmental restoration missions. However, questions remain as to the impact of turbidity and other unintended consequences of sediment placement. A significant challenge for unconfined sediment placement in near-marsh, shallow water areas is concern related to the degree of and persistence of associated turbidity. Degradation of water clarity associated with high turbidity is a key water quality management issue. The quantity and quality of underwater light drives primary productivity, governs the growth and survival of essential habitats, and controls the visual appearance of the water, influencing property values and tourism via aesthetic appeal. High turbidity can be particularly detrimental in shallower back bay environments which support a variety of submerged aquatic vegetation, and provide critical habitat to a variety of fish, shellfish, and crabs that are an integral part of the estuary food chain as well as supporting recreational and commercial fisheries The objective of this study is to document turbidity resulting from unconfined sediment placement in nearshore areas and on marshes from beneficial use projects designed to enhance marsh resilience.

STUDY SITE: GULL ISLAND, SEVEN MILE ISLAND, NJ

Observations were collected during a strategic placement activity within the Seven Mile Island Innovation Laboratory (SMIIL), a living laboratory in coastal New Jersey (Figure 1A). Like many other wetlands worldwide, marshes in this area are showing signs of degradation, are vulnerable to impacts from rising seas and storms, and could greatly benefit from improved RSM practices that utilize dredged material as a sediment resource. In 2019, the US Army Corps of Engineers (USACE), Philadelphia District (NAP), in conjunction with USACE's Engineering Research and Development Center (ERDC), the State of New Jersey, and The Wetlands Institute (TWI), launched SMIIL to evaluate innovative dredged material management practices and alternatives in coastal New Jersey with the goal to advance and improve dredging and marsh restoration techniques through research, collaboration, knowledge sharing, and practical application.

The SMIIL is defined geographically within the coastal region between Townsends Inlet at the north end of Avalon to Hereford Inlet at the south end of Stone Harbor, Cape May, New Jersey (Figure 1A, B) and includes both the back bay and oceanfront areas. The back bay area encompasses approximately 61 sq. km (24 sq. miles) of tidal wetlands, coastal lagoons, tidal channels, and bays extending from the barrier island (Seven Mile Island) to the mainland. Water depths are typically less than 1 m at Mean Lower Low Water (MLLW=-0.73 m NAVD88). Tidal conditions are characterized as mixed semidiurnal with a tidal range of approximately 1–2 m. Preliminary monitoring of hydrodynamics, turbidity, and total suspended solids at three stations from October to December 2019 in the central SMIIL area (Fall et al., 2021) found that apart from punctuated wind events, the area is generally calm, and waters are clear. Fall et al., (2021) found the area generally experiences small waves, <0.25 m, slow current speeds (~0.1 m/s), low turbidity (~10-20 NTUs), and low suspended sediment concentrations (~10–20 mg/L). While higher energetic conditions produced waves with heights of ~ 0.5-0.75 m, currents ~ 0.4 m/s and turbidities between ~ 100-200 NTUs (Fall et al., 2021).

In 2019 and 2020, NAP and TWI partnered with ERDC's Coastal and Hydraulics Laboratory (ERDC-CHL) and undertook a series of beneficial use projects on Gull and Sturgeon Islands, two islands in the back barrier system, to address marsh and wading bird colony vulnerability. During this period, NAP contracted with Barnegat Bay Dredging Company of Harvey Cedars, N.J. to dredge approximately 50,000 m^3 of sediment from a critical shoal in the New Jersey Intracoastal Waterway (NJIWW) federal channel to be placed in various areas on and in the nearshore of these two islands to drive habitat restoration, increase marsh plain elevation, and enhance marsh edge protection.

This field effort monitored turbidity from the 2020 placement of material near and on Gull Island (Figure 1B). From September-October the dredge *FULLERTON*, a cutterhead dredge, hydraulically pumped a mix of fine sand and muddy sediment from the NJIWW to two placement areas, referred to as *A* and *B*, using a Y-valve setup to split direct placement (Figure 1C). Analysis of sediment samples collected from the NJIWW had a fines content (d> 63 μm) of greater than 50% with a d_{50} of about 40 μm. At *A,* dredged material was pumped freely directly onto the marsh (Figure 2A). The goal of this placement was to lift this area of low marsh and un-vegetated mud flat to more resilient marsh elevations. At *B,* material was pumped from a floating discharge pipe along the southern edge of Gull Island (Figure 2B). The goal of this placement was to use dredged material to build a berm/bar for added protection of the eroding marsh edge along south Gull Island. The discharge pipe was set just off the marsh edge on floats so that it could be moved, as pumping commenced, to build a larger feature. At both locations, active placement occurred during daylight hours for about 8-10 hours/day.

METHODS

Monitoring During Dredged Material Placement

A week-long field effort was completed to characterize turbidity during the unconfined, open water placement. To monitor turbidity, roving surveys were conducted throughout the area using an In-Situ AquaTROLL 600 Multiparameter sonde (In-Situ) or an YSI EXO2 Multiparameter Water Quality Sonde (YSI), both equipped with optical turbidity sensors. The sondes were pole mounted and deployed over the side of a vessel, collecting observations roughly 0.4 m–0.6 m below the surface. This was roughly in the mid to lower water column depending on tidal phase. Due to vessel depth restrictions, turbidity surveys were restricted to the 3–4 hrs before and after high water. For the surveys, vessels maneuvered at dead slow speed (~1–2 knots) to prevent any vessel-induced turbidity.

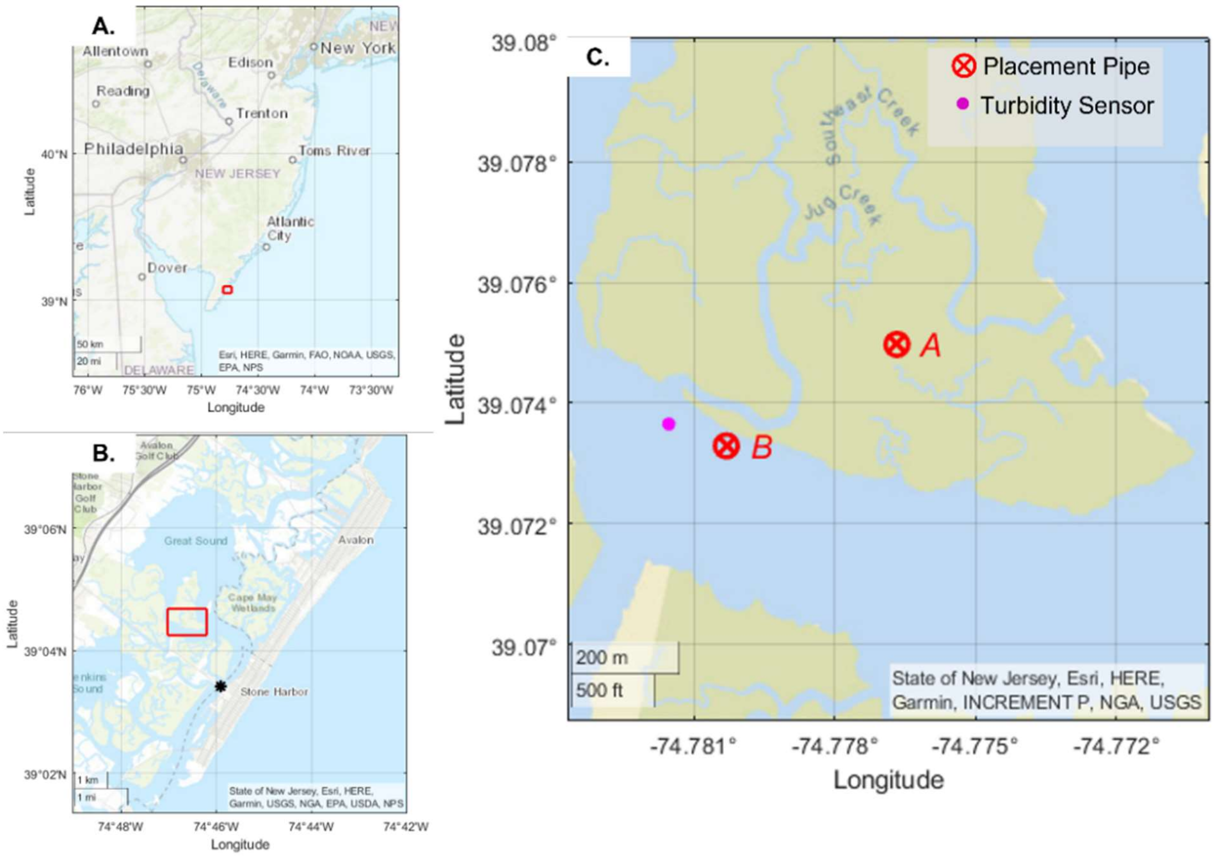

Figure 1. Study area map. The regional location of the study area is shown within the red box in (A). The area that constitutes SMIIL is shown in (B), with a red box to highlight the area of this study and black asterisk to show the location of USGS weather and tide gauge stations. The southern end of Gull Island is shown in (C), the locations of the placement pipes are indicated by red circles with x's in the middle and the location of the deployed turbidity sensor is indicated by the magenta dot.

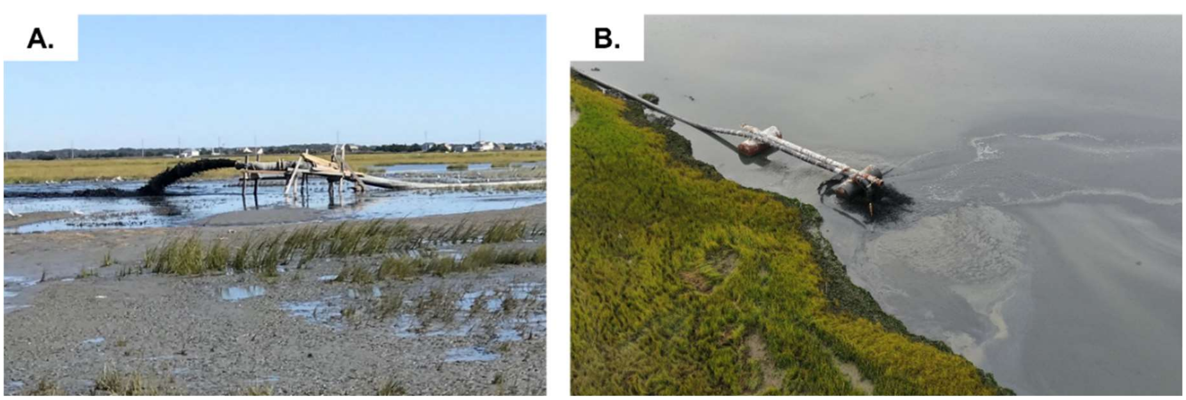

Figure 2. Discharge pipes on marsh (A) and in the nearshore (B, photo credit Gary Paul, courtesy of NAP) during Gull 2020 placement.

Turbidity monitoring was focused on the southern end of Gull (Figure 1C). The region was surveyed while material was being pumped (e.g. active pumping) on 9/23/20-9/25/20 and 9/29/20. For these surveys, the vessel cruised in parallel transects beginning within 10-20 m of the discharge pipe and progressed in an offshore direction. The region surveyed was determined by water level accessibility and the distance in which a turbidity plume was detected, such that turbidity was observed to be greater than conditions (~10-20 NTUS) observed in the area during pre-placement monitoring (Fall et al, 2021). This varied slightly each day, but generally, surveys covered an area around 10-20 m from the marsh edge to roughly 200 m off the marsh and extended alongshore between 0-300 m to the west and 300-500 m to the east of the placement pipe.

Post-Placement Monitoring

Placement ended at *A* and *B* on 10/27/20 and 10/8/2020, respectively. A post placement turbidity survey along the south end of Gull was done on 11/4/20 with a vessel mounted YSI.

Continuous Turbidity Monitoring

In addition to the roving surveys, an In-Situ Aquatroll 600 water-quality sonde equipped with a turbidity sensor was deployed off the south end of Gull Island, roughly 55 m from the marsh edge and about 100 m west of the discharge pipe (Figure 1C, magenta dot). The location was chosen to be close enough that the influence of the dredging could be detected without burying the sensor. The sensor was moored about 0.3 m above the bed and sampled every 15 min from 9/23/20-11/8/20.

Meteorological and Tidal Conditions

Water level elevation, wind speed, and wind direction were downloaded from a USGS tide gauge and weather station in Stone Harbor, NJ (https://waterdata.usgs.gov) to characterize meteorological and tidal conditions. The tide gauge and weather station are located approximately 3.2 km south of the study area (black asterisk, Figure 1B).

RESULTS

Turbidity During Placement

Turbidity in the mid to lower water column was monitored for roughly 2-3 hours during active placement on 9/23/20-9/25/20, and for 0.5 hours on 9/29/20. Water level elevation, wind speed, and wind direction during active placement are shown in Figure 3A, B. The timing of roving surveys is highlighted in red, and times corresponding to active placement at *B* is marked with magenta bars along the top. During most of the surveys (23^{rd} -25^{th}), winds were calm, about 1-2 m/s. On the 29th, slightly stronger (~5-7 m/s) northerly winds were observed. Tidal positions during surveys were inferred from the water level elevation data. The surveys captured three different tidal conditions: (1) transition from slack tide to ebb tide (2) flood tide and (3) ebb tide.

Roving turbidity surveys revealed that the turbidity plume associated with unconfined placement in the nearshore (e.g., site *B*) was localized, extending less than 50 m from the marsh edge and 100-200 m along shore, and that during calm conditions (wind speeds <5 m/s), plume direction and intensity in this area were related to tidal circulation (Figure 4A-D). During the transition from slack to ebb (Figure 4A), peak turbidities were confined to within 10-20 m east of the discharge pipe and ranged between 400-700 NTUs. To the west of the pipe, turbidity was between 10-20 NTUs, which was similar to previously reported levels in this area outside of dredging activities (e.g., Fall et al., 2021). During flood tides (Figure 4B, C), peak

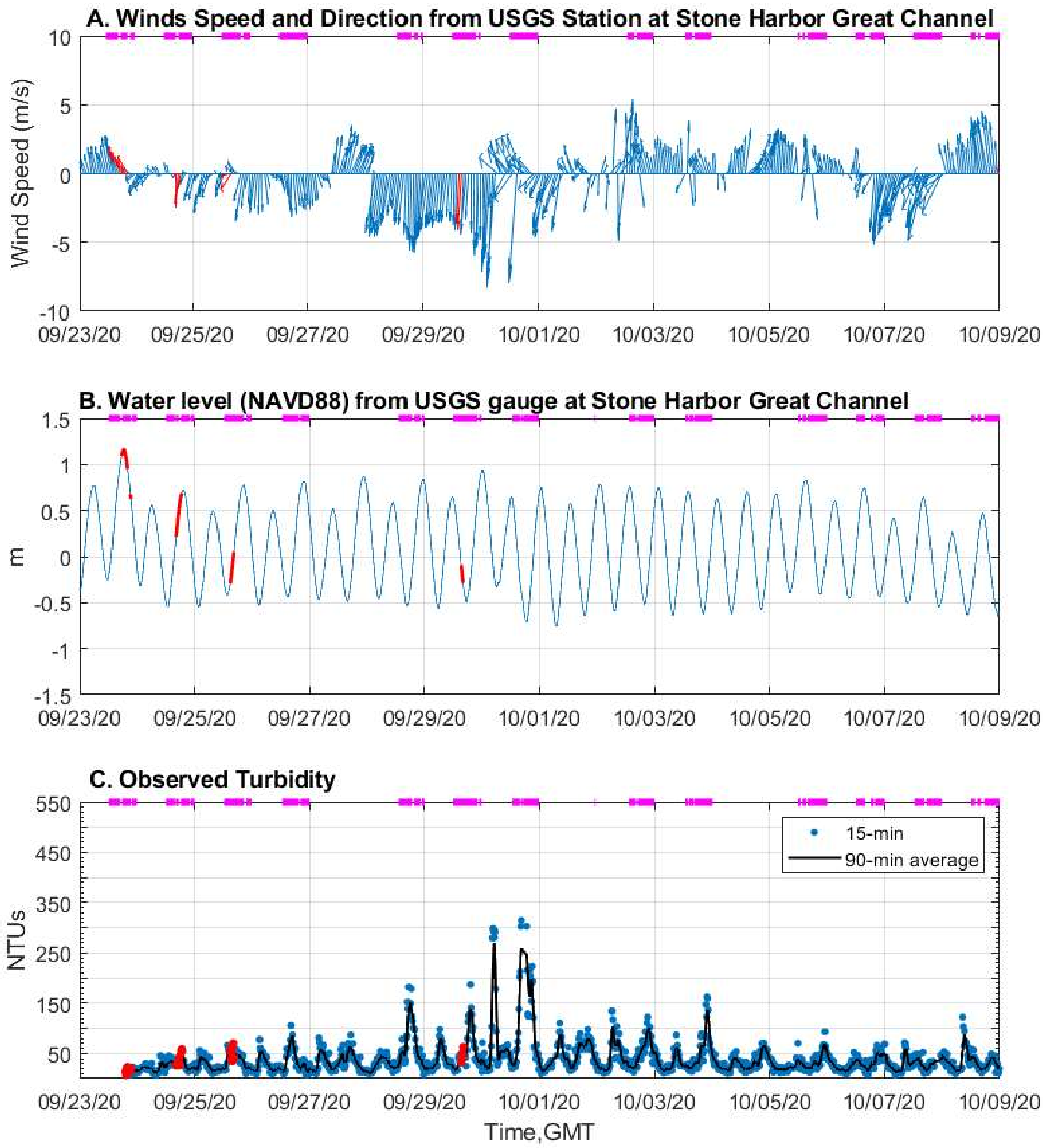

Figure 3. (A) Wind speed and direction from USGS 390325074455401 Stone Harbor weather station, NJ. Wind direction is plotted such that the arrow points to the direction the wind is blowing towards. (B) Water level elevation from USGS gauge station 014113. (C) Turbidity measured about 0.3 m above the bed, approximately 100 meters west of discharge pipe at placement B. Times are in GMT and the timing of roving surveys is highlighted in red, and times corresponding to active pumping out of pipe B is denoted by magenta bars along the top.

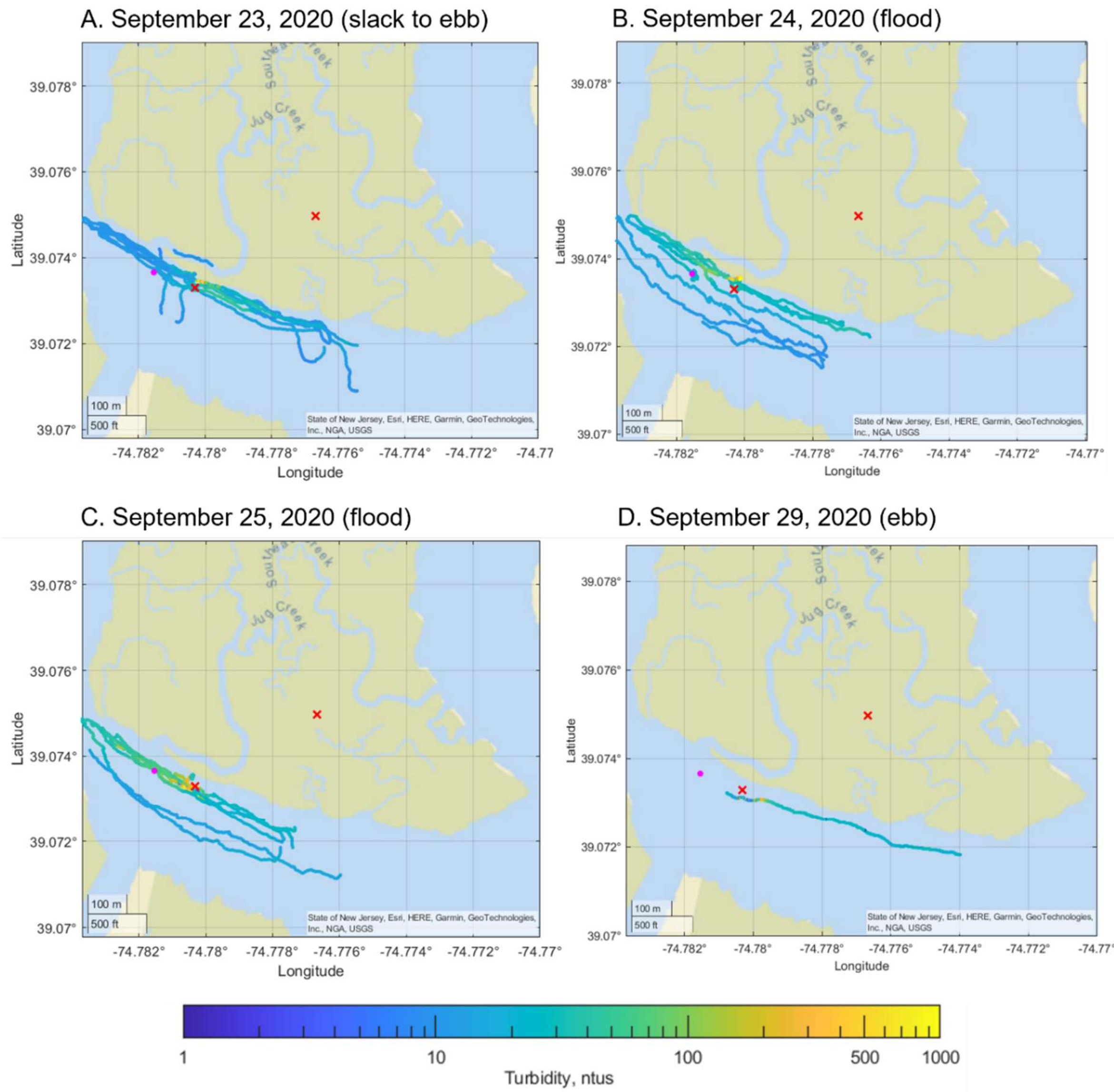

Figure 4. Turbidity collected 0.5-0.6 meters below the surface during active placement on (A) 9/23/2020, (B) 9/24/2020, (C) 9/25/2020, and (D) 9/29/20. Red x's indicate locations of placement pipes.

turbidities were higher (~900-1000 NTUs) and extended roughly 40 m west of the placement pipe. After roughly 40 m, plume intensity decreased to between 50-100 NTUs. This level of turbidity extended about 200 m along the shore and decreased with distance from the pipe to near background conditions. The survey completed on the 29th was the shortest, lasting about 0.5 hours, and was spatially limited with coverage focused east of the placement pipe and only 20-30 m to the west (Figure 4D). However, it captured observations during slightly stronger winds (Figure 3A) and during an ebbing tide (Figure 3B). Similarly, to the previous surveys, peak turbidites were localized. A peak near 1000 NTUs extended about 20-30 m southeast of the placement pipe and then decreased to background conditions.

Continuous observations of turbidity 0.3 m above the bed and 100 m west of the placement pipe is shown in Figure 3C. Over the entire placement period at *B*, 9/23/20-10/8/20, near bed turbidity was highly variable. Turbidity ranged from 7- 315 NTUs, with a mean turbidity around 39 +/- 1 NTUs, where ± denotes 1 standard error (SE) around the mean (Table 1).

Table 1. Mean turbidity during and post placement, as well as at times when tidal elevations are less than 0 m (i.e., lower water levels) and greater than 0 m (i.e., higher water levels). The ± denotes 1 standard error (SE) around the mean.

	Case	n	Mean ± 1 SE (NTUs)
During Placement	**Overall (9/23/21-10/08/21)**	**1460**	**39 ± 1.0**
	Tidal Elevation < 0 m	655	53 ± 2
	Tidal Elevation > 0 m	805	28 ± 1
Post Placement	**Overall (10/09/21-11/09/21)**	**2881**	**22 ± 1**
	Tidal Elevation < 0 m	1253	31 ± 2
	Tidal Elevation > 0 m	1628	15 ± 1

Noticeable differences in near-bed turbidity were observed between periods of higher and lower water levels such that peaks in near bed turbidity typically coincided with periods of lower water (Figure 4B, C). Observed peaks in turbidity at lower tidal elevation or water levels were generally between 50-150 NTUs (Figure 4C). Mean turbidity when tidal elevations were less than 0 m NAVD88 was 53 +/- 2 NTUs during placement. In comparison, when tidal elevations were greater than 0 m NAV88 at high mean turbidity was 28 +/- 1 NTUs (Table 1). Previously collected near bed turbidity observations in the region (Fall et al., 2021), also suggested noticeable differences in turbidity during higher and lower water levels. However, this data set was suspect, as there was a chance that at lower water levels the sensor made contact with the bed and generated turbidity. In this study, the sensor was affixed to a stable frame, to prevent such interference. Conceptually, higher near bed turbidity at lower water levels is explained by (1) surface waves and/or currents are felt more by the bed at low water and potentially suspend more material and (2) a shallower water column restricts the vertical spread and dilution of any suspended sediment.

Turbidity Following Placement

Roving surveys conducted the week following completion of placement observed turbidity levels near background in waters along the southern edge of Gull Island. A 3-hour turbidity survey using a vessel mounted YSI along the south of Gull and up two small tidal creeks was performed on November 4, 2020 (Figure 5). Again, to inform meteorological and tidal conditions, water level elevation, wind speed, and wind direction from a USGS tide gauge and weather station are presented in Figure 6A, B with timing of this survey indicated in red. The post-placement survey occurred as tide was ebbing with relatively calm (<5 m/s) northerly winds. No turbidity signal above typical background conditions reported in Fall et al., (2021) was observed within 100 m to the east and 300 m to the west of the placement pipe location at B. Higher turbidity levels, between about 30-80 NTUs, were observed in the two tidal channels in the southern portions of Gull Island, however no pre-placement turbidity data was collected in these creeks to establish a baseline for comparison.

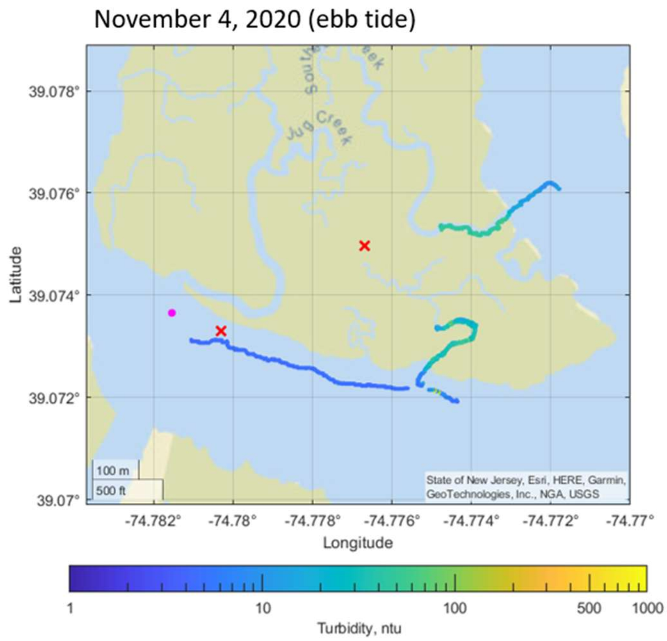

Figure 5. Turbidity collected 0.5-0.6 meters a month after placement ended on November 4, 2020. Red x's on indicate locations of discharge pipes associated with placement Area A (on island) and placement area B (open water).

Stationary monitoring of near bed turbidity near the placement pipe at *B* indicated an immediate return to baseline at the completion of placement at this site. Continuous observations of turbidity 0.3 m above the bed, 100 meters west of the placement pipe at *B* from 10/8/20-11/7/20 are presented in Figure 6D. At this location, near bed turbidity was generally between 20-100 NTUs. The mean turbidity over the period was 22 +/- 1 NTU. Similar to during placement (Figure 4D), peaks in turbidity corresponded to lower water, mean turbidity when tidal elevation was less than 0 m NAVD88 was 31 +/- 2 NTUs, was at higher water (tidal elevation > 0 m NAVD88) it was 15 +/- 1 (Table 1). Two notable turbidity events, in which turbidity levels reached about 500 NTUs, occurred around 10/15/20-10/16/20 and 11/2/20. Both events coincide with stronger, sustained winds (> 5 m/s blowing for ~24 hours) during low water. In fact, the latter event corresponds to the lowest water level observed in the given time frame.

CONCLUSIONS

Overall, results of the roving surveys suggested turbidity plumes associated with the unconfined, open water placement in the nearshore area were localized, only extending about 50 m off the marsh edge and <200 m along shore. Though current velocity was not explicitly measured in this study, observations suggest that energy in this system is typically not sufficient to keep sediment in suspension to allow for far spreading turbidity plumes. Data found that elevated turbidities were limited to times of active placement and confined to areas close to the placement pipe. Excluding areas adjacent to the placement pipe (within 40 m), turbidities were observed to be comparable or less than turbidities that were previously observed during higher energy events (Fall et al., 2021). Further, turbidity surveys found that during relative calm conditions (wind speeds <10 m/s), the magnitude and direction of the turbidity plume associated with unconfined, open water placement was predominately controlled by tidal circulation. Thus, in calm, back bay systems

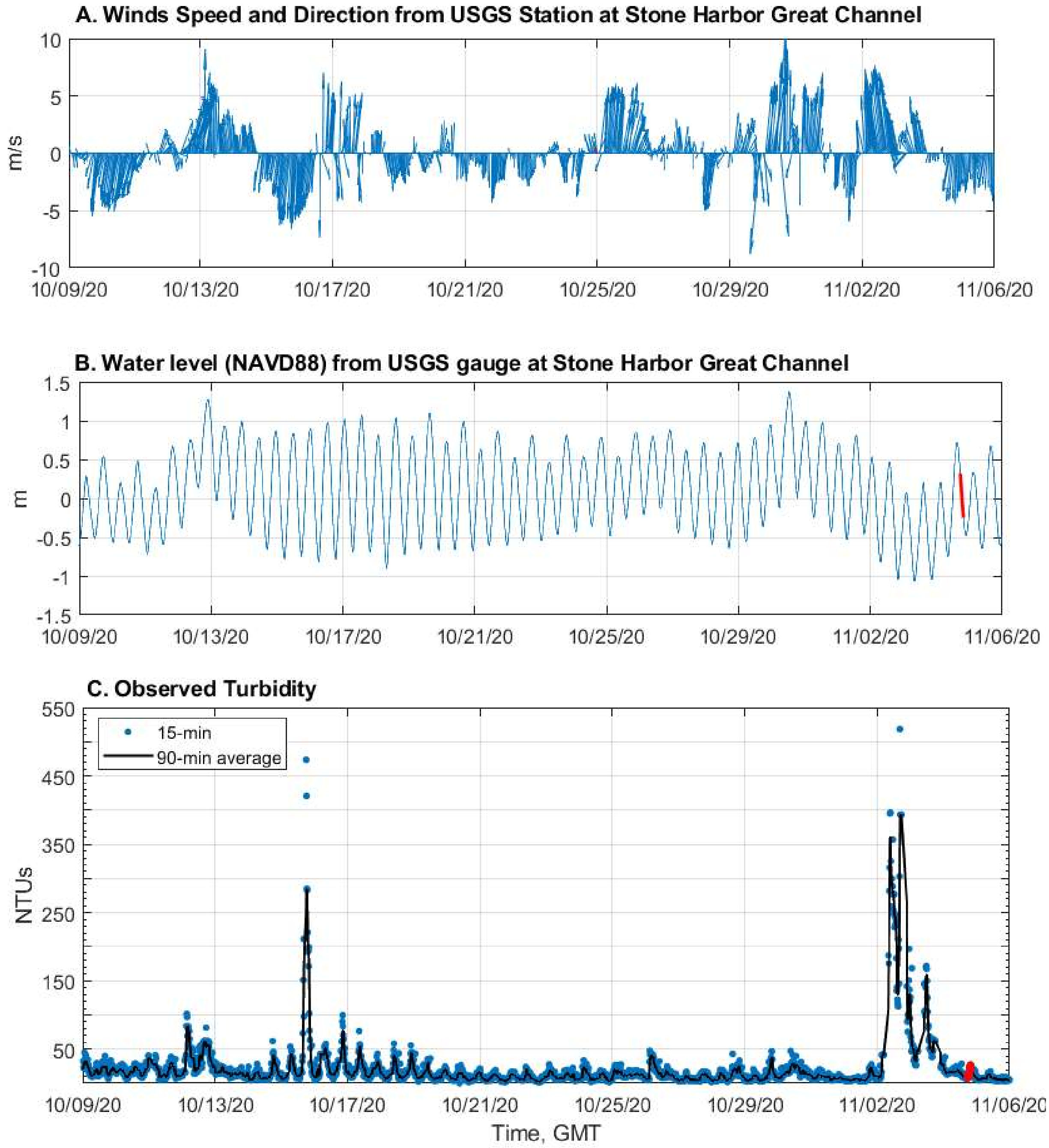

Figure 6. (A) Wind speed and direction from USGS 390325074455401 Stone Harbor weather station, NJ. Wind direction is plotted such that the arrow points to the direction the wind is blowing towards. (B) Water level elevation (E, ft) from USGS gauge station 01411360 at Stone Harbor. (C) Turbidity measured about 0.3 m above the bed, approximately 100 meters west of placement pipe at B. Times are in GMT and the timing of roving survey is highlighted in red.

open water placement practices are a promising method for increasing marsh and near marsh accretion rates, while having minimal far-field turbidity impacts.

FUTURE WORK

Observations in this study found that in this area the influence of excess turbidity associated with unconfined placement was minimal, which lends support to the concept of using fine-grained dredged sediment as a resource. However, these monitoring efforts focused solely on turbidity characteristics based on dredging and placement activities and did not include sufficient data to make strong conclusions regarding lasting impacts of the dredged material placement or if the observed data would be representative of future placement in the area or in similar areas. The next steps to improve quantification of sediment resuspension and transport and thus better predict placement impacts would be to relate turbidity measurements to a mass concentration of suspended sediments (e.g., TSS/SSC, etc.), and to better characterize currents in the system. A field effort scheduled for summer of 2022 will involve physical sampling coinciding with turbidity measurements to relate turbidity more accurately to a mass concentration of suspended sediments. During the same time, an upward looking acoustic doppler velocity profiler will be deployed to measure current velocities through the water. This effort provides more information on conditions in the region during times without dredge activities. Furthermore, the coincident measurements of current velocity, TSS, and turbidity are necessary to move forward in describing sediment transport, which in turn, can help inform future dredged material placements in this area and similar areas.

As a result of the dredge material placements at Gull Island, observable depositional features were created along the southern edge of the island. Bathymetric surveys show the persistence of these features two years after placement. Ongoing research is currently examining the geomorphological development of these features over time and how their presence may impact wave energy along the marsh edge.

REFERENCES

Arkema, K.K., Guannel, G., Verutes, G., Wood, S.A., Guerry, A., Ruckelshaus, M., Kareiva, P., Lacayo, M., and Silver, J.M. (2013). *Coastal habitats shield people and property from sea-level rise and storms.* Nature Climate Change. 913-918. https://doi.org/10.1038/nclimate1944

Bertness, M. D., Ewanchuk, P. J., and Silliman, B. R. (2002). *Anthropogenic modification of New England salt marsh landscapes.* Proceedings of the National Academy of Sciences, 99(3), 1395-1398. https://doi.org/10.1073/pnas.022447299

Church, J. A., & White, N. J. (2012). *Sea-level rise from the late 19th to the early 21st century.* Surveys in Geophysics, 32(4-5), 585-602. https://doi.org/10.1007/s10712-011-9119-1

Day, J.W., Kemp, G.P., Reed, D.J., Cahoon, D.R., Boumans, M.M., Suhayda, J.M., and Gambrell, R. (2011). *Vegetation death and rapid loss of surface elevation in two contrasting Mississippi delta salt marshes: The role of sedimentation, auto compaction and sea level rise.* Ecological Engineering 37: 229–240. https://doi.org/10.1016/j.ecoleng.2010.11.021

Fagherazzi, S.M., Kirwan, L., Mudd, S.M., Guntenspergen, G.R., Temmerman, S., D'Alpaos, A., van de Koppel, J., Rybczyk, J.M., Reyes, E., Craft, C., and Clough. J. (2012). *Numerical models of salt marsh evolution: Ecological, geomorphic, and climatic factors*. Reviews of Geophysics. https://doi.org/10.1029/2011RG000359

Fall, K.A., Perkey, D.W., Tyler, Z.J., & Welp, T.W. (2021). *Field measurement and monitoring of hydrodynamic and suspended sediment within the Seven Mile Island Innovation Laboratory, New Jersey.* ERDC/CHL TR-21-9, http://dx.doi.org/10.21079/11681/40980

Gailani, J.Z, Brutsché, K.E., Hartman, M. A, Godsey, E.S, Wang, P. (2019). *Strategic placement for beneficial use of dredged material.* ERDC/CHL SR-19-3, http://dx.doi.org/10.21079/11681/33169

Ganju, N.K., Zafner, D., Kirwan, M.L., Fagherazzi, S., D'Alpaos, A., and Carniello, L. (2017). *Spatially integrative metrics reveal hidden vulnerability of microtidal salt marshes.* Nat. Commun. 8, 14156 https://doi.org/10.1038/ncomms14156

Hartig E.K., Gornitz,V., Kolker, A., Mushacke, F., Fallon ,D. (2002). *Anthropogenic and climate-change impacts on salt marshes of Jamaica Bay, New York City.* Wetlands 22:71-89. https://doi.org/10.1672/0277-5212(2002)022[0071:AACCIO]2.0.CO;2

Kirwan, M.L., and Murray, A.B. (2007). *A coupled geomorphic and ecological model of tidal marsh evolution.* Proceedings of the National Academy of Sciences 104: 6118–6122. https://doi.org/10.1073/pnas.0700958104

Kopp, R.E., Andrews, C.J., Broccoli, A., Garner, A., Kreeger, D., Leichenko, R., Lin, N., Little, C.M, Miller, J.A., Miller, J.K., Miller, K., Moss, R., Orton, P., Parris, A., Robinson, D.A., Sweet, W., Walker, J., Weaver, C.P., White, K., Campo, M., Kaplan M. B., Herb J., and Auermuller, L. (2019). *New Jersey's Rising Seas and Changing Coastal Storms: Report of the 2019 Science and Technical Advisory Panel. Rutgers*, The State University of New Jersey. Prepared for the New Jersey Department of Environmental Protection. Trenton, New Jersey. https://doi.org/10.7282/t3-eeqr-mq48

Miller, D., Padeletti, A., Kreeger, D., Homsey, A., Tudor, R., Creveling, E., DePhilip, M.M, and Pindar, C. (2012). *Chapter 5—Aquatic Habitats* in the Technical Report for the Delaware Estuary & Basin. Partnership for the Delaware Estuary. PDE Report. No. 12-01, pp. 119-165. https://www.nj.gov/drbc/library/documents/TREB-PDE2012/Ch5-aquatic-habitats.pdf

Mudd, S.M. (2011). *The life and death of salt marshes in response to anthropogenic disturbance of sediment supply.* Geology 39: 511–512. https://doi.org/10.1130/focus052011.1

Sweet, W.V., Hamlington, B.D., Kopp, R.E., Weaver, C.P., Barnard, P.L., Bekaert, D., Brooks, W., Craghan, M., Duesek, G., Frederikse, T., Garner, G., Genz, A.S., Krasting, J.P., Larour, E., Marcy, D., Marra, J.J., Obeysekera, J., Osler, M., Pendleton, M., Roman, D., Schmied, L., Veatch, W., White, K.D., and Zuzak, C. (2022). *Global and Regional Sea Level Rise Scenarios for the United States: Updated Mean Projections and Extreme Water Level Probabilities Along U.S. Coastlines*. NOAA Technical Report NOS 01. National Oceanic and Atmospheric Administration, National Ocean Service, Silver Spring, MD, 111 pp. https://oceanservice.noaa.gov/hazards/sealevelrise/noaa-nostechrpt01-global-regional-SLR-scenarios-US.pdf

Peteet, D.M., Nichols, J., Kenna, T., Chang, C. Browne, J., Reza, M., Kovari, S., Liberman, L., and Stern-Protz, S. (2018). *Sediment starvation destroys New York City marshes' resistance to sea level rise.* PNAS Direct Submission. www.pnas.org/cgi/doi/10.1073/pnas/1715392115.

A MULTI-DECADAL ASSESSMENT OF DREDGED SEDIMENT BENEFICIAL USE PROJECTS PART 1: ECOLOGICAL OUTCOMES

Jacob F. Berkowitz[*†1], Nathan R. Beane[1], Nia R. Hurst[1],
Jacob F. Jung[1], and Kevin D. Philley[1]

ABSTRACT

Dredged sediment has been used to create and restore wetlands and other landforms for decades as part of beneficial use initiatives. Previous studies demonstrate that beneficial use projects yield ecological functions such as habitat maintenance, floodwater detention, and biogeochemical cycling. However, questions persist about the long term ecological trajectory of beneficial use projects due to 1) short monitoring timeframes and 2) the paucity of beneficial use sites that have reached maturity since most beneficial use projects were recently constructed. In response, ecological functions were assessed at six >40 year old dredged sediment beneficial use projects and adjacent reference areas with available historic post construction monitoring data. Results indicate that after four decades the beneficial use projects 1) generally achieved and maintained their target habitats, 2) became more similar to reference areas over time while remaining on unique trajectories, 3) display similar responses to environmental conditions as reference areas despite some persistent differences, and 4) continue to provide a wide array of ecological functions. Findings suggest that establishment of beneficial use success criteria should not over emphasize replicating reference conditions but should focus on achieving specific ecosystem functions (i.e., energy dissipation) and desirable outcomes (i.e., storm surge reduction). The analysis also highlights the need for additional research into long term beneficial use project trajectories, especially as new initiatives including Working with Nature and Engineering With Nature® continue to expand. A companion paper links these findings with an established ecosystem goods and services framework to holistically evaluate the long term outcomes of dredged sediment beneficial use projects.

Keywords: Natural and nature-based features, habitat, soil, vegetation communities, avian communities

INTRODUCTION

The U.S. Army Corps of Engineers (USACE) and cooperating agencies constructed multiple habitat improvement projects using dredged sediment from 1974 to 1978 (Landin et al. 1989). The projects sought to establish high quality target habitats including coastal marshes, upland meadows, and other desirable landscape features. These sites were monitored intermittently until 1987, documenting the capability of dredged sediment beneficial use activities to improve habitat. The projects document some of the first research into dredged sediment beneficial use applications in the United States. Early monitoring results suggested the projects improved habitat and demonstrated the utility of incorporating the beneficial use of dredged sediment into federal navigation, sediment management, and ecosystem restoration programs. Notably, this early research focused on habitat creation and enhancement and did not directly consider the full suite of functions and desirable outcomes (e.g., energy dissipation; storm surge reduction) frequently discussed today.

Subsequent research completed at dozens of dredged sediment placement locations over the last four decades highlights the interplay between habitat restoration and creation, engineering design, and environmental outcomes (Edwards and Proffitt 2003). For example, Faulkner and Poach (1996) assessed the functional capacity of created and natural wetlands, detecting similarities and differences between natural and built environments; Mallach and Leberg (1999) evaluated faunal community use of natural islands and islands created using dredged sediment, identifying variables that affected faunal community composition and abundance; and Yozzo et al. (2004) reported that dredged sediment is valuable for achieving a variety of habitat creation and restoration objectives including constructing artificial reefs, oyster reef restoration, intertidal wetland and mudflat creation, and the development of bird and wildlife islands as part of a regional sediment management strategy. These studies demonstrate that use of dredged sediment for habitat restoration and creation supports some of the ecological functions observed in natural systems, although the magnitude and trajectory of functions may differ (Faulkner and Poach 1996). Additionally, Streever (2000) reported that project designs incorporating elements that mimic natural processes begin to display ecosystem characteristics similar to natural systems faster and to a greater extent than landscape features constructed using traditional techniques.

The implementation of dredged sediment beneficial use projects has continued to expand over time, and the evaluation of beneficial use alternatives is now commonly incorporated into navigation programs, regional sediment management plans, and agency operational guidelines (Berkowitz and Szimanski 2020). However, while the number of dredging projects that incorporate beneficial use features continues to increase, the absolute volume of dredged sediment being used beneficially has not increased concurrently (ranging from 20 to 50% of the annual dredging volume in the United States; 1997-2017 average = 38%; Bell et al. 2021). To further promote beneficial use projects, ongoing research is focused on the capacity of natural processes to improve ecological endpoints while accomplishing engineering objectives including the maintenance of navigation channels (Daigneault et al. 2016). Additionally, efforts are underway to more fully capture the full suite of benefits provided by projects, which can improve the benefit-cost ratios used in evaluating dredged material management options (Foran et al. 2018; Ahadi et al. 2018). In the United States, the newest phase of ecosystem and dredged sediment management is encapsulated by the Engineering with Nature® (EWN) initiative, an intentional alignment of natural and engineering processes to efficiently and sustainably deliver economic, environmental, and social benefits (Table 1; Bridges et al. 2014). The beneficial use of dredged sediment is consistent with EWN within emerging water resources infrastructure paradigms (King et al. 2021).

Table 1. Key elements associated with the Engineering With Nature® initiative

1. Use science and engineering to produce operational efficiencies supporting sustainable delivery of project benefits.
2. Use natural processes to maximum benefit, reducing demands on limited resources, minimizing the environmental footprint of projects, and enhancing the quality of project benefits.
3. Broaden and extend the base of benefits provided by projects to include substantiated economic, social, and environmental benefits.
4. Use science-based collaboration to organize and focus interests, stakeholders, and partners to reduce social friction, resistance, and delays while producing more broadly acceptable projects.

The absence of long term studies evaluating dredged sediment beneficial use project outcomes remains one of the challenges precluding further expansion of beneficial use initiatives. These uncertainties result from the limited period (several decades) that dredged material management projects have incorporated opportunities for ecosystem improvement, representing much shorter time frames than reflected in natural system evolution (Coleman et al. 1998). The lack of constructed projects >50 years old precludes the collection and analysis of data from fully 'mature' beneficial use study locations. Also, few studies track the trajectory of beneficial use sites for extended periods, with most monitoring efforts occurring over short durations (often <2–5 years), which while generating valuable information D'Avanzo (1989) and others suggest is insufficient for documenting the long term benefits and outcomes of ecological restoration and creation projects. The paucity of long-term ecological monitoring data represents a significant ecological trajectory gap, limiting practitioner's ability to conduct project life cycle analysis and develop data driven success criteria and project milestones (Berkowitz et al. 2021a).

Another challenge to expanding beneficial use activities is cost, since beneficial use project implementation can be more expensive than traditional dredged sediment disposal options (PIANC 2009). However, to date few projects capture the true economic value delivered by beneficial use projects. Failure to account for all project benefits, in conjunction with the Federal Standard that has required implementation of the least cost alternative limits practitioners' ability to implement innovative dredged sediment management strategies (Brandon and Price 2007; Bell et al. 2021). As a result, there is a need to develop frameworks that document the full suite of positive outcomes associated with dredged sediment beneficial use projects, including ecological functions, goods, and services (Kolman 2014).

In response to these challenges, six historic dredged sediment beneficial use projects were assessed to document long term ecological outcomes and compare them with natural reference areas. The following provides descriptions of each study site and subsequent sections highlight the findings of vegetation community surveys, avian habitats assessments, and an evaluation of soil conditions at each location.

STUDY LOCATIONS

Natural resource assessments were conducted in 2019 at six historic dredged sediment beneficial use project locations. These projects were initially developed between 1974 and 1977, and post construction monitoring data was collected for up to 10 years (Landin et al.1989). The project locations represent some of the earliest beneficial use sites with monitoring data in the United States. As a result, the project sites provided a unique opportunity to investigate mid to long term outcomes of dredged sediment beneficial use initiatives. Additionally, the projects represented a range of geographic locations and target habitat types (e.g., marsh, meadow, dune), allowing for the evaluation of beneficial use outcomes in a variety of ecological settings (Figure 1).

The historic (1974 to 1987) monitoring documented vegetation, avian communities, and soil characteristics at each project location and compared their results with observations made at natural, unaltered reference areas (Newling and Landin 1985). Notably, the projects' design and early monitoring focused on improving habitat and did not consider economic parameters, social implications, or other factors that are now recognized within broader contexts such as the EWN initiative. However, the historic studies did attempt to utilize natural processes to improve habitat through the creation of target ecosystems (e.g., marshes, upland meadows). The following sections provide a brief description of each study site. They also indicate how each project reflects aspects of the EWN initiative to place the projects within a modern perspective and utilize them as proxies to evaluate how other projects will likely evolve at decadal timescales. While limited historical data were available regarding the sources, quantity, and pre-construction characteristics of the dredged sediments used at the project sites, Berkowitz et al. (2021) describes available data related to the study locations and provides a full accounting of all methods along with the complete 2019 assessment dataset.

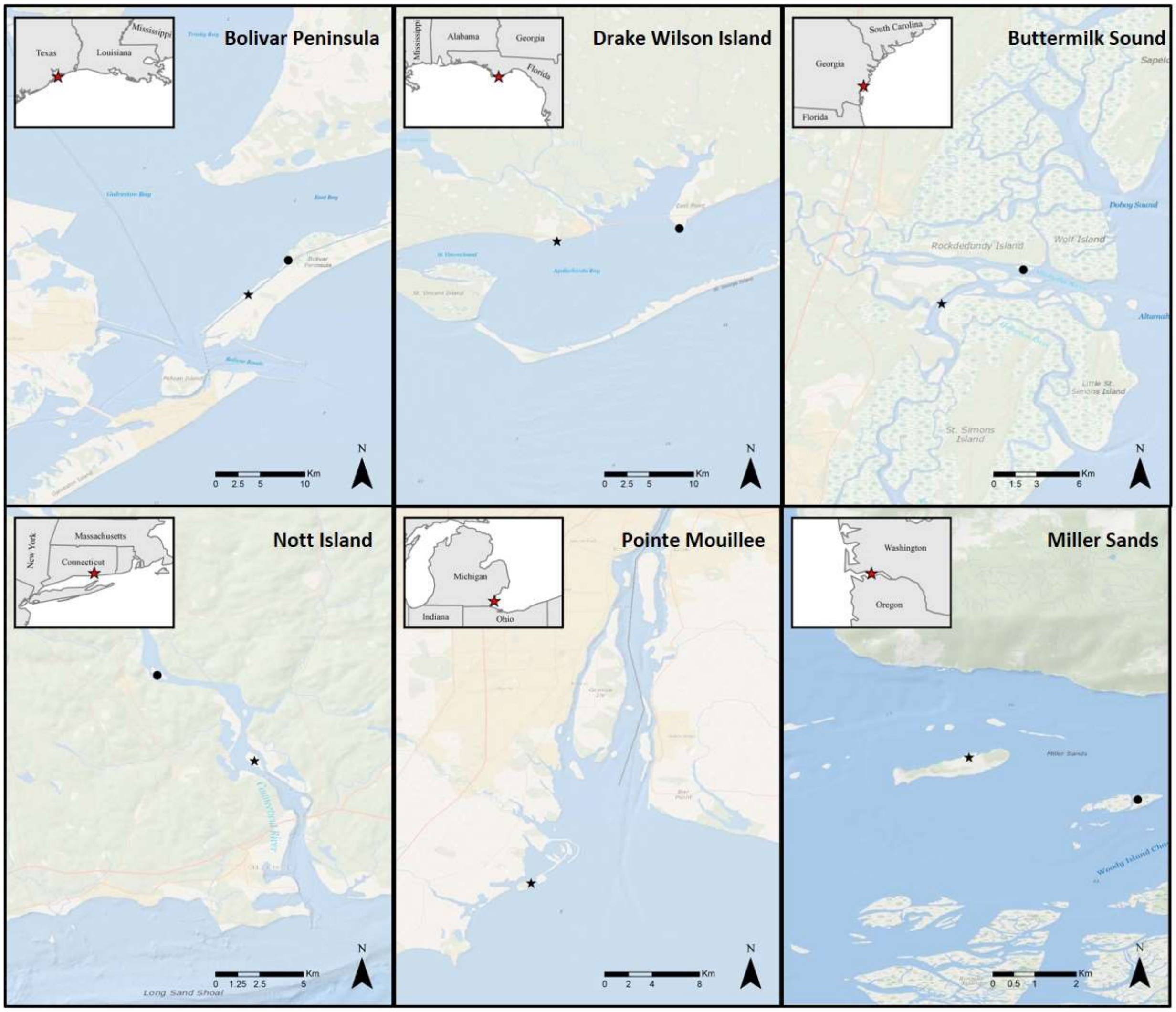

Figure 1. Location of the beneficial use projects (★) and reference areas (●) assessed.

Bolivar Peninsula, TX

The Bolivar Peninsula beneficial use project is located adjacent to the Gulf Intercoastal Waterway near the Houston Ship Channel. An 11.1 ha area of dredged sediment previously deposited on the peninsula was contoured using construction equipment in 1976 to create an elevation gradient capable of supporting the establishment of upland, high marsh, and low marsh/intertidal habitats. Vegetation was established using fertilizer and plantings. A reference area composed of saltmarsh habitat, Pepper Grove, was selected near the beneficial use project location. Differences in elevation between the beneficial use site and the reference marsh were identified as important during the historic studies and indicated that plant growth at the project site equaled or exceeded growth at the reference area. However, root biomass remained lower at the project site than in the unaltered natural marsh. The early monitoring efforts indicated that elevation gradients strongly influenced species survival and vigor. This project included features that align with aspects of the EWN initiative because it mimicked natural elevation gradients to establish a variety of habitats and target vegetation community types. Additionally, after the initial construction phase natural tidal processes and ecological succession were allowed to drive landscape evolution within the project area. The project also avoided the inclusion of hardened engineering features, which are used in a variety of shoreline stabilization and other restoration contexts but have been shown to alter a number of ecological functions and processes to varying degrees (Fischenich 2003; Peters et al. 2015).

Drake Wilson Island, FL

Drake Wilson Island was constructed in 1976 using sediment derived from the adjacent Two-Mile Channel, a navigation route linking the Gulf Intercostal Waterway, Apalachicola Bay, and the Gulf of Mexico. The 5 ha project sought to develop an emergent marsh and woody upland habitats by placing hydraulically pumped fine-grained silty dredged sediment onto older coarse-grained sandy dredged sediment deposited during previous dredging operations. Transplants of smooth cordgrass and salt meadow cordgrass were installed in wetland areas. Landin et al. (1989) reported that by 1982 the site was almost totally covered with vegetation with only one small open water pond located at the dredged material dispersal outlet pipe. Researchers noted that the entire complex was vegetated and heavily used by wildlife in 1986, 10 years after project construction. A reference marsh, Cat Point, was located east of the study site. This project included features that align with aspects of the EWN initiative because it utilized fine-grained dredged sediment that mimicked substrates found in the natural marshes of the region, and a weir was installed to ensure tidal inundation patterns were established during the initial post construction phase. Natural tidal exchange occurred following dredged material consolidation and the weir was allowed to degrade. Plantings utilizing locally sourced vegetation avoided the inclusion of hardened engineering features, and following initial construction natural processes (e.g., tides, species succession) were allowed to drive environmental evolution at the project site.

Buttermilk Sound, GA

The 2.1 ha beneficial use project at Buttermilk Sound was developed in 1975, adjacent to the Atlantic Intracoastal Waterway near the mouth of the Altamaha River, GA. The objectives of the project were to: 1) convert a ~5 m high dredged sediment sand mound to intertidal marsh habitat; 2) document changes in the field site over time; and 3) demonstrate that a stable marsh could be created using sandy dredged sediment. The area was graded and planted with vegetation. A nearby reference marsh, Hardhead Island, was selected to assess characteristics of a naturally evolved wetland ecosystem. Within five years of construction, the project was reported to be visually indistinguishable from unaltered marshes (Landin et al.1989). This project included features that align with aspects of the EWN initiative because it established an environmental gradient of elevation with tidal inundation similar to unaltered areas in the region, and allowed for natural processes (e.g., tidal creek migration) to occur while avoiding the use of hardened features.

Nott Island, CT

The Nott Island beneficial use site is located on the Connecticut River, one of the state’s most vital waterways. The 3.2 ha project was developed in 1974 in an area where historic dredged sediment deposits raised the elevation of the island by several meters. The unvegetated sand mound was considered low quality habitat due to nutrient limitations associated with coarse soil textures and steep slopes that precluded the establishment of vegetative communities and the utilization of the site by faunal communities of interest. The beneficial use project included grading the site to create an upland meadow, incorporating fine-grained dredged sediment into the sandy substrate to improve conditions for plant growth, and amending the soils with lime and fertilizer. Vegetation was successfully established to create habitat for birds and other species, with approximately 80% of the planted area vegetated within the first growing season (Landin et al.1989). Eustasia Island was selected as a reference monitoring site due to its similar geomorphology, substrate, and proximal location. This project included features that align with aspects of the EWN initiative because it incorporated finer grained sediment into the sandy dredged sediment deposits to mimic soil textures found in unaltered habitats in the region, encouraged the establishment of plant communities of high habitat value, allowed natural patterns of plant species succession to occur following construction, and avoided the inclusion of hardened engineering features.

Pointe Mouillee, MI

The Pointe Mouillee beneficial use project is adjacent to the western shore of Lake Erie. Historically, the area included a barrier beach that protected an extensive marsh complex from wind and wave driven erosion. However, the barrier beach was destroyed by high energy storm events coinciding with a period of high lake levels in the 1960's, which induced extensive erosion and degradation of >1,618 ha of marsh. In response, a 148 ha area was diked to provide protection for the degraded adjacent marsh from wind and wave erosion and a Confined Disposal Facility (CDF) was constructed to hold dredged sediment removed from the Lake Erie Ship Channel. The CDF was strategically situated and designed to match the configuration of the historic barrier island to support revitalization of the degraded marsh. The project objectives included: 1) protect and stabilize the wetlands and adjacent wildlife management area; 2) reestablish marsh habitat through sedimentation and plant colonization; 3) establish a multi-use site that includes a visitors' center and recreational opportunities; and 4) provide for the deposition of dredged sediment from Lake Erie harbors and channels. Historic monitoring data indicated that plant colonization took place within three growing seasons after construction (Landin et al. 1989). No suitable reference monitoring location was identified near the Pointe Mouillee beneficial use site. The Pointe Mouillee project included features that align with aspects of the EWN initiative because it protected the remaining adjacent marshes, provided habitat and recreational opportunities while maintaining the capacity to manage dredged sediment from the nearby navigation channel. Unlike the other projects examined, the Pointe Mouillee project utilized hard infrastructure including dikes and riprap to stabilize project features and prevent erosion as well as to contain contaminated sediment. This infrastructure was needed for wave energy attenuation which served to replace the ecological functions (e.g., energy dissipation) provided by the eroded barrier beach that previously protected the surrounding marshes and to prevent the contaminated sediment release identified in a subset of the dredged sediment in the project area.

Miller Sands, OR

The Miller Sands beneficial use project is located in the Columbia River adjacent to a navigation channel serving Astoria, OR. The original island was constructed with dredged sediment in 1932, and it received additional dredged sediment during maintenance operations at approximately four year intervals (Landin et al. 1989). The beneficial use project resulted in development of three distinct habitats including 1) upland meadows, 2) wetland marshes, and 3) dunes. In 1974, the upland portion was disked using farming implements to prepare a seed bed, which was subsequently planted. Other parts of the island were graded to an intertidal elevation and planted to establish wetland habitats; and the sand spit was planted with beachgrass interspersed with sand fencing. A nearby marsh, Snag Island, was selected as a reference location. This project includes features that align with aspects of the principles and objectives of EWN because it was designed to provide a variety of habitat types common in the region based on a gradient of elevations and associated inundation patterns. Additionally, the project utilized natural processes (e.g., erosion of the sand berm and dunes, sediment accretion in wetlands) to contour and shape the island after construction, while providing for the management of dredged sediment in the Columbia River navigation channel. Specific plantings were selected for each target habitat based on inundation tolerances and the project avoided the inclusion of hardened engineering features.

METHODS

The 2019 assessment utilized the same metrics and sampling techniques applied in the original monitoring program conducted in the 1970s and 1980s, including assessments of vegetation community composition and distribution, avian habitat utilization, and soil characteristics at each beneficial use site and at unaltered reference locations. This approach allowed for time series comparisons between the unaltered reference locations and the historic monitoring data to evaluate how the project sites matured and evolved over the past four decades (Newling & Landin 1985).

Vegetation sampling utilized triplicate sample points within representative locations of each plant community observed at the beneficial use sites and corresponding reference locations (Figure 1). Plant communities were classified based on dominant and/or diagnostic species assemblages. Triplicate 1 m^2 quadrats were used to estimate percent cover of all species and triplicate 0.25 m^2 quadrats were used to calculate the stem density (i.e., the number of vegetative stems occurring within a defined area) of all species present.

Avian community surveys utilized fixed point count locations at an intensity of one sample point per 10 ha, resulting in a total of 117 unique point count locations. At each sample point an unlimited distance, five minute point count was conducted, during which all individual birds seen or heard were recorded. Birds that were flying over the survey area were tallied separately, as the observer could not determine whether these birds were using the survey area. The survey then calculated total and average bird abundance and species richness at each study site.

For the soils assessment, triplicate 5 cm soil cores were collected from the surface horizons within each vegetation community at the beneficial use project locations and at the associated reference areas. Soils data were not available from the Pointe Mouillee, MI, beneficial use site or the Cat Point, FL, reference area. Soils were homogenized and analyzed for moisture content (MC); bulk density (BD); loss on ignition (LOI), which provides a measure of soil organic matter content; pH; salinity; extractable ammonium (NH_4^+), nitrate (NO_3^-), and soluble reactive phosphorus (SRP); and total carbon (TC), nitrogen (TN), and phosphorus (TP). Triplicate 10 cm soil cores were also collected for below ground biomass (BGB) determinations. Soil samples were processed following the same methods used during the 1970s assessment period. Berkowitz et al. (2021) provides comprehensive description of all analytical methodologies.

RESULTS AND DISCUSSION

The 2019 assessment indicates that the historic beneficial use projects successfully developed the target habitat types described in the project objectives and early monitoring reports, and 13 of the 15 target habitats remain functional more than four decades after project construction (Table 2; Figures 2a and 2b). The finding that most of the target habitats were achieved and remain functional after multiple decades is significant because few studies evaluate beneficial use outcomes over long periods. Further, sustainability is one of the goals of the EWN initiative and the fact that, in general, the target habitats remain functional while continuing to respond to natural processes suggests that beneficial use projects can yield long term benefits across a variety of landforms, regions, and ecological contexts.

The beneficial use projects have evolved both ecologically and geomorphologically over time, with most study areas displaying increases in total landcover and vegetation abundance, coupled with decreases in the spatial extent of open water and barren areas (Berkowitz et al. 2021). In total, the projects have resulted in the creation of >70 ha of new land since construction and have expanded at an average rate of 0.33 ha/yr (range = -0.16 to 1.6 ha/yr). These findings, in conjunction with the vegetation, avian, and soil condition

Table 2. Summary of target habitats, success of sustained development over >40 years, challenges, and management opportunities to improve conditions at beneficial use sites.

Location	Target habitat type	Present after >40 years	Functional without intervention	Challenges	Management opportunities
Bolivar Peninsula, TX	Low marsh	No	No	Erosion	Sediment placement
	High marsh	Yes	Yes	None	None
	Herbaceous upland	Yes	Yes	Undesirable species	Selective species control
	Woody upland	Yes	Yes	Planted species outside of native range	Selective species control
Drake Wilson Island, FL	Low marsh	Yes	Yes	Erosion	Sediment placement
	High marsh	Yes	Yes	Erosion	Sediment placement
	Woody upland	Yes	Yes	None	None
Buttermilk Sound, GA	Low marsh	Yes	Yes	None	None
	High marsh	Yes	Yes	None	None
	Unvegetated upland	Yes	No	Woody species encroachment	Selective species control
Nott Island, CT	Upland meadow	Yes	Yes	Undesirable species, poor soil quality	Selective species control, soil amendments
Pointe Mouillee, MI	Freshwater marsh	Yes	No	Woody species encroachment, Invasive species	Selective species control
Miller Sands, OR	Upland meadow	Yes	No	Poor soil quality, woody species encroachment	Selective species control, soil amendments
	Tidal marsh	Yes	Yes	Erosion, Invasive species	Selective species control
	Dune	Yes	No	Erosion	Sediment placement

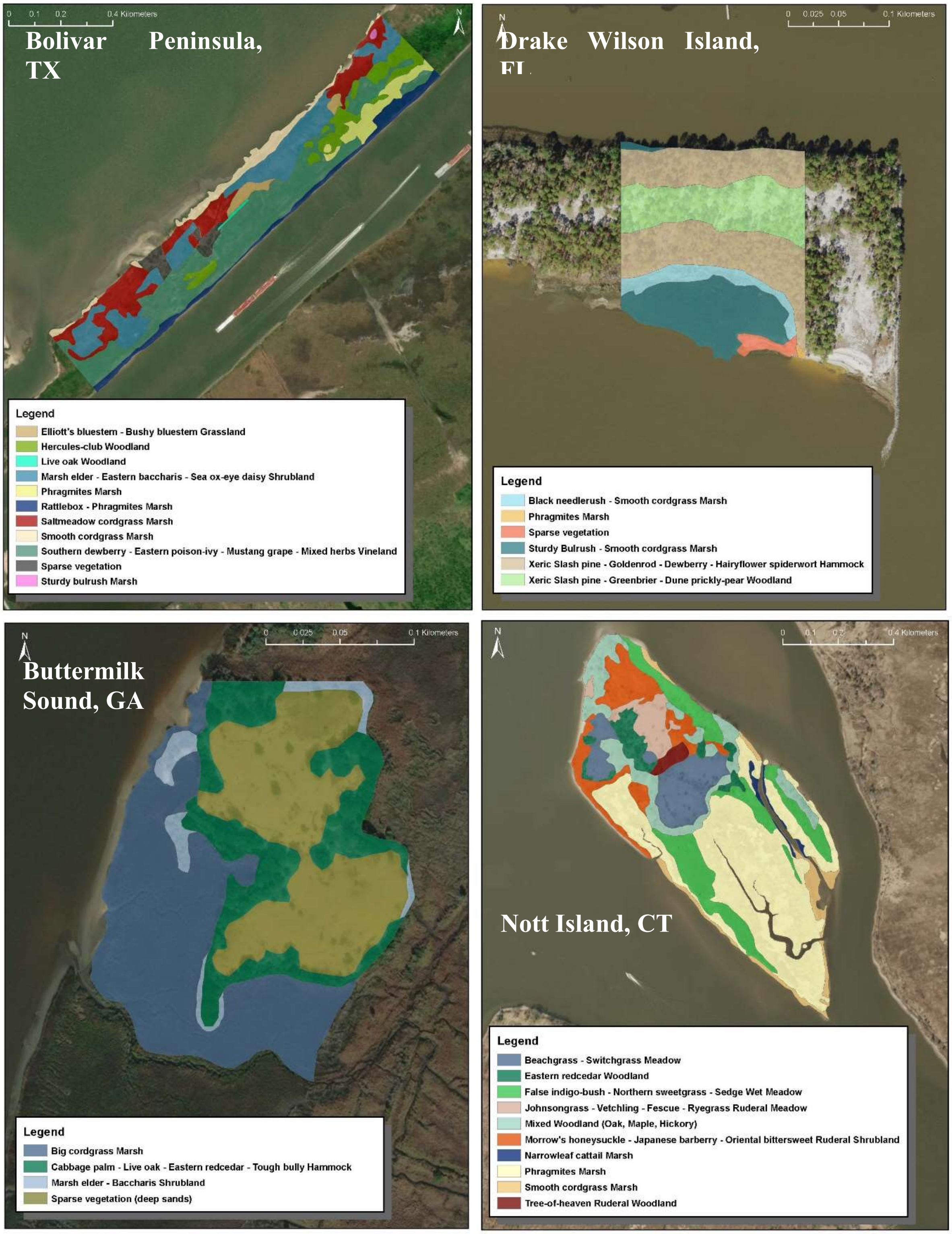

Figure 2a. Vegetation community composition at beneficial use projects after >40 years.

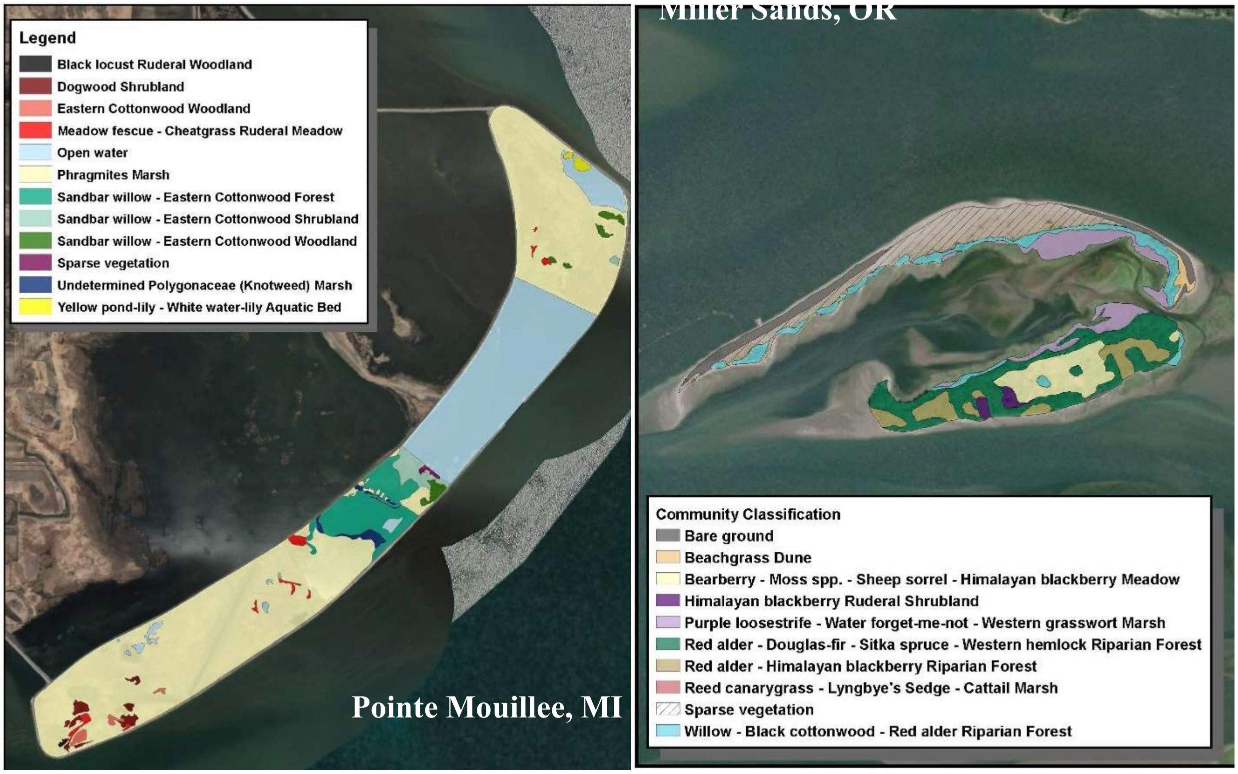

Figure 2b. Vegetation community composition at beneficial use projects after >40 years.

data described below demonstrate that the beneficial use projects provide a variety of habitat, physical, and biogeochemical cycling functions.

Vegetation community assessment

Diverse vegetation community assemblages were identified at the six beneficial use project sites, ranging from densely vegetated marshes to forested areas and sparsely vegetated dunes. Notably, the spatial distribution of habitat components and/or the suite of species present today differs from the conditions observed during post construction and early monitoring surveys at some study locations. For example, a number of the species planted and/or initially established at the Bolivar Peninsula are either absent or occur as minor components of the plant community.

The shift in species composition following construction is not unanticipated as ecological succession occurs in response to biotic and abiotic factors that drive the success of individual species and plant communities over time, and natural processes were allowed to drive the evolution of the project areas after establishment (Zedler 2000). In some cases, the disconnect between project designs and steady state conditions can be attributed to inappropriate species selection, including the planting of some species (e.g., sand pine in coastal TX) outside of their native ranges.

At other project locations, target habitat communities may not be sustainable without further interventions to maintain favorable conditions. For example, additional periodic sediment placement may be required to maintain low marsh communities (Bolivar Peninsula, TX), woody plant removal would help sustain open sandy habitats (Buttermilk Sound, GA), and soil amendments would help improve the conditions for vegetative growth and habitat development in areas subject to nutrient limitations (Miller Sands, OR). Additionally, nuisance, invasive, and nonnative species pose a significant challenge to achieving ecological goals across many of the project sites, as well as the reference locations evaluated. In general, the number and abundance of undesirable species has increased during the post construction period at both beneficial use and reference areas (Berkowitz et al. 2021).

The available data indicates that initial planting efforts had limited effects on the distribution of plant species after 40 years of ecological succession, and others have suggested that post restoration plantings may not be necessary in areas with rapid natural recruitment (Mitsch et al. 1998). In the current study, plantings of saltmeadow cordgrass and smooth cordgrass at Drake Wilson Island, FL have become components of more complex communities, as other species that were not planted have been naturally recruited. In other cases, planted species (e.g., black needlerush at Buttermilk Sound, GA) are no longer observed within the project area, or occur to a much lower extent. Despite the absence of some planted species at beneficial use locations >40 years after construction, establishing appropriate species following project construction likely has advantages even if those plant communities do not persist or decline in abundance over decadal timescales. Initiating vegetative growth after disturbance has been shown to stabilize soils and sediment, accelerate dewatering, provide habitat, retain sediment and build elevation, improve soil health, and promote ecological functions (Bailey et al. 2019). Additionally, establishing plant communities can influence the ecological trajectory of project areas, even if the planted species fail to persist, migrate in response to environmental conditions, or become integrated into a more diverse plant assemblage. The establishment of desirable species has also been shown to delay or preclude the invasion of undesirable species and planted species can deliver ecological functions and other benefits more quickly and to a greater magnitude while site maturation and natural patterns of vegetation succession occur (Simonstad et al. 2006).

The habitat complexity of study locations generally increased over time. At Bolivar Peninsula, TX, distinct plant communities increased from 6 to 10 between 1988 and 2019 (Table 3). The increase in complexity over time is not unexpected given natural patterns of vegetation recruitment, response to disturbance, and the adjustment of plant communities to local conditions following four decades of ecological succession. Increases in species complexity following restoration have been reported in other studies, especially as seed banks become enriched. Baldwin (2004) provides a model outlining plant distribution patterns following marsh habitat restoration, indicating that increases in species richness are not expected to continue indefinitely as the project sites reach equilibrium. However, species composition will naturally respond to disturbances (e.g., floods, fire) and changes in ecological conditions (e.g., climate; invasive species), at both beneficial use and reference locations.

Table 3. Summary of vegetation community characteristics at dredged sediment beneficial use (BU) and reference locations. NA = no data available.

	Vegetation community assemblages (count)		**Dominant species richness in target community types (count)**			
Location	**BU location**	**Reference location**	**Habitat type**	**BU (2019)**	**BU (historic)**	**Reference (2019)**
Bolivar Peninsula, TX	10	1	Low marsh	4	2	2
Drake Wilson Island, FL	6	8	Low marsh	2	2	2
Buttermilk Sound, GA	4	2	Marsh	4	3	3
Nott Island, CT	10	4	Meadow	16	5	NA
Pointe Mouillee, MI	7	NA	Marsh	7	4	NA
Miller Sands, OR	7	1	Marsh	18	17	15

The vegetation community assemblages observed in the beneficial use sites were more ecologically complex than the reference areas, with the exception of the Cat Point, FL, reference area which exhibited barrier beach features absent at the Drake Wilson Island beneficial use project (Table 3). This increased complexity results from a combination of factors including the wider degree of topographic relief within the constructed project areas relative to the reference areas. This finding may seem counterintuitive, as other studies have identified the lack of topographic heterogeneity, especially with regards to surface microtopography as a limitation in many restoration projects (Bruland and Richardson 2005). However, within the locations examined, the habitat improvement project designs deliberately created ecological gradients that would support a variety of habitat types. This included the intentional establishment of upland, transitional, high marsh, low marsh, and other target habitats with appropriate elevations and hydropatterns.

The number of species in target communities has remained stable or increased over time at the beneficial use project sites, which display higher species richness than the reference locations. This suggests that while the beneficial use projects have reached a moderate to high level of maturity, based on the high stem densities and percentage of ground cover, they do not directly mimic the plant community assemblages of reference areas. The differences in species richness aligns with other studies that report engineered or restored areas often exhibit more species than their natural counterparts (Ehrenfeld 2000). The differences in species composition and richness may include intentionally or unintentionally introduced plants, the response of plant communities to varying substrate conditions, or reflect disturbance patterns associated with restoration design and implementation (Baldwin and Derico 1999). While the beneficial use project plant communities have trended towards conditions observed in reference areas, differences continue to persist after more than four decades of ecological succession.

Notably, both beneficial use locations and reference areas appear to be responding to environmental conditions in similar ways. For example, stem densities in both restored and reference areas follow similar patterns over the multi-decadal assessment period despite the observed differences in species composition and habitat complexity (Figure 3). Additionally, more invasive species were observed at both beneficial use and reference areas during 2019 than during previous investigations and those species appear to be increasing in abundance (Berkowitz et al. 2021). This trend has been reported in other areas, where anthropogenic alterations coupled with natural patterns of disturbance and plant dispersal are leading to increased alien invasions (Richardson et al. 2007). This further suggests that the vegetation communities at beneficial use locations are exhibiting similar responses to changing environmental conditions that impact plant distribution and growth at reference areas.

In summary, the vegetation communities established at the beneficial use sites undoubtedly improved habitat compared with preconstruction conditions. The project areas generally reflect the species assemblages at reference locations and have become more similar to reference locations with age, but do not replicate the reference systems. Notably, these plant communities have persisted for more than four decades without the need for major intervention. The findings of the current study align with previous research suggesting that beneficial use projects that incorporate ecological drivers (i.e., hydroperiod, salinity tolerance, and topography) into project design are sustainable and may have advantages over projects constructed with conventional techniques (Streever 2000). Additional studies directly comparing the long-term outcomes of beneficial use projects constructed using a variety of techniques is recommended to further investigate the relationship between ecological outcomes and construction techniques.

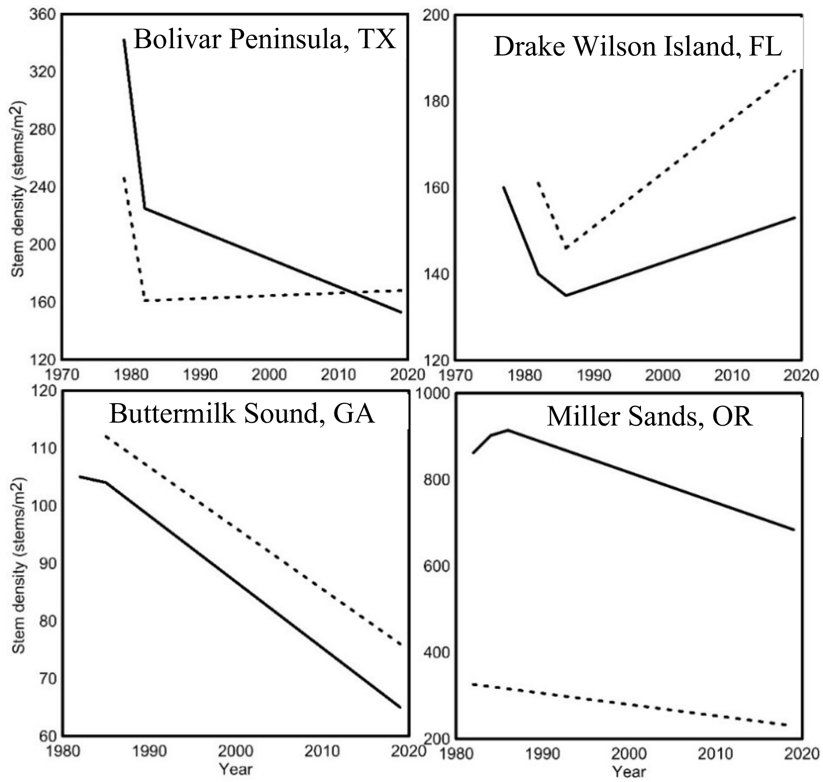

Figure 3. Comparison of vegetation stem density data from marsh habitats at beneficial use locations (solid lines) and reference areas (dashed lines) over time. Note that while the magnitude of stem densities varies, the constructed and reference locations display similar patterns and responses to environmental conditions over time.

Results also suggest that management activities could further improve site conditions especially with regard to selective species management, and in some cases the implementation of periodic disturbance regimes (e.g., additional dredged sediment deposition) may improve the functionality and sustainability of some landscape features including low marshes and coastal fringe wetlands (Table 2). For example, using thin layers of sediment to increase the elevation of areas threatened by subsidence and high rates of relative sea level rise has been shown to improve habitat conditions for plant growth, biogeochemical cycling, and other ecological functions (Puchkoff and Lawrence 2022).

Avian community assessment

The evaluation of avian communities provides additional data to document the habitat functions delivered by beneficial use projects (Neckles et al. 2002). The targeted, intentional placement of dredged sediment has been shown to effectively develop new habitats or improve existing habitat for birds (Erwin et al. 2003), and habitats supported with dredged sediment have proven as productive as natural marshes in some circumstances (Streever 2000). A total of 3,790 individual birds were encountered during the 2019 assessment, including wetland dependent birds which were present at each of the beneficial use project sites (Table 4). Marsh wren, red-winged blackbird, and yellow warbler were the most commonly detected species. Other common wetland breeding species frequently observed included boat-tailed grackle, clapper rail, common yellowthroat, and willow flycatcher (Berkowitz et al. 2021).

Table 4. Summary of avian species detected at beneficial use (BU) and reference area (RA) locations during 2019. †Indicates locations surveyed outside of the breeding season and colored text identifies paired beneficial use and reference sites.

Location	Number of point counts surveyed	Total abundance including flyovers	Total abundance excluding flyovers	Average abundance per point including flyovers	Average abundance per point excluding flyovers
Bolivar Peninsula, TX (BU)†	21	406	256	19	12.2
Pepper Grove, TX (RA)†	6	746	586	124	97.7
Drake Wilson Island, FL (BU)	8	129	78	16	9.8
Cat Point, FL (RA)	6	100	60	17	10
Buttermilk Sound, GA (BU)	6	118	93	20	15.5
Hardhead Island, GA (RA)	8	115	59	14	7.4
Nott Island, CT (BU)	11	202	179	18	16.3
Eustasia Island, CT (RA)	9	132	127	15	14.1
Pointe Mouillee (BU)	16	871	558	54	34.9
Miller Sands, OR (BU)	22	776	715	35	32.1
Snag Island, OR (RA)	3	155	147	52	49

Data from locations surveyed during the breeding season indicated that the beneficial use sites support higher total species richness and average species richness than unaltered reference areas (Table 5). Habitats were generally more diverse within beneficial use areas, which likely contributed to the higher levels of species abundance and richness compared to reference sites. However, the highest total and average species richness was observed at the Pepper Grove reference site, but this area was surveyed outside of the breeding season during the spring migration period when high numbers of waterbirds use the area as stopover habitat.

Ospreys and bald eagles are raptor species indicative of high habitat quality that utilize coastal and inland freshwater areas. Ospreys were observed at 50% of beneficial use sites conducted by Landin et al. (1989) but were observed at all sites during 2019, including nesting activities at the Nott Island, CT, and Pointe Mouillee, MI, beneficial use sites (Table 6). Bald eagles were also observed nesting successfully at the Miller Sands, OR, beneficial use project site. While osprey and bald eagle populations have recovered following legislative protection and the ban of the chemical dichlorodiphenyltrichloroethane (DDT). The 2019 avian community assessment demonstrates that these species are using the dredged sediment beneficial use sites for foraging and nesting, and that their habitat utilization has increased over the past >40 years.

The beneficial use projects examined herein continue to serve as important habitat for a diversity of avian communities since their creation over 40 years ago. The establishment of vegetation communities ranging from coastal salt marshes to upland habitats support a wide range of bird species, including numerous Species of Conservation Concern (USFWS 2021). While the habitat improvement sites continue to support abundant populations of birds, the 2019 survey observed differences in species assemblages and abundances when comparing the assessment data with both previous survey efforts and results collected at reference areas.

Table 5. Avian species richness detected at the beneficial use (BU) and reference area (RA) locations during 2019. †Indicates locations surveyed outside of the breeding season and colored text identifies paired beneficial use and reference sites.

Location	Number of point counts surveyed	Total species richness including flyovers	Total species richness excluding flyovers	Average species richness per point including flyovers	Average species richness per point excluding flyovers
Bolivar Peninsula, TX (BU)†	21	61	49	7.5	4.52
Pepper Grove, TX (RA)†	6	60	57	18.33	14.17
Drake Wilson Island, FL (BU)	8	32	24	10.88	7.5
Cat Point, FL (RA)	6	28	20	8.83	5.5
Buttermilk Sound, GA (BU)	8	16	12	8.0	6.17
Hardhead Island, GA (RA)	6	14	5	6.0	3.5
Nott Island, CT (BU)	11	43	36	9.75	9.45
Eustasia Island, CT (RA)	9	30	26	8.77	8.22
Pointe Mouillee (BU)	16	52	45	10.81	8.06
Miller Sands, OR (BU)	22	39	37	8.77	8.09
Snag Island, OR (RA)	3	16	11	7.0	5.0

Table 6. Comparison of observations of osprey and bald eagles at projects during the historic monitoring (Landin et al.1989) and the 2019 assessment.

Beneficial use location	Osprey		Bald Eagle	
	1989	2019	1989	2019
Bolivar Peninsula, TX	X	X		
Drake Wilson Island, FL		X		X
Buttermilk Sound, GA	X	X		
Nott Island, CT	X	X		X
Pointe Mouillee, MI		X		
Miller Sands, OR		X	X	X

In general, restoration sites had higher species richness compared to reference sites due to a greater diversity of habitats. Other studies have reported successful establishment of avian populations at beneficial use project sites, including rates of bird usage that exceed undisturbed reference areas (Warren et al. 2002). The intentional placement and manipulation of dredged sediment to yield gradients of elevation, soil conditions, and hydrologic regime resulted in the development of a wider range of target habitat types compared to reference sites, and a diversity of habitats is more attractive to nesting and foraging birds than homogenous expanses of a single habitat type (Rafe et al. 1985). The observed differences in species composition between constructed and natural areas result from broad habitat type effects (e.g., marsh, scrub-shrub, forest) and the presence of more complex habitats at the beneficial use locations; not differences in microhabitats within a particular habitat type (e.g., lack of tidal creeks in salt marshes). This suggests that future projects seeking to maximize avian community diversity should focus on creating a variety of target habitats as opposed to mimicking the less complex landforms observed at reference areas. Berkowitz et al. (2021) reports the full dataset generated during the 2019 avian community assessment.

Differences in species assemblages were also observed between the 2019 surveys and previous surveys described in Landin et al. (1989). In the years immediately following construction of the beneficial use projects numerous shorebirds, terns, and skimmers were documented on unvegetated dredged sediment deposits. However, with the exception of American white pelicans on the dune habitat at Miller Sands, beach nesting habitat is no longer available to support this community of waterbirds after more than four decades of ecological succession. This is not unexpected considering the duration of time since construction activities occurred. Areas that receive dredged sediment undergo a series of plant successional changes that eventually result in the development of dense herbaceous, shrub-scrub, or forested communities in the absence of frequent inundation or intermittent disturbance (Soots and Parnell 1975). Since the construction of the beneficial use projects, the vegetation communities have reached the later stages of succession and extensive shorebird habitat is no longer available at the study locations.

In summary, the 2019 surveys documented the persistence of moderate to high levels of avian community diversity and abundance at the beneficial use project locations. The results confirm the findings of previous studies that report islands constructed off the mainland are often more successful for bird reproduction because they lack mammalian predators in comparison to landscape features connected to the mainland. Over the past 40 years, plant succession and soil development occurred, and bird communities responded as anticipated with declines in habitat for beach-nesting birds and the expansion of habitat for species that forage and nest in densely vegetated areas. As a result, the periodic placement of additional dredged sediment at the project sites or the intentional removal of woody vegetation represent potential management activities that would increase habitat for beach-nesting species while providing opportunities to maintain the adjacent navigation channels (Guilfoyle et al. 2019).

Soils assessment

Soils and sediment characteristics play a key role in determining ecological outcomes (Muñoz Rojas 2018). Soils provide the physical substrate supporting habitats and also supply seed sources, nutrients, and organic matter to initiate plant establishment and succession (Abella et al. 2020). However, soils can limit or postpone the success of projects when their characteristics are not adequately considered (Eviner and Hawkes 2008). Common soil parameters limiting positive beneficial use outcomes include high salinity, low nutrient or organic matter content, bulk densities too low to support plant establishment or too high (compaction) to promote root penetration, and the alteration of soil structure during project construction (Mendes et al. 2019). Additionally, soils in dredged sediment beneficial use project areas may exhibit textures, microbial and fungal communities, and other characteristics that differ from reference conditions. These factors influence soil-plant interactions, and, although soils generally become more similar to natural substrates over time, the differences in soil characteristics can persist for several decades or even centuries after the implementation of management activities (Craft et al. 2002). The following reports changes in soil conditions >40 years after beneficial use project implementation and discusses soil characteristics in comparison to natural reference areas.

Each of the beneficial use projects displayed improvements in soil physiochemical properties with age, including increasing nutrient content and improved physical structure (Table 7). For example, loss on ignition (LOI), which reflects soil organic matter content and serves as a proxy measure of soil organic carbon storage, increased over time and in most cases approached the values observed in reference locations. This demonstrates that carbon is accumulating in the beneficial use sites, increasing soil water holding capacity and decreasing bulk density to improve conditions for flora and fauna. Extractable NO_3^- and NH_4^+ also increased over time, indicating that enhanced nutrient and biogeochemical cycling is occurring in beneficial use sites.

Table 7. Summary of soil characteristics in marsh communities at beneficial use (BU) and reference areas (RA) observed during the 2019 assessment and available historic monitoring data. Colored text identifies paired beneficial use and reference sites. The full soil assessment results are provided in Berkowitz et al. (2021).

Parameter†	Bolivar Peninsula (BU; 2019)	Pepper Grove (RA; 2019)	Bolivar Peninsula (BU; 1970s)	Drake Wilson Island (BU; 2019)	Drake Wilson Island (BU; 1979)
BD (g/cm^3)	1.26±0.12	0.60±0.19	N/A	0.37±0.04	N/A
pH	7.72±0.27	7.12±0.09	8.0-8.9	6.47±0.1	7.88–8.21
Salinity (ppt)	1.43±0.15	1.70±0.30	5.9	1.05±0.09	10–18
BGB (g/m^2)	54±13	292±62	N/A	446±64	N/A
NO_3^- (mg/kg)	1.78±0.07	7.64±2.23	<0.60	1.37±0.26	N/A
NH_4^+ (mg/kg)	7.29±2.98	11.71±8.6	<0.02	6±0.65	N/A
SRP (mg/kg)	0.63±0.32	0.27±0.27	<0.2	0.18±0.1	N/A
LOI (%)	3.82±0.59	10.4±3.6	>0.56	19.8±2.14	4.53–6.10
TC (%)	1.3±0.3	3.87±1.62	N/A	8.32±0.97	N/A
TN (%)	0.09±0.02	0.3±0.11	N/A	0.49±0.054	N/A
TP (mg/g)	286±54	397±24	N/A	421±46.5	N/A

Parameter	Buttermilk Sound (BU; 2019)	Hardhead Island (RA; 2019)	Buttermilk Sound (BU; 1979)	Miller Sands (BU; 2019)	Snag Island (RA; 2019)	Miller Sands (BU; 1979)
BD (g/cm^3)	0.49±0.40	0.37±0.04	N/A	0.98±0.04	0.73±0.1	N/A
pH	6.5±0.08	6.26±0.25	6.9–7.2	5.55±0.13	6.76±0.23	7.0
Salinity (ppt)	0.23±0.04	1.4±0.12	N/A	0.01±0.01	0.03±0.03	N/A
BGB (g/m^2)	188±96	191±86	N/A	104±20	106±28	128
NO_3^- (mg/kg)	1.86±1.75	3.3±0.74	0.1–0.75	4.92±1.01	7.3±3.23	<1.1
NH_4^+ (mg/kg)	24.4±14.5	4.92±0.98	8.7–12.2	2.71±0.67	20.4±15.3	5–12
SRP (mg/kg)	0.33±0.12	0.35±0.12	N/A	0.88±0.09	0.43±0.26	N/A
LOI (%)	13.7±4.5	18.4±2.7	<5.5	4.87±0.61	5.57±1.36	0.32
TC (%)	5.76±1.9	6.72±1.37	N/A	2.77±0.49	2.55±0.62	N/A
TN (%)	0.45±0.14	0.54±0.06	0.1–0.97	0.21±0.034	0.22±0.041	N/A
TP (mg/g)	543±102	938±73	<10.4	551±15.2	667±129	N/A

†Bulk density (BD); below ground biomass (BGB); nitrate-nitrogen (NO_3^-); ammonium-nitrogen (NH_4^+); soluble reactive phosphorus (SRP); loss on ignition (LOI); total carbon (TC); total nitrogen (TN); total phosphorus (TP);

Beneficial use sites not only displayed improvements in soil characteristics relative to initial post-construction conditions, but also currently exhibit a number of similarities with their natural reference counterparts. Only one site, Bolivar Peninsula, TX, displayed more than three of the soil properties that substantially differed from conditions at reference locations. Where variations were observed, most can be attributed to the heterogeneity of habitat types present at beneficial use projects compared with unaltered reference locations (Berkowitz et al. 2021).

Despite the similarities observed in most soil parameters at beneficial use and reference areas, the belowground biomass at the beneficial use sites remains significantly lower than reference conditions. However, increases in this parameter over time were proportional and correlated to increases in soil organic matter, total carbon, and total nitrogen measurements. The lower belowground biomass content in beneficial use sites is not unexpected, as restoration projects often contain less carbon following construction and store carbon differently than natural areas (Mitsch and Hernandez 2013). This finding aligns with results of other studies, including Abbott et al. (2019), who report lower carbon concentrations in marshes created using dredged sediment across a 32-year chronosequence than in natural marshes. However, the observed trends in soil carbon and biomass accumulation suggest that soil carbon assimilation and nutrient concentrations will continue to improve over time. In other words, the soils at the beneficial use sites have yet to reach the same level of ecological equilibria observed in the assessment of vegetation and faunal communities described above. These findings further support the conclusion that the beneficial use projects share characteristics with reference areas yet remain on unique ecological trajectories and emphasizes the utility of incorporating soil parameters into post construction monitoring programs.

Notably, the study locations display the ability to sustain the soil physical substrate needed to support the growth of robust rooted vegetation communities. Additionally, observations of stratified layers of sediment in the wetland habitats indicate that the beneficial use sites are detaining sediments during flooding, which provides a proxy measure of the energy dissipation and other physical/hydrologic functions (Berkowitz et al. 2021). Further, each of the wetland areas exhibited field indicators of hydric soils which document biogeochemical cycling functions including the retention and transformation of elements and compounds, carbon sequestration, and nutrient cycling functions are occurring at the study locations (USDA-NRCS 2018). These findings, in conjunction with the vegetation and avian community survey results highlight the fact that the dredged sediment beneficial use projects are performing the full suite of ecological functions associated with each targeted habitat type.

CONCLUSIONS

The assessment of six historic dredged sediment beneficial use sites demonstrates that positive project outcomes can be sustained over multiple decades, and that these outcomes can be achieved with limited use of hard structures or intervention (e.g., extensive management). The fact that the target habitats remain functional after more than four decades and have evolved through natural processes to provide moderate to high habitat values represents a significant finding that can be used to promote expansion of beneficial use initiatives. Results indicate that beneficial use projects incorporating a variety of landforms and environmental conditions (e.g., elevation and inundation gradients) delivered a wide array of ecological functions derived from habitat, biogeochemical cycling, and physical processes. The inclusion of these project features yielded a diversity of vegetation and avian community assemblages that exceeded observations made at some of the unaltered reference areas evaluated.

The study results improve our understanding of how projects will evolve ecologically over long periods and inform future project design, implementation, and management. For example, while vegetation communities, bird utilization, and soil properties became more similar to unaltered reference areas over time, they remain on distinct trajectories that do not directly reflect reference conditions (Moreno-Mateos et al. 2012). This does not suggest that the projects failed to achieve their objectives but instead highlights the need to develop realistic project milestones based on the time needed for delivery of ecosystem functions and other benefits. Further, the use of habitat quality or value as the major determinant of project success has likely been over-emphasized and practitioners should work toward basing success criteria on the full suite of habitat, biogeochemical, and physical/hydrologic ecological functions delivered by beneficial use projects as well as associated positive societal outcomes. To promote this approach, a companion paper further explores this concept within a dredged sediment beneficial use context utilizing an established ecosystem functions, goods, and services framework.

REFERENCES

Abbott, K.M., Elsey-Quirk, T., and DeLaune, R.D. (2019). “Factors influencing blue carbon accumulation across a 32-year chronosequence of created coastal marshes,” Ecosphere 10(8).

Abella, S.R., Hodel, J.L., and Schetter, T.A. (2020). "Unusually High-Quality Soil Seed Banks in a Midwestern US Oak Savanna Region: Variation with Land Use History, Habitat Restoration, and Soil Properties," Restoration Ecology 28(5):1100–1112.

Ahadi, K., Sullivan, K.M., and Mitchell, K.N. (2018). “Budgeting maintenance dredging projects under uncertainty to improve the inland waterway network performance”. Transportation Research Part E: Logistics and Transportation Review 119:63-87.

Bailey, P., Miller, S.J., Cary, T.J., Bourne, S.G. and Sekoni, T.A. (2019). Plant community approach to establishing vegetation on DMPAs and CDFs,” Vicksburg, MS: US Army Engineer Research and Development Center Technical Note ERDC/TN EWN-19-2 http://dx.doi.org/10.21079/11681/32296.

Baldwin, A.H. (2004). "Restoring complex vegetation in urban settings: the case of tidal freshwater marshes," Urban Ecosystems 7(2):125-137.

Baldwin, A.H. and Derico, E.F. (1999). “The seed bank of a restored tidal freshwater marsh in Washington, DC,” Urban Ecosystems 3(1):5-20.

Bell, K.S., Boyd, B.M., Goetz, S.L., Hayes, D.F., Magar, V.S., and Suedel, B. (2021). “Overcoming barriers to beneficial use of dredged sediments in the US,” WEDA Journal of Dredging 19(2): 20-42.

Berkowitz, J.F., Beane, N.R., Hurst, N.R., Jung, J.F., and Philley, K.D. (2021). “An assessment of Long-Term, Multipurpose Ecosystem Functions and Engineering Benefits Derived from Historical Dredged Sediment Beneficial Use Projects,” Vicksburg, MS: US Army Engineer Research and Development Center Technical Report ERDC/TR-21-4.

Berkowitz, J.F. and Szimanski, D. (2020). “Documenting Engineering with Nature®, Implementation Within the US Army Corps of Engineers Baltimore District-Completed Projects and Opportunities for Chronosequence Analysis”. Vicksburg, MS: US Army Engineer Research and Development Center Technical Note ERDC/TN EWN-20-3.

Brandon, D.L. and Price, R.A., (2007). Summary of available guidance and best practices for determining suitability of dredged material for beneficial uses. Vicksburg, MS: US Army Engineer Research and Development Center Technical Report ERDC/EL TR-07-27.

Bridges, T.S., Lillycrop, J., Wilson, J.R., Fredette, T.J., Suedel, B., Banks, C.J., and Russo, E.J. (2014). “Engineering with Nature Promotes Triple-Win Outcomes,” Terra Et Aqua. 135:17-23.

Bruland, G.L., and Richardson, C.J. (2005). “Hydrologic, Edaphic, and Vegetative Responses to Microtopographic Reestablishment in a Restored Wetland.” Restoration Ecology 13(3):515–523 https://doi.org/10.1111/j.1526-100X.2005.00064.x.

Coleman, J.M., Roberts, H.H., and Stone, G.W. (1998). “Mississippi River delta: an overview,” Journal of Coastal Research 14(3):699-716.

Craft, C., Broome, S. and Campbell, C. (2002). “Fifteen years of vegetation and soil development after brackish-water marsh creation,” Restoration Ecology 10(2):248-258.

Daigneault, A., Brown, P., and Gawith, D. (2016). "Dredging versus hedging: Comparing hard infrastructure to ecosystem-based adaptation to flooding." Ecological Economics 122:25-35.

D'Avanzo, C. (1989) "Long-term evaluation of wetland creation projects." Wetland Creation and Restoration: The Status of the Science 2:75-84.

Edwards, K.R. and Proffitt, C.E., (2003). “Comparison of wetland structural characteristics between created and natural salt marshes in southwest Louisiana, USA”. Wetlands, 23(2):344-356.

Ehrenfeld, J.G. (2000). “Evaluating Wetlands within an Urban Context,” Ecological Engineering 15(3-4):253–265.

Eviner, V.T. and Hawkes, C.V. (2008). “Embracing variability in the application of plant–soil interactions to the restoration of communities and ecosystems,” Restoration Ecology 16(4):713-729.

Erwin, R.M., Allen, D.H. and Jenkins, D. (2003). “Created versus natural coastal islands: Atlantic waterbird populations, habitat choices, and management implications,” Estuaries 26(4):949-955.

Faulkner, S.P., and Poach, M.E. (1996). “Functional comparison of created and natural wetlands in the Atchafalaya Delta, Louisiana”. Vicksburg, MS: US Army Engineer Waterways Experiment Station Technical Report WRP-RE-16.

Fischenich, J.C. (2003). “Effects of riprap on riverine and riparian ecosystems”. Vicksburg, MS: US Army Engineer Research and Development Center Technical Note ERDC/TR-03-04.

Foran, C.M., Burks-Copes, K.A., Berkowitz, J., Corbino, J. and Suedel, B.C. (2018). “Quantifying Wildlife and Navigation Benefits of a Dredging Beneficial-Use Project in the Lower Atchafalaya River: A Demonstration of Engineering with Nature®,” Integrated environmental assessment and management 14(6):759-768.

Guilfoyle, M.P., Jung, J.F., Fischer Jr, R.A., and Dickerson, D.D. (2019). “Developing best management practices for coastal engineering projects that benefit Atlantic Coast shoreline-dependent species,” Vicksburg, MS: US Army Engineer Research and Development Center Technical Note ERDC/TN EMRRP-SI-38.

King, J., Holmes, R., Burkholder, S., Holzman, J. and Suedel, B., (2021). “Advancing Nature-Based Solutions by Leveraging Engineering With Nature®(EWN®) Strategies and Landscape Architectural Practices in Highly Collaborative Settings,” Integrated Environmental Assessment and Management, published online June 29, 2021.

Kolman, R. (2014). “Introducing ecosystems services for port development,” Environment and Sustainability, 62:181-183.

Landin, M.C., Webb, J.W., and Knutson, P.L (1989). "Long-Term Monitoring of Eleven Corps of Engineers Habitat Development Field Sites Built of Dredged Material, 1974–1987," Vicksburg, MS: U.S. Army Engineer Waterways Experiment Station Technical Report D-89-1.

Mallach, T.J., & Leberg, P.L. (1999). "Use of dredged material substrates by nesting terns and black skimmers". The Journal of Wildlife Management, 63(1):137-146.

Mendes, M.S., Latawiec, A.E., Sansevero, J.B., Crouzeilles, R., Moraes, L.F., Castro, A., Alves-Pinto, H.N., Brancalion, P.H., Rodrigues, R.R., Chazdon, R.L. and Barros, F.S. (2019). "Look down—there is a gap—the need to include soil data in Atlantic Forest restoration," Restoration Ecology 27(2):361-370.

Mitsch, W.J., and Hernandez M.E. (2013). "Landscape and climate change threats to wetlands of North and Central America," Aquatic Sciences 75:133–149.

Mitsch. W.J., Wu, X., Nairn, R.W., Weihe, P.E., Wang, N., Deal, R., and Boucher, C.E. (1998). Creating and restoring wetlands: A whole-ecosystem experiment in self-design. BioScience 48: 1019–1030.Moreno-Mateos, D., Power, M.E., Comín, F.A. and Yockteng, R. (2012). "Structural and functional loss in restored wetland ecosystems". PLoS Biology 10.

Muñoz-Rojas, M.)2018). "Soil Quality Indicators: Critical Tools in Ecosystem Restoration," Current Opinion in Environmental Science and Health 5:47–52.

Neckles, H.A., Dionne, M., Burdick, D.M., Roman, C.T., Buchsbaum, R. and Hutchins, E. (2002). "A monitoring protocol to assess tidal restoration of salt marshes on local and regional scales," Restoration Ecology 10(3):556-563.

Newling, C.J., and Landin, M.C. (1985). "Long-Term Monitoring of Habitat Development at Upland and Wetland Dredged Material Disposal Sites, 1974–1982," Vicksburg, MS: US Army Engineer Waterways Experiment Station Technical Report D-85-5.

Peters, J.R., Yeager, L.A., and Layman, C.A. (2015). "Comparison of fish assemblages in restored and natural mangrove habitats along an urban shoreline." Bulletin of Marine Science, 91(2):125-139.

PIANC. (2009). "Dredged Sediment as a Resource: Options and Constraints," EnviCom Working Group 104: PIANC, Brussels, Belgium.

Puchkoff, A.L. and Lawrence, B.A. (2022). "Experimental sediment addition in salt-marsh management: Plant-soil carbon dynamics in southern New England," Ecological Engineering, 175.

Rafe, R.W., Usher, M.B. and Jefferson, R.G. (1985). "Birds on reserves: the influence of area and habitat on species richness," Journal of Applied Ecology 22:327-335.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. and Hobbs, R.J. (2007). "Riparian vegetation: degradation, alien plant invasions, and restoration prospects," Diversity and Distributions 13(1):126-139.

Simenstad, C., Reed, D., and Ford, M. (2006). "When is Restoration Not?: Incorporating Landscape-Scale Processes to Restore Self-Sustaining Ecosystems In Coastal Wetland Restoration," Ecological Engineering 26(1):27–39.

Soots Jr, R.F. and Parnell, J.F. (1975). “Ecological succession of breeding birds in relation to plant succession on dredge islands in North Carolina,” University of North Carolina Sea Grant Publication UNC-SG-75-27.

Streever, W.J. (2000). "Spartina alterniflora marshes on dredged material: a critical review of the ongoing debate over success," Wetlands Ecology and Management 8, (5):295-316.

United States Fish and Wildlife Service (USFWS). (2021). “Birds of Conservation Concern 2021”. United States Department of the Interior, U.S. Fish and Wildlife Service, Migratory Birds, Falls Church, Virginia. http://www.fws.gov/birds/management/managed-species/birds-of-conservation-concern.

United States Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). (2018). “Field Indicators of Hydric Soils in the United States, Version 8.2”. L.M. Vasilas, G.W. Hurt, and J.F. Berkowitz (eds.). USDA-NRCS, in cooperation with the National Technical Committee for Hydric Soils.

Warren, R.S., Fell, P.E., Rozsa, R., Brawley, A.H., Orsted, A.C., Olson, E.T., Swamy, V. and Niering, W.A. (2002). “Salt marsh restoration in Connecticut: 20 years of science and management,” Restoration Ecology 10(3):497-513.

Yozzo, D.J., Wilber, P., Will, R.J. (2004). “Beneficial use of dredged material for habitat creation, enhancement, and restoration in New York–New Jersey Harbor,” Journal of Environmental Management 73: 39–52.

Zedler, J.B. (2000). “Progress in Wetland Restoration Ecology,” Trends in Ecology and Evolution 15(10):402–407.

ACKNOWLEDGEMENTS

Funding for this study was provided by the U.S. Army Corps of Engineers Dredging Operations Environmental Research Program. We thank the following USACE collaborators who made this study possible by assisting with site access and laboratory analysis: Mr. Jose Vargas (Galveston District); Mr. Richard Allen (Mobile District); Mr. Glenn Bacon (Savannah District); Dr. Kyle Tidwell (Portland District); Mr. Todd Randall (New England District); and Ms. Christina Saltus and Ms. Sydney Bufkin (Engineer Research and Development Center). The authors also thank three peer reviewers and the handling editor for providing insightful and valuable feedback.

A MULTI-DECADAL ASSESSMENT OF DREDGED SEDIMENT BENEFICIAL USE PROJECTS PART 2: ECOSYSTEM FUNCTIONS, GOODS, AND SERVICES

Jacob F. Berkowitz[†1], Nia R. Hurst[1], Nathan R. Beane[1], Kevin D. Philley[1], and Jacob F. Jung[1]

ABSTRACT

Data and observations made at > 40-year-old dredged sediment beneficial use project sites were used to link ecosystem functions (e.g., maintenance of floral and faunal habitat, energy dissipation) with an established ecosystem goods and services framework (e.g., navigation channel maintenance, hazard reduction, ecosystem sustainability). This approach works toward quantifying the full suite of positive outcomes dredged sediment beneficial use projects provide to the environment and society. Ecological functions are derived from physical, biogeochemical, and habitat processes which occur on different timeframes and to varying magnitudes, and these functional drivers control the delivery of ecosystem goods and services. For example, physically dominated ecological functions are typically delivered more quickly (weeks to months) after project implementation than functions requiring the maturation of plant communities or other biologically mediated processes (years to decades). As a result, coupling ecological functions with the resulting ecosystem goods and services informs dredged sediment beneficial use decisions by communicating the relative influence of specific design features or management actions on project outcomes. These analyses also support the development of conceptual ecological benefits trajectories across decadal timelines. Future research will be needed to improve the quantification of ecological functions, and the resulting goods and services in a dredged sediment beneficial use context. The need for better quantification tools is expected to increase with implementation of Working with Nature, Engineering With Nature, and natural and nature-based feature initiatives. A companion paper evaluates the long-term ecological outcomes of dredged sediment beneficial use project implementation, demonstrating the capacity of beneficial use projects to sustainably deliver a variety of ecosystem functions over multiple decades.

Keywords: Natural and nature-based features, ecological functions, ecosystem goods and services, restoration trajectory curves

INTRODUCTION

There is a paucity of data on the long-term outcomes of dredged sediment beneficial use projects from an ecological function, goods, and services perspective. The lack of studies linking dredged material beneficial use initiatives with ecosystem functions, goods, and services is two-fold. First, few beneficial use projects were implemented more than 30 years ago, limiting the capacity to conduct long-term analyses. Second, the ecosystem function, goods, and services typologies evolved relatively recently, and are only now being incorporated into engineering and environmental management practices (Bouwma et al. 2018). In this paper, we utilize data and observations made at six historic (> 40-year-old) dredged sediment beneficial use projects to evaluate multi-decadal ecosystem function, goods, and services outcomes. This is accomplished by 1) providing a background of ecosystem functions, goods, and services within a dredged sediment beneficial use context, 2) describing the ecosystem functions occurring at each study location, 3) and linking them to the ecosystem goods and service framework described in Waigner et al. (2020). Additionally, the ecological trajectory of beneficial use sites is discussed to highlight the relative timing of

[1] US Army Engineer Research and Development Center, 3909 Halls Ferry Rd, Vicksburg, MS, 39180, USA

ecosystem functions, goods, and services delivery as well as the impact that specific project design features can have on environmental and societal outcomes.

BACKGROUND

Ecological functions are defined as the normal activities or actions that take place in an ecosystem, such as the maintenance of habitat for flora and fauna, detention of floodwater and reduction of storm surges, and the biogeochemical cycling of nutrients and other compounds (Novitski et al. 1996). These functions result from complex interactions between the structural components in an ecosystem (e.g., plants, animals, soil, water, and the atmosphere), the surrounding watershed and landscape (e.g., geomorphic setting), and the processes that link these structural components such as overbank flooding, evapotranspiration, chemical reactions in the soil, predation, and primary productivity (Smith et al. 2013). In general, ecological functions can be grouped into three broad categories (physical, biogeochemical, and habitat functions) based upon the underlying processes predominantly driving the function. These functions work in concert to maintain ecosystem integrity and sustainability. The hierarchy demonstrating the relationship between ecological indicators (or structural elements), processes, functions, and suites of functions is shown in Figure 1.

In addition to providing for ecosystem integrity, ecological functions also deliver benefits for society known as ecosystem goods and services that help to sustain or enhance human life (Brown et al. 2007). Ecosystem goods and services include the opportunity to harvest fish or other species for human consumption, reduced flood risk damages to infrastructure, improved water quality, and other beneficial outcomes (Gómez-Baggethun et al. 2010). As such, ecosystems represent capital assets, whose benefits and services can be assessed and quantified in a variety of frameworks, including economic and engineering contexts (Daily et al. 2000). Many studies have evaluated the complex ecosystem functions, goods, and services associated with different ecosystems, and an array of approaches to assess these outcomes have been applied across the United States and internationally (De Groot et al. 2002). In a dredging context, Wellman and Gregory (2002) discussed the incorporation of ecosystem goods and services into a coastal management trade-off analysis, suggesting the consideration of these metrics can help to achieve both economic and ecosystem integrity objectives.

Recently, an ecosystem goods and services framework was developed in support of U.S. Army Corps of Engineers project planning activities (Table 1; Waigner et al. 2020). The framework identified ecosystem goods and services categories that can be applied to a number of disciplines, including the management of dredged sediment (Kolman 2014). The Central Dredging Association (2013) described ecosystem goods and services as a new way of thinking, a tool for decision-making about sustainable development, and capable of re-focusing discussions about dredging and the environment to identify proactive, opportunity driven, win-win situations. That report highlighted the beneficial use of dredged sediment as a mechanism to increase the delivery of ecosystem goods and services, but also identified challenges for assessing, valuating, and incorporating goods and services into the design of dredging projects.

The following example examines how dredging and the beneficial use of dredged sediment can alter ecosystem functions, goods, and services in both positive and negative ways. The dredging of canals through wetlands and the placement of dredged sediments adjacent to the canals (known as spoil banks) in Louisiana, USA, has contributed to the conversion of large wetlands areas to open water features (Britsch and Dunbar 1993). The extirpated wetlands no longer provide ecological functions such as flood water retention and the reduction of storm surges, exacerbating coastal land loss in the region (Turner and McClenachan 2018). Additionally, the conversion of wetlands to open water decreased the delivery of ecosystem goods and services such as the capacity of the system to naturally reduce storm impacts to infrastructure (hazard mitigation), provide a sustainable fishery (food provisioning), and improve water quality by removing excess nitrogen which contributes to hypoxia (water purification; ecosystem

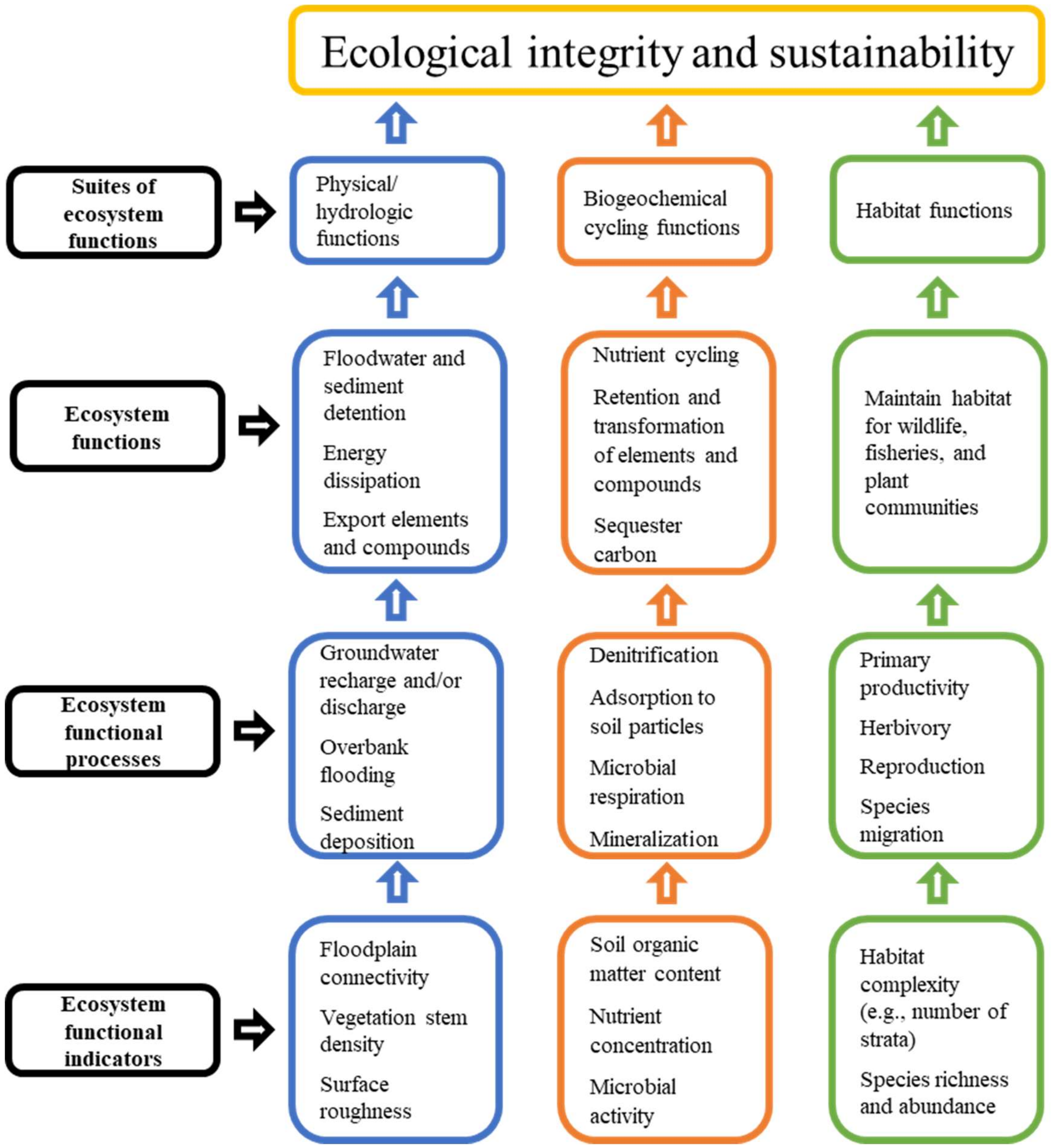

Figure 1. Hierarchy of ecological functions.

sustainability) (Bianchi et al. 2010; Chesney 2000). Conversely, dredged sediments have been used to restore and improve ecological functions in the degraded areas. In the case of the Louisiana canals, Baustian and Turner (2006) measured increases in ecosystem indicators related to habitat, hydrologic, and biogeochemical functions after canals were re-filled with dredged sediment. While not directly measured, restoring the impacted wetlands with dredged sediment appears to have increased the delivery of a number of ecosystem goods and services, including ecosystem sustainability, hazard mitigation, water purification, and climate regulation. Evidence that ecosystem functions, goods, and services have been enhanced using dredged sediments includes observed increases in soil organic matter (an ecological indicator) following canal restoration which shows that carbon sequestration (a biogeochemical functional process) and climate regulation (an ecosystem service) resulted from the dredged sediment placement.

Table 1. Framework of ecosystem goods and services categories, associated definitions, and examples related to dredged sediment beneficial use (adapted from Waigner et al. 2020).

Category	Definition	Beneficial use example
Ecosystem sustainability	Maintenance of ecosystems' structural and functional qualities and resilience to adapt to change over time.	Sediment placement to increase habitat in support of population viability for threatened species.
Hazard mitigation	Ecosystem-induced reduction of risk of or vulnerability to floods and storms that threaten property, infrastructure, safety, or natural resources.	Using dredged sediment to restore coastal marshes to decrease storm surges and reduce flood damages to infrastructure.
Navigation maintenance	Provision of unobstructed waterborne transport as supported by sediment reduction and water regulation by functioning ecosystems.	Use of dredged sediment to construct projects that improve hydrodynamics and prevent channel infilling.
Cultural, spiritual and educational support	Maintenance of sites and landscapes with spiritual or religious significance, contribute to a sense of place, or sustain cultural heritage, including traditional ways of life. Also includes opportunities for scientific discovery and education.	Use of dredged sediment to protect important archeological or indigenous sites such as shell middens from erosion or other threats.
Recreation and aesthetics	Quantity and quality of recreational opportunities and the aesthetic enjoyment provided by the condition and relative placement of landscape and ecosystem features pleasing to one or more of the five senses. This may also include property value enhancement.	Development or maintenance of beaches, wetlands, or other features using dredged sediment that provides for tourism, recreational hunting and fishing, camping, and attractive scenery for landowners and the public.
Food, raw goods, and materials provisioning	Provisioning of commercial or subsistence production of food and raw goods and materials.	Dredged sediment placement that supports hunting, fishing, and foraging opportunities; or supports timber harvest and aggregate re-use.
Water purification	The filtration and removal of excess nutrients or pollutants by ecosystems.	Constructing wetlands using dredged sediment naturally removes of excess nutrients and retains/detoxifies pollutants.
Climate regulation	Ecosystem moderation of adverse climate effects via greenhouse gas sequestration.	Construction or maintenance of wetlands and seagrass beds using dredged sediment.

The incorporation of ecosystem goods and services into dredged sediment management aligns with a recent paradigm shift elevating the concept that beneficial use projects can deliberately align engineering and environmental stewardship missions to maximize outcomes for both navigation and the ecosystem (Bridges et al. 2014). While there has been a growing recognition that incorporating measures of ecosystem functions, goods and services into dredged sediment management programs can help achieve both engineering and environmental objectives (International Association of Dredging Companies 2013; Kolman 2014), few studies have holistically evaluated these metrics in a dredged sediment beneficial use context. For example, Jenkins et al. (2010) estimated the monetary value of restoring wetland habitats in the Mississippi River Valley, focusing on greenhouse gas reductions and recreation. While valuable, that study did not consider other functions and benefits such as flood risk reduction or navigation channel maintenance. Foran et al. (2018) linked ecological functions with navigation channel maintenance, greenhouse gas dynamics, and water quality improvements at one beneficial use study site in Louisiana, providing an example of the integration of ecosystem services into the quantification of favorable outcomes. This study takes another step forward toward holistically communicating beneficial use project benefits by evaluating the long-term (>40 year) delivery of ecosystem functions, goods, and services at six geographically and ecologically dredged material management sites constructed to improve habitat while supporting sustainable navigation.

STUDY LOCATIONS AND APPROACH

Six historic dredged sediment beneficial use projects designed to improve habitat were developed between 1974 and 1977, and post construction monitoring data was collected until 1987 (Landin et al.1989). These project locations represent some of the earliest beneficial use sites with monitoring data in the United States. The data collected provide a unique opportunity to evaluate the mid- to long-term outcomes of dredged sediment beneficial use initiatives within an ecosystem function, goods, and services context. Additionally, the projects represent a range of geographic locations and target habitat types (e.g., marsh, meadow, dune), allowing for the evaluation of beneficial use outcomes in a variety of ecological settings (Figure 2). Table 2 provides a brief description of each project sites' characteristics. Additional details about the study sites and beneficial use activities are available in Berkowitz et al. (2021). To assess the multidecadal ecosystem functions, goods, and services delivered by the projects a four-tiered analysis was conducted as described below:

Tier 1 - The monitoring data from each of the historic projects summarized by Landin et al. (1989) and others was reviewed to identify direct measures and indicators of ecosystem functions. For example, throughout the 1976-1987 monitoring period, high bird species abundance and diversity and robust plant communities were recorded, demonstrating that habitat functions were occurring as a result of the dredged sediment placement activities.

Tier 2 - A field sampling campaign was completed in 2019 at each of the six historic dredged sediment beneficial use sites to assess vegetation community structure, avian community composition, and soil

Figure 2. Location of the historic beneficial use projects.

Table 2. Brief description of historic dredged sediment beneficial use projects.

Location	Year	Size (ha)	Beneficial use activity	Target habitats
Bolivar Peninsula, TX	1976	11.1	Historic pile of unvegetated dredged sediment adjacent to the Houston ship channel contoured using construction equipment to create elevation and inundation gradients for floral and faunal habitat improvements. The site was then planted with a range of flora based on elevation, salt tolerance, and inundation frequency.	Low marsh High marsh Herbaceous upland Woody upland
Drake Wilson Island, FL	1976	5	Hydraulically pumped fine grained silty dredged sediment from the Gulf Intercoastal Waterway deposited onto older course sandy dredged sediment. The area was planted with a range of flora based on elevation, salt tolerance, and inundation frequency.	Low marsh High marsh Woody upland
Buttermilk Sound, GA	1975	2.1	Converted a ~5 m high unvegetated dredged sediment sand mound adjacent to the Atlantic Intracoastal Waterway to a gradient of intertidal elevations. The area was planted with a range of flora based on elevation, salt tolerance, and inundation frequency.	Low marsh High marsh Unvegetated upland
Nott Island, CT	1974	3.2	Re-contoured an unvegetated, steeply sloped dredged sand mound adjacent to the Connecticut River. Soils were amended with fine grained dredged sediment and fertilizer, then planted.	Upland meadow
Pointe Mouillee, MI	1979	148	Strategically situated area diked to protect degraded adjacent marsh and reestablish habitat using dredged sediments, establish a visitors' center and recreational opportunities, and store Lake Erie shipping channels and harbor dredged sediments.	Freshwater marsh
Miller Sands, OR	1974	94.7	Regraded historic dredged sediment mound adjacent to the Columbia River navigation channel to develop elevation and inundation gradients. Disked and amended upland areas with fertilizer, deposited fine dredged sediments in marsh areas, and placed sand (with sand fencing) to establish dunes. The area was planted with a range of flora based on elevation, salt tolerance, and inundation frequency.	Tidal marsh Upland meadow Dune

physicochemical properties (Berkowitz et al. 2021). During the site visits and upon subsequent analysis of field and laboratory data, measurements and observations of multiple ecological functional indicators were identified within each target habitat (e.g., low marsh, vegetated upland, dune). The ecosystem functional indicators considered include both direct measures of function (e.g., avian community habitat usage) and proxy measures indicative of ecosystem functions (e.g., documented field indicators of hydric soils provide indirect, but diagnostic evidence of nutrient cycling) (Figures 1 and 3).

Tier 3 - Based on ecosystem functional indicators documented during the assessment, the ecosystem functions being delivered at each project site were identified. For example, the presence of sediment deposits and stratified soil layers (i.e., repeating layers of mineral and organic soil materials) provides

Figure 3. Examples (from left) of ecological indicators linked with ecosystem functions including direct observations of faunal habitat use at Pointe Mouillee, MI; stratified soil layers indicate floodwater and sediment detention functions at Drake Wilson Island, FL; and field indicators of hydric soils (e.g., iron translocation) signify the retention and transformation of elements and compounds function at Bolivar Peninsula, TX.

evidence that energy dissipation and floodwater and sediment functions are occurring (USACE 2012). Similarly, the presence of organic-rich soil horizons and field indicators of hydric soils demonstrate that carbon sequestration, retention and transformation of elements and compounds, and nutrient cycling functions are occurring (USDA-NRCS 2018).

Tier 4 - The identified ecosystem functions were linked with ecosystem goods and services that benefit society using the framework described in Waigner et al. (2020). This approach has been used before, and ecological functional indicators have proven valuable for quantifying ecosystem functions (Berkowitz and White 2013) and allowing them to be coupled with ecosystem goods and services (McLaughlin and Cohen 2013).

RESULTS AND DISCUSSION

The historic habitat improvement projects constructed with dredged sediment successfully established each of the target habitats (Landin et al. 1989), and in general the target habitats have persisted for the past four decades (Berkowitz et al. 2021). While monitoring occurred for approximately 10 years following project construction, it focused on habitat improvement and the historic studies failed to evaluate the full suite of ecosystem functions occurring at the beneficial use sites. However, data from those studies in conjunction with the 2019 assessment of ecological indicators was used to document the wide array of ecosystem functions occurring at each project location (Table 3).

Results suggest that the beneficial use of dredged sediment yields positive ecological outcomes that are sustainable over periods exceeding 40 years at the study locations. This is significant because prior to construction of the dredged sediment beneficial use projects, the study locations provided very limited ecosystem functions with regard to habitat (Landin et al., 1989). As a result, the improvements induced by the projects represent a significant ‘lift’ in ecosystem functions (Yan et al., 2021). Further, this is one of the first studies to document the long-term delivery of ecosystem functions by dredged sediment beneficial use projects across a multi-decadal period. The ecosystem functions identified were assigned based on the various target habitats documented during the ecological assessment because different landforms are subject to different processes that induce ecological functions (Table 4; Berkowitz et al. 2021). For example, areas that do not come into contact with runoff and floodwaters lack the capacity to detain significant amounts of water and suspended sediments.

Table 3. Ecological functions observed at the six historic dredged sediment beneficial use projects and associated ecological indicators used to document each function.

Ecological functions	Ecological indicators
Physical functions	
Floodwater and sediment detention - the capacity of the ecosystem to temporarily store water and sediment following rain events, overbank flooding, & high tides.	Inundation and soil saturation, microtopographic relief, vegetation stem density, sediment deposits, stratified soil layers, soil bulk density
Energy dissipation - the capacity of the ecosystem to attenuate and decrease energy from wind and waves	Inundation and soil saturation, vegetation stem density, roughness, sediment deposits, water marks, drift deposits, algal mats
Export elements and compounds - the capacity of the ecosystem to export dissolved and particulate organic carbon, nutrients, sediment, and other materials to down-stream or down gradient areas	Inundation and soil saturation, water-stained leaves, soil organic matter content, drainage patterns, field indicators of hydric soils
Biogeochemical functions	
Nutrient cycling - The capacity of an ecosystem to convert nutrients from inorganic forms to organic forms and back through biogeochemical processes such as photosynthesis and microbial decomposition	Organic material production and storage, inundation and soil saturation, soil organic matter accumulation, field indicators of hydric soils
Retention and transformation of elements and compounds - the capacity of an ecosystem to temporarily or permanently store and transform metals, organic chemicals, and other substances through processes such as adsorption to soil particles, oxidation, reduction, and microbial degradation	Inundation and soil saturation, soil organic matter accumulation, field indicators of hydric soils, presence of reduced iron, oxidized rhizospheres along living roots
Sequester carbon - The capacity of an ecosystem to accumulate soil organic matter and store carbon, providing a long-term sink for greenhouse gases	Inundation and soil saturation, soil organic matter accumulation, below ground biomass, field indicators of hydric soils
Habitat functions	
Maintain habitat for wildlife, fisheries, and plant communities - the capacity of an ecosystem to provide the environment necessary to support the characteristic fish and wildlife species during part of their life cycles	Direct observations of faunal utilization, vegetative structural complexity, species richness and abundance, evidence of succession

The observed ecosystem functions occurring in each habitat type documented at the study locations (Table 4) were linked with ecosystem goods and services provided by the dredged sediment beneficial use projects using the relationships shown in Figure 4. Results indicate that the number and type of ecosystem functions, goods, and services delivered by each project depends on the distribution of habitats components created and the ecosystem functions occurring at those landforms (Table 5; Swanson et al. 1988). For example, the Bolivar Peninsula, TX project created four distinct target habitats that each provide for the maintenance of plants and animals habitats (functions) that contributes to ecosystem sustainability (goods and services). However, the herbaceous and shrubby uplands at that study location lack the soil organic matter characteristics and patterns of frequent inundation associated with the carbon sequestration and energy

Table 4. Summary of ecological functions occurring at each beneficial use project location.

Ecological functions	Study locations and target habitat types														
	Bolivar Peninsula, TX				Drake Wilson Island, FL			Buttermilk Sound, GA			Nott Isla., CT	Pointe Mou., MI	Miller Sands, OR		
	Low marsh	High marsh	Herbaceous upland	Woody upland	Low marsh	High marsh	Woody upland	Low marsh	High marsh	Unvegetated upland	Upland meadow	Freshwater marsh	Upland meadow	Tidal marsh	Dune
Floodwater and sediment retention	X	X			X	X		X	X			X		X	X
Energy dissipation	X	X			X	X		X	X			X		X	X
Export elements & compounds	X	X			X	X		X	X			X		X	
Nutrient cycling	X	X	X	X	X	X	X	X	X		X	X		X	X
Retention and transformation of elements and compounds	X	X			X	X		X	X			X		X	
Sequester carbon	X	X			X	X		X	X			X		X	
Maintain habitat for wildlife, fisheries, and plant communities	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X

dissipation functions as well as the associated climate regulation and hazard mitigation ecosystem goods and services categories (amongst others).

This highlights the interplay between landforms, ecosystem functions, and the delivery of ecosystem goods and services within a single project area. Additionally, the distribution of functions, goods, and services varied across beneficial use sites. For example, the results of the assessments at Bolivar Peninsula, TX and Miller Sands, OR indicate that projects designed to create a variety of landscape features, elevation, and patterns of inundation promote a higher diversity of ecosystem functions, goods, and services than projects that only contain a single geomorphological feature (e.g., the upland meadow at Nott Island, CT).

Results suggest that the establishment of marshes and wetlands using dredged sediment yielded a larger number of ecological functions than transitional or upland landscape features, and therefore deliver more categories of ecological goods and services. This occurs because wetlands are very productive ecosystems, subject to a higher degree of ecological dynamism (fluctuating water tables, floods) than other landforms, providing additional opportunities to confer positive societal outcomes (Costanza et al. 1989; Nyman 2011). Our findings align with the existing literature which reports that wetlands and other aquatic resources supply ecosystem functions, goods, and services at levels that exceed those of other habitats (Barbier 2013; Gunderson et al. 2016). The differences in the number of ecosystem functions, goods, and services should

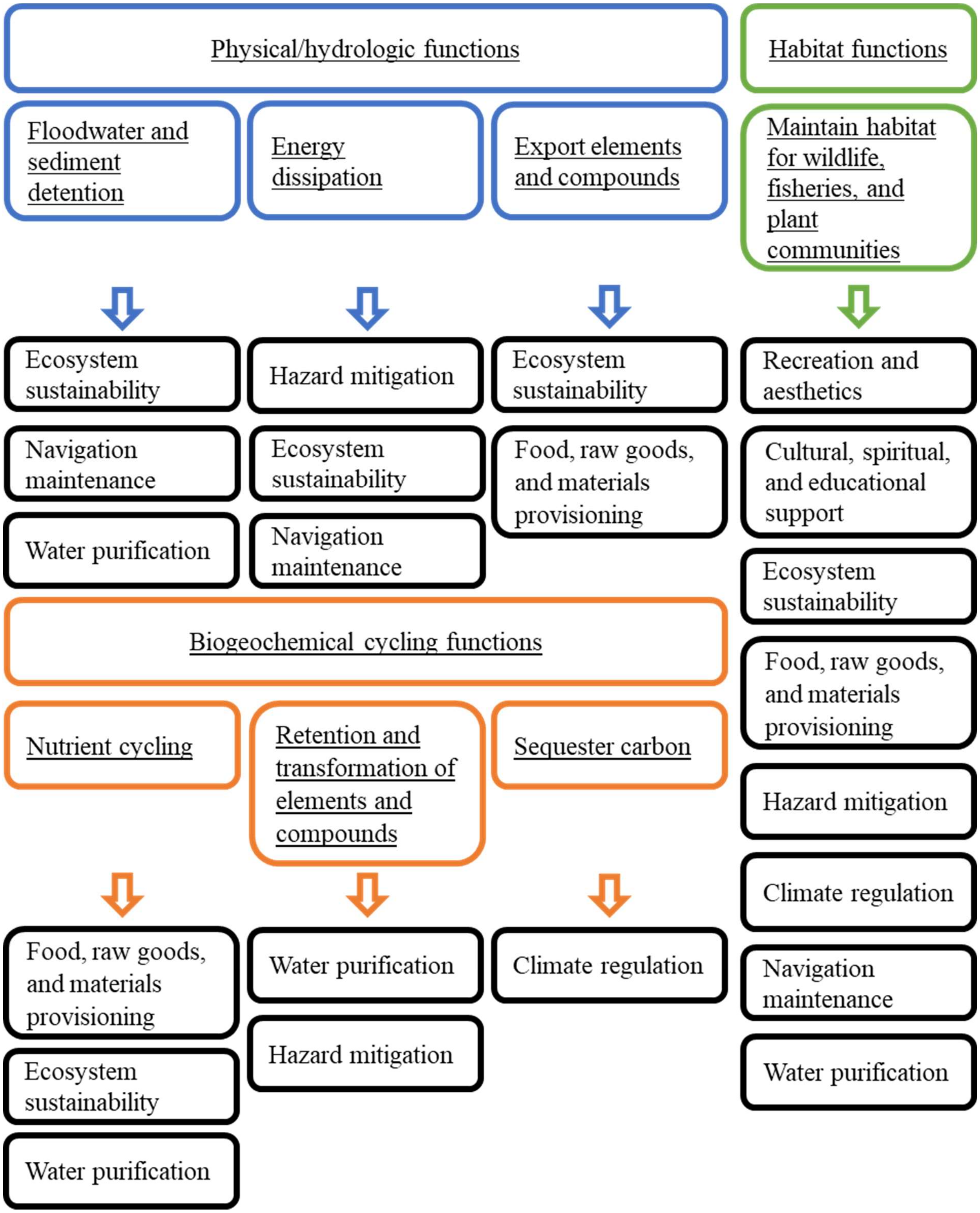

Figure 4. Relationship between ecological functions (underlined text in colored boxes) and resulting ecosystem goods and services (black boxes) that benefit society.

Table 5. Summary of ecosystem goods and services categories being delivered by each beneficial use project location.

Ecosystem goods and services categories	Study locations and target habitat types														
	Bolivar Peninsula, TX				Drake Wilson Island, FL			Buttermilk Sound, GA			Nott Isla., CT	Pointe Mou., MI	Miller Sands, OR		
	Low marsh	High marsh	Herbaceous upland	Woody upland	Low marsh	High marsh	Woody upland	Low marsh	High marsh	Unvegetated upland	Upland meadow	Freshwater marsh	Upland meadow	Tidal marsh	Dune
Ecosystem sustainability	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Hazard mitigation	X	X			X	X		X	X			X		X	
Navigation maintenance	X	X			X	X		X	X			X		X	
Cultural, spiritual, and educational support												X			
Recreation and aesthetics	X	X					X			X	X	X	X		
Food, raw goods, and materials provisioning	X	X						X	X			X		X	
Water purification	X	X			X	X		X	X			X		X	
Climate regulation	X	X			X	X		X	X			X		X	

not be interpreted to suggest that wetlands are 'better' than other landscape features in terms of potential beneficial use project outcomes, but instead highlights the fact that recognizing how ecosystem functions, goods, and services differ across the landscape can inform project design and management. Additionally, ecosystem components are not isolated and conditions or activities occurring in upland environments can impact the delivery of ecosystem functions, goods, and services in adjacent areas, including wetlands (Jones et al. 2018). As a result, management activities that apply regional or watershed perspectives (including dredging operations) are needed to maximize the delivery of ecosystem functions, goods, and services (Boerema and Meire 2017).

Notably, the observed relationships between the landforms created using dredged sediment and the delivery of ecosystem functions, goods, and services not only document beneficial use project outcomes, but also provides a mechanism to deliberately incorporate project design features that achieve specific environmental and societal objectives. For example, Berkowitz et al. (2016) demonstrated that biogeochemical and nutrient cycling (functions) and the associated improvements to water quality (goods and services) were maximized in dredged sediment beneficial use project features exposed to frequent inundation with carbon rich substrates. As such, projects seeking to remove excess nutrients for improved water quality should incorporate habitat components that mimic these environmental conditions. Similarly, Davis et al. (2021) linked ecosystem processes (i.e., sediment retention, habitat) with ecological goods and services (erosion, flood hazard mitigation) to quantify the flood risk reduction benefits of constructing barrier islands with dredged sediments, suggesting that specific ecological targets (surface elevation, percent cover of rooted vegetation) be incorporated into project designs and used to guide future management activities including additional dredged sediment deposition.

Finally, the results of the historic dredged sediment beneficial use assessment can also be used to develop conceptual ecosystem function trajectory curves to inform project life cycles (Figure 5; top panel). Previous studies demonstrate that the rate of ecological function delivery differs across functional categories, with physically derived functions delivering positive outcomes faster than biologically mediated processes (Meyer et al. 2008). For example, projects yield physical-process derived functions (energy dissipation)

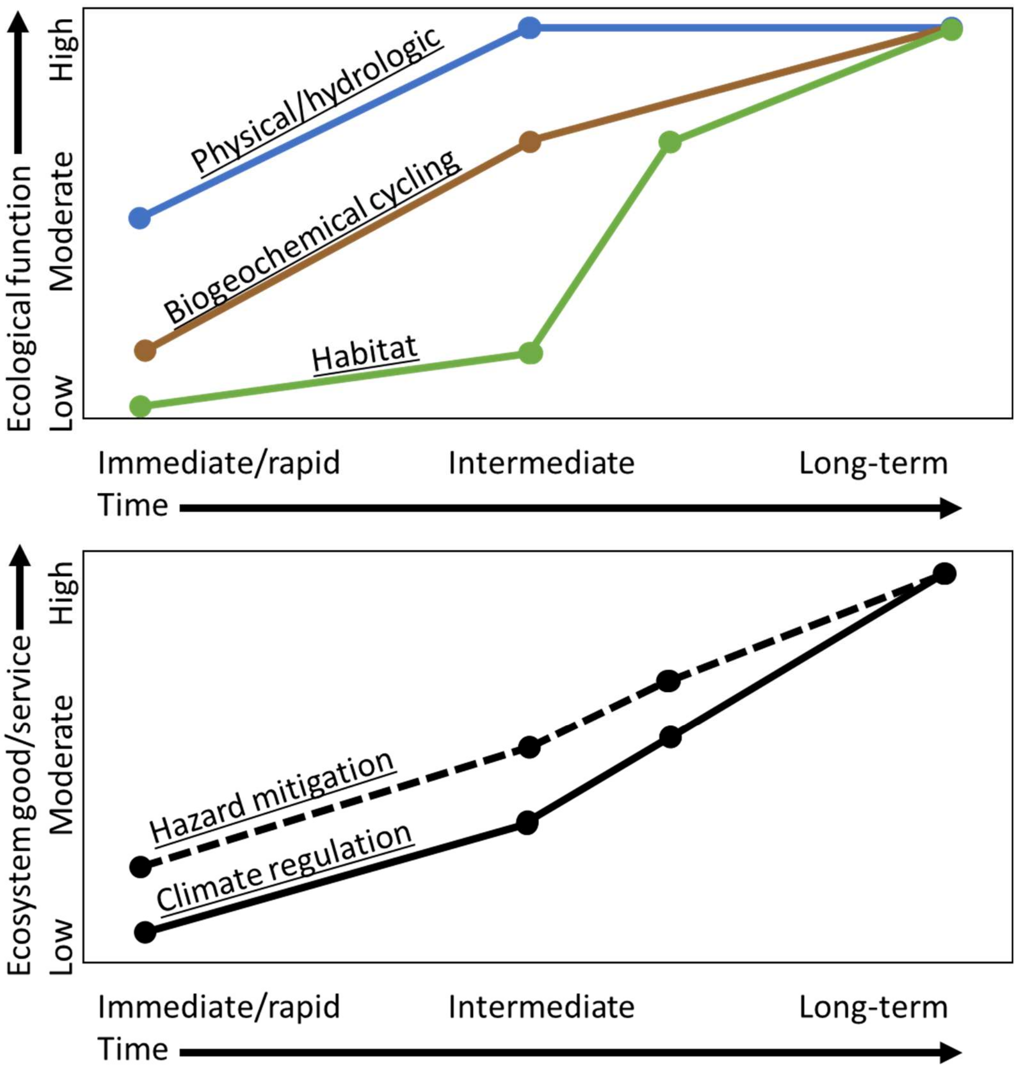

Figure 5. Conceptual trajectories for suites of ecological function (top panel) and two ecosystem services (bottom panel) following beneficial use project implementation.

immediately after construction, while most biogeochemical cycling and habitat functions require the accumulation of labile nutrients, microbial communities, and vegetative structures to become established (Berkowitz and White 2013). For example, dunes constructed using dredged sediment can dissipate energy immediately, while the development of mature forested habitats requires several decades (Davis et al. 2021; Berkowitz 2013).

Feedback loops alter the magnitude and rate of ecological functional trajectories over a project life cycle, especially with regard to biologically mediated process which experience threshold effects such as the establishment of woody vegetation or canopy closure. For example, the initial benefits delivered by dune establishment are enhanced with the expansion of beach grasses that fortify the features and help to entrap additional sediment over time (Feagan et al. 2015). Additionally, the shape of the trajectories differs across landform and habitat types as ecological succession occurs. For example, while unvegetated dredged sediment deposits provide valuable habitat for shore-nesting birds following project construction, habitat for those species declines as vegetation becomes established and forest growth induces a habitat shift toward species that require woody plants (Soots and Parnell, 1975).

Conceptual ecosystem goods and services trajectory curves can also be generated using the assessment results (Figure 5; lower panel). These curves are derived from the relative proportion of functional drivers depicted in the functional trajectory diagram, providing an example of how ecosystem functions, goods, and services are inter-related and can be linked to estimate anticipated changes over time. This approach provides a relative scale to estimate when, and to what extent, environmental and societal project objectives are likely to be realized. This has value because goods and services predominantly derived from physically driven ecosystem functions are likely to be delivered more rapidly and to a greater extent than outcomes associated with biologically mediated habitat and biogeochemical functions that take additional time to mature. For example, the hazard mitigation service is derived from a combination of physical, biogeochemical, and habitat functions (Figure 4) while the climate regulation service is predominantly associated with biogeochemical and habitat functions. As a result, the delivery of hazard mitigation benefits occurs faster and to a larger extent because of the influence of physical functions compared with climate regulation via soil carbon accumulation which is inherently biologically mediated (Berkowitz et al. 2021). Understanding and communicating these effects to practitioners and stakeholder groups can improve the perception of dredged sediment beneficial use projects by coupling ecological processes with societal objectives. While conceptionally derived, these curves provide a mechanism to link ecological functions and ecosystem goods and services in a systematic way that can be used to further promote specific project objectives, support alternatives analysis, and address uncertainties related to dredged material beneficial use project outcomes.

CONCLUSIONS

In order to address the economic and environmental challenges facing society in the coming decades, practitioners must focus on maximizing the available ecosystem functions, goods, and services that dredged sediment beneficial use projects can provide. Evaluating historic dredged sediment beneficial use projects is valuable because it provides a platform to document long-term project outcomes. Our findings suggest that linking ecological functions with an established ecosystem goods and services framework provides a mechanism to document the full suite of positive project environmental and societal outcomes provided by dredged sediment beneficial use projects. This approach will further promote continued innovation and help to offset increasing project construction costs, manage risk and uncertainty related to the use of natural processes and natural infrastructure, support holistic project life-cycle analysis, and effectively communicate project benefits to a variety of stakeholders. These analyses also assist with the quantification of the relative benefits delivered by specific design features or management activities and the trajectory of those benefits over time. As a result, we recommend that practitioners incorporate the concepts described herein into their projects. We anticipate that the proposed approach will be revised and improved iteratively as additional research quantifies and parameterizes models and other tools linking project features and management activities with changes in ecological functions, goods, and services.

REFERENCES

Barbier, E.B. (2013). "Valuing ecosystem services for coastal wetland protection and restoration: Progress and challenges," Resources, 2(3):213-230.

Baustian, J.J. and Eugene Turner, R. (2006). "Restoration Success of Backfilling Canals in Coastal Louisiana Marshes," Restoration Ecology, 14: 636-644. https://doi.org/10.1111/j.1526-100X.2006.00175.x

Berkowitz, J.F. (2013). "Development of restoration trajectory metrics in reforested bottomland hardwood forests applying a rapid assessment approach," Ecological indicators 34:600-606.

Berkowitz, J.F., and White, J.R. (2013). "Linking wetland functional rapid assessment models with quantitative hydrological and biogeochemical measurements across a restoration chronosequence." Soil Science Society of America Journal, 77(4):1442-1451.

Berkowitz, J.F., Green, L., VanZomeren, C.M. and White, J.R. (2016). "Evaluating soil properties and potential nitrate removal in wetlands created using an engineering with nature based dredged material placement technique," Ecological Engineering 97:381-388.

Berkowitz, J.F., Beane, N.R., Hurst, N.R., Jung, J.F., and Philley, K.D. (2021). "An assessment of Long-Term, Multipurpose Ecosystem Functions and Engineering Benefits Derived from Historical Dredged Sediment Beneficial Use Projects," Vicksburg, MS: US Army Engineer Research and Development Center Technical Report ERDC/TR-21-4.

Bianchi, T.S., DiMarco, S.F., Cowan Jr, J.H., Hetland, R.D., Chapman, P., Day, J.W. and Allison, M.A. (2010). "The science of hypoxia in the Northern Gulf of Mexico: a review," Science of the Total Environment 408(7):1471-1484.

Boerema, A. and Meire, P. (2017). "Management for estuarine ecosystem services: a review," Ecological Engineering, 98:172-182.

Bouwma, I., Schleyer, C., Primmer, E., Winkler, K.J., Berry, P., Young, J., Carmen, E., Špulerová, J., Bezák, P., Preda, E. and Vadineanu, A. (2018). "Adoption of the ecosystem services concept in EU policies," Ecosystem Services 29:213-222.

Bridges, T.S., Lillycrop, J., Wilson, J.R., Fredette, T.J., Suedel, B., Banks, C.J., and Russo, E.J. (2014). "Engineering with Nature Promotes Triple-Win Outcomes," Terra Et Aqua. 135:17-23.

Britsch, L. D., & Dunbar, J. B. (1993). "Land loss rates: Louisiana coastal plain," Journal of Coastal Research, 324-338.

Brown, T.C., Bergstrom, J.C. and Loomis, J.B. (2007). "Defining, valuing, and providing ecosystem goods and services," Natural Resources Journal 47(2):329-376.

Central Dredging Association. (2013). "Ecosystem Services and Dredging and Marine Construction," CEDA Information Paper, May 2013.

Chesney, E.J., Baltz, D.M. and Thomas, R.G. (2000). "Louisiana estuarine and coastal fisheries and habitats: perspectives from a fish's eye view," Ecological Applications, 10(2):350-366.

Costanza, R., Farber, S.C., and Maxwell, J. (1989). "Valuation and management of wetland ecosystems," Ecological economics 1(4):335-361.

Daily, G.C., Söderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P.R., Folke, C., Jansson, A., Jansson, B.O., Kautsky, N. and Levin, S. (2000). "The value of nature and the nature of value," Science 289(5478):395-396.

Davis, J., Whitfield, P., Szimanski, D., Golden, B.R., Whitbeck, M., Gailani, J., Herman, B., Tritinger, A., Dillon, S.C. and King, J. (2022). "A framework for evaluating island restoration performance: A case study from the Chesapeake Bay. Integrated Environmental Assessment and Management," 18(1):42-48.

Davis, J., P. Whitfield, D. Szimanski, B. R. Golden, M. Whitbeck, J. Gailani, B. Herman, A. Tritinger, S. C. Dillon, and J. King. (2021). "A Framework for Evaluating Island Restoration Performance: A Case Study from the Chesapeake Bay." Integrated Environmental Assessment and Management. https://doi.org/10.1002/ieam.4437.

De Groot, R.S., Wilson, M.A., and Boumans, R.M. (2002). "A Typology for the Classification, Description and Valuation of Ecosystem Functions, Goods and Services," Ecological Economics 41(3):393–408.

Feagin, R.A., Figlus, J., Zinnert, J.C., Sigren, J., Martínez, M.L., Silva, R., Smith, W.K., Cox, D., Young, D.R. and Carter, G. (2015). "Going with the flow or against the grain? The promise of vegetation for protecting beaches, dunes, and barrier islands from erosion," Frontiers in Ecology and the Environment, 13(4):203-210.

Foran, C.M., Burks-Copes, K.A., Berkowitz, J., Corbino, J. and Suedel, B.C. (2018). "Quantifying Wildlife and Navigation Benefits of a Dredging Beneficial-Use Project in the Lower Atchafalaya River: A Demonstration of Engineering with Nature®," Integrated environmental assessment and management 14(6):759-768.

Gómez-Baggethun, E., De Groot, R., Lomas, P.L. and Montes, C. (2010). "The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes," Ecological economics 69(6):1209-1218.

Gunderson, L. H., Cosens, B., and Garmestani, A.S. (2016). "Adaptive governance of riverine and wetland ecosystem goods and services," Journal of environmental management, 183:353-360.

International Association of Dredging Companies. (2013). "Facts About Ecosystem Services & Dredging," An Information Update from the IADC, 4.

Jenkins, W.A., Murray, B.C., Kramer, R.A., and Faulkner, S.P. (2010). "Valuing ecosystem services from wetlands restoration in the Mississippi Alluvial Valley," Ecological Economics 69(5):1051-1061.

Jones, C.N., McLaughlin, D.L., Henson, K., Haas, C.A. and Kaplan, D.A. (2018). "From salamanders to greenhouse gases: does upland management affect wetland functions?" Frontiers in Ecology and the Environment, 16(1):14-19.

Kolman, R. (2014). "Introducing ecosystems services for port development," Environment and Sustainability, 62:181-183.

Landin, M.C., Webb, J.W., and Knutson, P.L (1989). "Long-Term Monitoring of Eleven Corps of Engineers Habitat Development Field Sites Built of Dredged Material, 1974–1987," Vicksburg, MS: U.S. Army Engineer Waterways Experiment Station Technical Report D-89-1.

McLaughlin, D.L. and Cohen, M.J. (2013). "Realizing ecosystem services: wetland hydrologic function along a gradient of ecosystem condition," Ecological Applications, 23(7):1619-1631.

Meyer, C.K., Baer, S.G. and Whiles, M.R. (2008). "Ecosystem recovery across a chronosequence of restored wetlands in the Platte River Valley," Ecosystems, 11(2):193-208.

Novitski, R.P., Smith, R.D. and Fretwell, J.D. (1996). "Wetland functions, values, and assessment," National Summary on Wetland Resources, USGS Water Supply Paper, 2425:79-86.

Nyman, J.A. (2011). "Ecological functions of wetlands". Wetlands:115-128. Springer, Dordrecht.

R. Daniel Smith, Chris V. Noble, and Jacob F. Berkowitz (2013). "Hydrogeomorphic (HGM) Approach to Assessing Wetland Functions: Guidelines for Developing Guidebooks (Version 2)," ERDC/EL TR-13-11, Environmental Laboratory, Engineer Research and Development Center, Vicksburg, MS.

Soots Jr, R.F. and Parnell, J.F. (1975). "Ecological succession of breeding birds in relation to plant succession on dredge islands in North Carolina," University of North Carolina Sea Grant Publication UNC-SG-75-27.

Swanson, F.J., Kratz, T.K., Caine, N. and Woodmansee, R.G. (1988). "Landform effects on ecosystem patterns and processes," BioScience, 38(2):92-98.

Turner, R.E. and McClenachan, G. (2018). "Reversing wetland death from 35,000 cuts: Opportunities to restore Louisiana's dredged canals," PloS one, 13(12).

United States Army Corps of Engineers (USACE). (2012). "Regional supplement to the Corps of Engineers wetland delineation manual: North Central and Northeast region (Version 2.0), ed. J.F. Berkowitz, R. Lichvar, C.V. Noble, J.S. Wakeley. Vicksburg, MS: US Army Engineer Waterways Experiment Station Technical Report ERDC/EL TR-12-1.

United States Department of Agriculture, Natural Resources Conservation Service (USDA-NRCS). (2018). "Field Indicators of Hydric Soils in the United States, Version 8.2". L.M. Vasilas, G.W. Hurt, and J.F. Berkowitz (eds.). USDA-NRCS, in cooperation with the National Technical Committee for Hydric Soils.

Wainger, L.A., McMurray, A., Griscom, H.R., Murray, E.O., Cushing, J.A., Theiling, C.H. and Komlos, S.B. (2020). "A proposed ecosystem services analysis framework for the US Army Corps of Engineers," Vicksburg, MS: US Army Engineer Research and Development Center Technical Note ERDC/El SR 20-2.

Wellman, K., and Gregory, R. (2002). "Trade-off Analysis for the Use of Environmental Windows," Journal of Dredging, 4(2).

Yan, N., Liu, G., Xu, L., Deng, X. and Casazza, M. (2021). "Energy-based eco-credit accounting method for wetland mitigation banking," Water Research, 210.

ACKNOWLEDGEMENTS

Funding for this study was provided by the U.S. Army Corps of Engineers Dredging Operations Environmental Research Program. We thank the following USACE collaborators who made this study possible by assisting with site access and laboratory analysis: Mr. Jose Vargas (Galveston District); Mr. Richard Allen (Mobile District); Mr. Glenn Bacon (Savannah District); Dr. Kyle Tidwell (Portland District); Mr. Todd Randall (New England District); and Ms. Christina Saltus and Ms. Sydney Bufkin (Engineer Research and Development Center). The authors also thank three peer reviewers and the handling editor for providing insightful and valuable feedback.

AIMS AND SCOPE OF THE JOURNAL

WEDA's peer-reviewed papers in this practice-oriented journal present solutions to dredging, dredged material placement, and related issues. Submissions on all related topics are of interest to our readers. Examples include, but are not limited to, dredging techniques, hydrographic surveys, dredge automation, dredge safety, instrumentation, design of dredging projects, dredged material placement, transport of sediment through pipes, design of dredging equipment, dredging software, physical modeling of equipment and processes, cost estimating, environmental dredging, beneficial uses, Engineering with Nature®, contaminated sediments, litigation, economic aspects, and case studies.

The *Journal of Dredging* is published by the Western Dredging Association (WEDA) to disseminate technical and project information on all dredging-related topics to the broadest possible audience. In addition to the Journal, WEDA sponsors annual technical conferences covering the Western Hemisphere and smaller regional conferences focused on more geographically relevant topics. WEDA collaborates with the Central Dredging Association (CEDA) and Eastern Dredging Association (EDA) to hold the World Dredging Congress (WODCON) every three years. Historical conference proceedings are available at http://westerndredging.org.

NOTES FOR CONTRIBUTORS

The Journal of Dredging welcomes technical papers from all prospective authors regardless of employment or location. Editorials and commentaries will be considered provided they are fact-based and of interest to the Journal readership. Published manuscripts will not contain evident commercialism or promote private or political interests.

Submission of a manuscript implies that it is not under consideration for publication elsewhere and that original, previously unpublished work is being presented. Authors should obtain all necessary approvals for publication from employers or others involved before submitting a paper.

Papers should be concisely written in U.S. English, free of grammatical and spelling errors, and contain appropriate illustrations, figures, and equations. Papers must be appropriately cited to show that the work is scientifically sound and based upon previously published scientific literature. Published manuscripts will not exceed 20 total pages. Appendices are not allowed.

Manuscript preparation information, formatting requirements, and submission information can be found at https://www.westerndredging.org/journal. Authors are encouraged to use the Microsoft Word template provided on that webpage to facilitate proper manuscript formatting.

Authors will be responsible for producing the final, print-ready versions of the paper and providing them to the journal in PDF format unless other arrangements are made in advance with the Editor. The journal will be reproduced directly from the camera-ready manuscripts supplied by the authors.

Copyright of published papers will be the property of the Western Dredging Association. Authors are granted permission to reprint the paper, provided the copyright information is retained.

Send all manuscript and journal-related questions to journal@westerndredging.org.

LEARN MORE ABOUT THE WESTERN DREDGING ASSOCIATION AT

www.WesternDredging.org

WEDA EMAIL ADDRESS

Info@WesternDredging.org

Dredging Creates a Strong Economy and Cleaner Environment