EFFECTS OF SEDIMENTATION ON WINTER FLOUNDER EGGS IN LABORATORY EXPERIMENTS

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ABSTRACT

Dredging activities may result in suspension and subsequent settling of sediment near fish spawning habitat; this may bury fish eggs, potentially affecting local fisheries. The objective of this study was to determine the effects of sedimentation depth on the viability of winter flounder (*Pseudopleuronectes americanus)* eggs incubated under laboratory conditions. Experiments were conducted using sediment collected from three routinely dredged Connecticut waterways. Survival (averaged across the three sediments) at nominal sedimentation depths of 0.3, 0.6, and 1.2 mm was 86.6, 81.1 and 73.9%, respectively, and not significantly reduced relatively to the control. Survival was significantly reduced at the 3.0 mm sediment depth for hatched eggs across the three sediments. Results of the current study corroborated closely with prior research on winter flounder egg burial hatch success and demonstrated that egg performance was consistent across the three sediments evaluated. The results suggest that a flounder egg (≈ 0.75 mm diameter) may be 80% buried (0.6 mm) without substantially affecting hatch success. Combined data indicated low or no effects (EC5) on both hatching success and normal embryo development at less than 1.06 mm depth. Before sedimentation risk can be appropriately assessed and managed, field measurements of the incremental contribution to sediment layers attributable to dredging operations, as well as other environmental factors such as timing of burial (with respect to egg development) and local hydrodynamic conditions need to be considered.

Keywords: Dredging, winter flounder (*Pseudopleuronectes americanus)*, sedimentation, risk.

INTRODUCTION

Dredging operations can resuspend sediment, creating a plume in the water column which is transported away from the dredge by ambient conditions and eventually settles to bottom substrates. Assessment of the plume and its associated sedimentation is important when complying with regulations dictated by environmental windows (EW). EWs restrict dredging operations to specified time periods to minimize exposure of nearby sensitive biota or habitats (Reine et al. 1998; Suedel et al. 2008). EWs are most commonly established to prevent potential impact to sport and anadromous fish (Reine et al. 1998). The U.S. Army Corps of Engineers New England District (NAE) is subject to EWs in numerous waterways where dredging operations overlap with winter flounder (*Pseudopleuronectes americanus*; WFL) spawning areas. Other aquatic species are also protected by EWs in the same waterways, which compounds the limitations by expanding the time window associated with multiple EWs, resulting in dredging being restricted to a few months of the year. While these are day-to-day practices in regulating dredging activities, there is limited information concerning species-specific biological effects relevant to resuspended sediment or sedimentation, which questions the use of EWs as an effective management practice.

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A primary concern for dredges working in New England waterways is the overlap of dredging operations with WLF spawning areas, which can potentially result in sedimentation of spawned eggs and attribute to egg mortality. Many naturally spawning fish populations experience high levels of mortality during the egg stage. This may be due to angling, predation, poor spawning substrate, severe storm or wind and wave action dislodging eggs, and fungus (e.g., Johnson 1961; Scott and O'Bier 1962; Dahlberg 1979; Steinhart et al. 2005). Additionally, excessive natural or anthropogenic sedimentation may increase egg mortality by burying the eggs (Ventling-Schwank and Livingstone 1994; Berry et al. 2011). Most egg burial research has focused on salmonids, so there is a paucity of data for many other fish species. Various investigators have shown that different grain sizes of sediment can affect egg survival differently and sedimentation of finer-grained sediment particles (<0.85 mm) can be the most detrimental to fish eggs (Reiser and White 1988; Greig et al. 2005; Jensen et al. 2009; Bowerman et al. 2014). The organic matter in the sediment can decompose and use dissolved oxygen. The interstitial water associated with burial may also affect the rate of transfer of dissolved substances into, or from the sediment which can effect egg respiration leading to development deficiencies and mortality (Silver et al. 1963).

The U.S. Army Engineer Research and Development Center, Environmental Laboratory (ERDC-EL) performed a series of laboratory experiments to evaluate the effects of sedimentation on WFL eggs, a primary concern for dredges operating in New England waterways. These effects-based data will be used to help evaluate the EWs as an effective management practice for protecting WFL populations in the region.

METHODS

Winter Flounder

WFL eggs approximately 48 h post-fertilization were obtained from the University of New Hampshire (UNH), Durham, NH, where they were manually spawned from brood stock collected from the Gulf of Maine. WFL is federally managed as two inshore stocks that include the Gulf of Maine, Southern New England/Middle Atlantic (SNE/MA), and one offshore stock at Georges Bank (Pereira et al 1999). A simple paired mating strategy was used in which one male was crossed with one female with no individual used more than once. This strategy was used to produce three unique families for testing. To prevent clumping of the eggs, a mixture of test sediment collected from NAE waterways (25 ml sediment mixed into 500 ml of seawater) was introduced into the hatching cone where the eggs and sediment were gently mixed by aeration for 45 min. Test sediment collected from three navigation channels (see test sediment section) was used for declumping to simulate field conditions. Eggs were declumped because individual eggs were required to create a monolayer when buried during the experiment. Each batch of eggs was treated and tested with the same test sediment. Eggs were shipped priority overnight in a cold shipping container to maintain hatchery water temperatures (5-8.5°C) to the ERDC-EL, Vicksburg, MS. When the eggs were received the water temperature ranged from 2.7-4.8 C, dissolved oxygen 33-42 mg/L, pH 7.1-7.2 and salinity 28-30 ppt. The eggs were acclimated to laboratory water for 1 h before being used in the experiment.

Test Sediment

This study's objective was to study the physical (rather than chemical) effects of suspended sediment. Therefore, sediment was collected at Federal navigation channel locations with a known history of low chemical contamination. Test sediments were collected in the Long Island Sound from shoaled areas in navigation channels located in the Patchogue River near Westbrook, CT, Milford Harbor near Milford, CT, and Mianus River, near Mianus CT. These areas are routinely dredged (at least once every 5 years) in shallow water areas where WFL may spawn.

Sediments were collected using a grab sampler, placed in plastic buckets, shipped to the ERDC-EL and stored at 4^{o}C until use. In the laboratory, the samples were composited, mechanically homogenized, and then sieved (through 1 cm) to remove large debris. Immediately prior to sieving the sediment was subsampled and submitted for chemical and physicochemical analyses. Physicochemical analyses consisted of grain size, total organic carbon, organic matter, pH, moisture content, and cation exchange capacity. Chemical analyses of sediment was conducted using USEPAs Hazard Waste Solid Waste Test Methods (SW-846); USEPA Method 8082 for PCB Aroclors, USEPA Method 8081A for organochlorine pesticides, USEPA Method 8270C for semi-volatile compounds, and USEPA 6000/7000 series metals (USEPA 2013). Grain size analysis was also performed on each sediment post-processing.

A methodology was developed simulating dredge plume settling in the laboratory in lieu of wet sieving, utilizing aspects of the dredge elutriate preparation (1 part sediment/4 parts lab water; USEPA/USACE 1998) and the hydrometer method (Bouyoucos 1962) for measuring grain size to develop a means of separating sand and fine-grained (silts and clays) to obtain the desired grain size for use in testing. To begin, 1 L of wet sediment was mixed with 4 L of lab water in a clean 7.6 L "mixing" bucket. The mixture was agitated by manually stirring for 1 minute. Stoke's Law equation was used to calculate the settling velocity (m/s) of the particle size of interest based on diameter and density of the particle, water temperature, density and viscosity of water, and acceleration due to gravity. After settling was complete, the "mixing" bucket was slowly decanted by pouring the unsettled portion (supernatant) into a 19 L "holding" bucket. Care was taken to minimize decanting the settled material in the mixing bucket. The supernatant was then allowed to settle in the holding bucket for 24 hours.

Sedimentation Experiments

The Fish Larvae and Egg Exposure System (FLEES) at the ERDC-EL, Vicksburg, MS was used to conduct the experiments (Lutz et al. 2012). Eggs from UNH were exposed separately to Patchogue, Mianus and Milford sediments and were approximately 48 h old at the beginning of each experiment. Eggs were buried within 48 h of their spawn date to simulate a worst-case scenario in situ (eggs buried throughout most of the egg life stage with no energy to remove sediment off eggs). Eggs were covered under four sediment depths (0.3, 0.6, 1.2, and 3 mm) and control (0 mm) in the FLEES laboratory (lighting 12:12-hour light-to-dark photoperiod) until hatching was completed. Six replicates of the control and each depth treatment (30 chambers per sediment; total 90 chambers) were arranged randomly in three water baths. Hatching was complete if no larval fish was observed 72 h after the last larval fish was observed.

Exposure Chamber

Eggs were covered in an exposure chamber 6.35 cm diameter (internal diameter) by 10.16 cm long made of clear acrylic pipe (Figure 1). The pipe was attached to a 7.62 cm^2 base of clear acrylic sheet. A drain (1.27 cm diameter) was drilled 1.27 cm from the top of the chamber to maintain a volume of approximately 260 ml. Rubber bands (Castration Bands, Neogen Corporation; Lansing, MI) were used to install an elbow tube fitting (1.27 cm hose by 1.27 cm hose, inside diameter, black high-density polyethylene [HDPE] elbow, United States Plastic Corporation; Lima, OH) in the overflow drain with the elbow extending in to the chamber. Silicone was used to attach a 150 µm screen (Nylon screen, Pentair Aquatic Eco-Systems; Sanford, NC) onto the end of the elbow located inside the chamber to prevent escape. The water stream entering the chamber was deflected by a 2.54 cm^2 acrylic sheet that was submerged just beneath the water surface; thereby preventing disturbance of the sediment layer. The treatments and controls were arranged randomly in the FLEES.

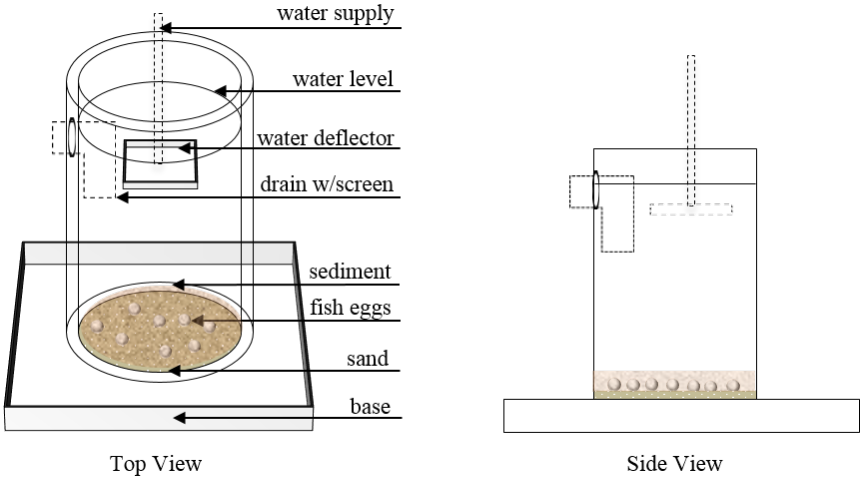

Figure 1. Top and side views of an exposure chamber and components.

Creating the Sediment Layer

To simulate field conditions (Wilber et al. 2013), a sand substrate was used (Fisher Scientific) with grain size ranging from 0.1 to 0.5 mm. The day prior to the experiment, a ceramic spoon was used to place sand in each chamber. The sand was introduced into each chamber filled a third full with test water. A vortex lab mixer was used to visually level the sand, achieving a roughly 1 mm substrate upon which to place the eggs. After leveling, the average sand depth was determined by completing eight measurements within each chamber. The chambers were then stored in a refrigerator (4°C) overnight until needed for the experiment the next day. Prior to the experiment, sediment treatment depths were created by establishing a correlation between sediment wet weights (to nearest 0.0001 g) and sediment depths (to nearest 0.01 mm) in the exposure chambers. A simple linear regression was used to model the relationship and make predictions about sediment depth to achieve desired depths. The day prior to the experiment, test sediment was homogenized, then a stainless steel spatula was used to place sediment in a plastic weighing dish. After weighing, the contents in the dish were rinsed into a plastic centrifuge tube (50 ml) with test water and stored in a refrigerator (4°C) overnight.

Next, an exposure chamber containing sand substrate and water was removed from the refrigerator and placed in an ice bath. A sample of eggs were transferred to a small petri dish filled with test water and viewed under a stereo microscope to determine viability (i.e., clear with visible cell division). A total of 50 eggs were suctioned from the Petri dish using a small glass Pasteur pipette and then stocked into the exposure chamber using care to not disturb the sand layer. The centrifuge tube containing the sediment depth treatment was removed from the refrigerator and mechanically mixed on a vortex lab mixer to create a slurry. To prevent disturbing the sand substrate, a layer of 1.25 cm plastic thermal balls was placed on the water surface in the exposure chamber and used to diffuse the slurry's energy as it was introduced. To introduce the slurry, a syringe (without a needle) was clamped on a stand and positioned over the exposure chamber, over a thermal ball. The slurry was then rinsed with test water into the syringe which introduced the slurry over the thermal balls, thereby diffusing the slurry into the chamber without disturbing the sand. A quiescent environment was necessary after the slurry was added to an exposure chamber to achieve an even sediment layer at the bottom of the chamber. To reduce vibration when the sediment was initially settling, the chambers were placed back into the refrigerator (4°C) for the first 24 hours on a level anti-vibration platform.

Measuring Sediment Depth

After 24 h, the thermal balls were removed from the water surface and the sediment depth was measured using a stainless steel digital caliper (Absolute Digital Calipers, Mitutoyo, Aurora, IL, USA) with an attached fine metal point. To measure, the exposure chamber was placed on a countertop, the caliper clamped on a stand and positioned over the chamber, and then lowered until it was submerged about halfway into the chamber (2-3 cm above the sediment surface). Next, the caliper fine adjustment knob was used to slowly lower the caliper depth plunger until it touched the sediment surface. The caliper was then zeroed and the fine adjustment knob used to slowly lower the plunger through the sediment and sand until it touched the chamber bottom. The resulting caliper reading represented the total depth (sand + sediment) and was recorded. To determine the average total depth eight measurements were taken within each chamber. To determine the sediment depth, the average sand depth measurement (measured the same way in each chamber the previous day) was subtracted from average total depth measurement for each chamber. After measurement, the chambers were placed in their appropriate water baths for the remainder of the experiment.

Water Delivery

The FLEES is equipped with a data acquisition device and LabVIEW software (National Instruments; Austin, TX) that automates water flow. Exposure chambers were partially submerged into temperature controlled FLEES water baths after sediment settled for 24 h on the anti-vibration platform in the refrigerator. A Zumwalt water delivery apparatus (Zumwalt et al. 1994) was used to deliver water introduced by FLEES into eight exposure chambers simultaneously. The Zumwalt apparatus is a glass, water-splitting chamber with eight holes drilled in the glass bottom. Silicone stoppers are drilled with a core borer and used to hold 50 ml polypropylene syringes in place in the glass bottom. Water was dispensed through 16 gauge, 1.5 in. needles with the sharp points removed. For each needle, a small hole was drilled near the top of the needle cover, and then the cover was placed back over the needle.

A 2.54 cm^2 piece of flat acrylic sheet was cut, and then a hole large enough to friction fit the needle cover was drilled in the middle. The needle cover was pushed through the hole until it extended just beyond the surface of the opposite side. Silicone was used to glue the cover in place. The needle and cover were installed onto the syringe. The Zumwalt apparatus was attached to threaded stainless steel rods and lowered until the acrylic square was submerged just beneath the water surface in the chamber. The Zumwalt apparatus was leveled by adjusting the nylon nuts on the stainless steel rods. Water introduced into the Zumwalt apparatus exited the syringe through the needle cover and was diffused by the deflector plate. This effectively prevented the sediment layer from being disturbed during the experiment. Twelve Zumwalt delivery apparatus (four per water bath) were used simultaneously to perform the sedimentation experiments.
Water was introduced to equal the volume of the chamber at least 6-7 times per day. Three exposure chambers were randomly selected from each treatment of each sediment on days 3 and 9 to create a sediment oxygen profile with the Unisense Microprofiling System (Unisense A/S, Denmark). The system allowed for precise positioning and moving of a dissolved oxygen microelectrode through the sediment at 100 µm increments. Until hatching occurred, dissolved oxygen, pH, salinity and temperature were monitored daily, approximately 3 cm above the sediment surface.

Data Analysis

Larvae were removed at least once per day, beginning at the first sign of hatching. If larval fish were observed, the chamber was removed from the water bath without disturbing the sediment layer and placed on a light box where the fish were suctioned using a 5 ml plastic transfer pipette. The fish were transferred to a Petri dish for observation and the chamber was returned to the experiment. Abnormal larvae were those that did not uncoil or straighten out after hatch or were scoliotic. Dead larval fish were pale in appearance with no heart movement. Live larval fish were transparent with some pigmentation and a visible heart movement (Figure 2). When no larvae were observed 72 h after the last larval fish was removed, the chamber was terminated.

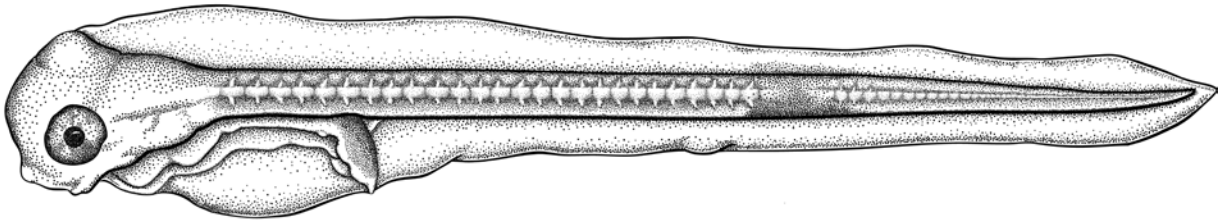

Figure 2. Newly hatched winter flounder larvae showing visible pigmentation and heart (Drawing by Cathryn Ososky).

RESULTS

Sediment Characterization

Grain size analysis results showed all test sediments consisted of less than 22% sand (Table 1). Patchogue and Mianus River sediments consisted primarily of silts, 67% and 49%, respectively. Milford Harbor sediment had 45% silt and 42% clay. The results of the chemical analysis indicated PCBs (as Aroclors) and pesticides in the sediments were below detection limits. All 17 RCRA metals were detected in the sediments. Of the PAHs, only fluoranthene and pyrene were detected. To assess chemical contamination, the detected concentrations were compared to appropriate sediment quality guidelines. Concentrations for metals (copper, lead, mercury, nickel) and PAHs (fluoranthene, pyrene) were below concentrations that could cause potential harm (NOAA ERL & ERM: NOAA Screening Quick Reference Tables; Buchman 2008). There were no screening criteria for barium, beryllium, manganese, selenium, thallium, and vanadium.

Table 1. Grain size analysis for sediments collected from Patchogue River, Mianus River and Milford Harbor after coarse and medium sized sand particles were removed.

	Particle Size (%)		
Sediment	**Sand**	**Silt**	**Clay**
Patchogue River	17	67	15
Mianus River	21	49	30
Milford Harbor	13	45	42

Sediment Depth

The measured sediment depths after 24 h of settling were within the expected range of the targeted burial depths (Table 2). Variability was low providing evidence for a relatively uniform sediment layer. On average, the chambers were removed 22 ± 2 days after the experiment started. Sediment settled during the experiment resulting in a percentage decrease in depth ranging from -3 to -31%.

Table 2. Average (standard deviation) sediment depth (mm; 8 measurements per replicate; N=48 measurements per treatment) after 24 h of settling and at the end of the experiment. Average sand depth was measured the day prior to the start of each experiment.

Sediment	Nominal Depth	24 h Depth	Settle Time (days)	End Depth	Change (%)	Sand Depth
Mianus River	0	0	21	0	0	1.04 (0.04)
	0.3	0.25 (0.06)	27	0.22 (0.06)	-13	1.08 (0.06)
	0.6	0.46 (0.03)	25	0.37 (0.03)	-19	1.07 (0.04)
	1.2	1.22 (0.08)	23	0.94 (0.06)	-23	1.00 (0.04)
	3	2.76 (0.07)	22	2.11 (0.09)	-24	1.05 (0.04)
Milford Harbor	0	0	20	0	0	1.06 (0.05)
	0.3	0.27 (0.04)	22	0.18 (0.06)	-31	1.05 (0.07)
	0.6	0.51 (0.07)	22	0.4 (0.05)	-23	1.05 (0.04)
	1.2	1.05 (0.05)	21	0.75 (0.08)	-28	1.08 (0.05)
	3	2.81 (0.06)	21	2.19 (0.06)	-22	1.08 (0.03)
Patchogue River	0	0	20	0	0	0.98 (0.07)
	0.3	0.35 (0.03)	24	0.34 (0.05)	-3	1.01 (0.05)
	0.6	0.55 (0.04)	24	0.52 (0.08)	-6	1.00 (0.04)
	1.2	1.31 (0.07)	23	1.08 (0.07)	-17	0.99 (0.04)
	3	3.01 (0.06)	20	2.54 (0.05)	-16	0.99 (0.05)

Water Quality

Water temperatures averaged 5.2°C (range 3.3-6.3°C) during the experiment. The average pH and salinity was 7.96 and 30.92 ppt, respectively. These conditions are within normal ranges to successfully hatch WFL eggs as was evidenced by an average 86.2% control hatch in this experiment (Williams 1975). Dissolved oxygen above the sediment layer (DO) averaged 6.94 mg/L and ranged from 0.88 to 10.94 mg/L. Factors likely influencing the DO in this experiment included (1) the depth of the fine test sediments; (2) penetration of surface water into the sediment; (3) the biological or chemical oxygen demand by the test sediments and the associated organic fraction; and (4) water volume exchange in the chamber. DO in the test water entering the exposure chambers was always greater than 8 mg/L during the course of the experiment. DO profiles acquired showed a noticeable decrease with depth only for the 3 mm treatment. Only the Patchogue sediment on day 3 showed a larger DO consumption close to the water-sediment interface for the 3 mm treatment. All other treatments, regardless of day, showed no noticeable decrease with test sediment depth. This suggested overlying water was penetrating the sediment; therefore, a combination of consumption of DO by the sediment and water exchange rate were likely contributors to the observed decrease in DO in chambers.

Sedimentation Effects

The average percent hatch for the control groups was 86.2 ± 3.4% and showed no substantial difference between the three sediment treatments (Table 3). The average percent hatch for eggs buried 48 h post fertilization was 81.9 ± 0.1% for 0.3, 0.6, and 1.2 mm treatments for all three sediments; while the percent hatch at 3 mm averaged 0.8%. This effect is attributed to low DO concentrations at the greater sediment depth. For all three test sediments evaluated, several sublethal effects were observed with increasing sediment depth (Figure 4). The percentage of hatched larvae that were morphologically normal and larval survival declined with sediment depth. Abnormalities included scoliotic larvae and larvae that did not straighten from their pre-hatch coiled posture. These fish often had a c-shaped notochord, or coiled tail. The most common abnormality was a reduction in the finfold. Although not quantified, many of the dead eggs had embryos that had developed to the larval stage but did not hatch. For dead larvae there was no sign that larvae did not completely exit the chorion, therefore it is presumed they died shortly after hatch. The duration of incubation (time from fertilization to hatch) increased and was most notable for treatments 1.2 and 3 mm. However, once larvae began to hatch there was no difference in length of hatching (time from first hatch to last larval fish hatching; Table 3).

Table 3. A summary of hatching success for the three test sediments.

Sediment	Depth (mm)	% Hatch	% Normal	% Dead Eggs	% Dead Larvae	Time to Hatch	Time Elapsed
Mianus River	0	82.3 ± 19.2	66.3 ± 17.5	14.7 ± 18.6	1.7 ± 1.5	15.3 ± 0.5	3.5 ± 1.0
	0.25	83.7 ± 9.2	68.4 ± 15.6	4.7 ± 2.7	8.1 ± 12.2	15.3 ± 0.5	3.5 ± 1.0
	0.46	62.7 ± 35.5	46.7 ± 26	27.7 ± 35.2	7.3 ± 6.3	15.6 ± 0.9	3.6 ± 1.3
	1.22	63.3 ± 42.9	47.8 ± 33.9	4.7 ± 6.4	13.9 ± 22.4	15.4 ± 1.5	4.0 ± 1.2
	2.76	2.0 ± 4.0	2.3 ± 5.6	33.7 ± 27.5	8.7 ± 12.9	18.0 ± 2.8	4.0
Milford Harbor	0	89.0 ± 11.1	74.2 ± 16.6	5.3 ± 4.5	3.4 ± 7.5	14.8 ± 0.4	2.7 ± 0.8
	0.27	88.3 ± 12.7	77 ± 18.8	4.3 ± 4.8	5.1 ± 7.8	15.2 ± 0.4	2.5 ± 1.6
	0.51	91.0 ± 5.0	64.7 ± 10.6	3.7 ± 2.9	3.4 ± 2.1	15.3 ± 0.5	3.0 ± 0.9
	1.05	84.7 ± 11.2	66.8 ± 12.4	5.7 ± 5.9	4.6 ± 3.6	15.5 ± 0.5	3.3 ± 1.0
	2.81	0	0	97.3 ± 4.7	0	0	0
Patchogue River	0	87.2 ±21.8	75.1 ± 16.5	6.9 ± 10.2	3.6 ± 8.0	14.0 ± 1.3	4.3 ± 0.8
	0.35	86.4 ±13.0	72.5 ± 6.1	3.0 ± 2.8	4.7 ± 5.0	15.0 ± 0.0	3.5 ± 1.0
	0.55	89.0 ±15.5	71.1 ± 11.6	5.3 ± 10.3	3.8 ± 3.1	15.2 ± 0.4	3.3 ± 1.0
	1.31	73.0 ±34.6	70.6 ± 24	1.3 ± 1.6	15.1 ± 22.7	17.0 ± 2.4	3.5 ± 1.5
	3.01	0.3 ±0.8	0	92.0 ± 12.3	0	22	0

% hatch = larvae/eggs; % normal development = normal/total larvae; % dead eggs = dead eggs/eggs stocked; % dead larvae = dead larvae/total larvae; time to hatch = date of first hatch – spawn date; time elapsed = date of last hatch – date of first hatch.

When the results are combined, the sediment depth that would likely have an effect on 5, 10 and 50% (effective concentration, EC5, EC10, and EC50) of the population of test organisms is 1.06, 1.17 and 1.59 mm, respectively (Table 4). The No Observable Effects Concentration (NOEC) and Lowest Observable Effects Concentrations (LOEC) was 1.31 and 2.76, respectively, for combined data. NOEC is the highest concentration that was used in the test that that did not cause an effect that is statistically different from the control ($p < 0.05$). The LOEC is the lowest tested concentration that is significantly different from the control.

Table 4. Statistical summaries estimated for the winter flounder hatching success endpoint.

Sediment	NOEC	LOEC	EC5	EC10	EC50
Mianus	1.22	2.76	0.76 (0.00-1.18)	0.98 (0.01-1.42)	1.56 (0.64-3.60)
Milford	1.05	2.81	1.05 (0.95-1.14)	1.19 (1.09-1.28)	1.72 (1.63-1.80)
Patchogue	1.31	3.01	1.12 (1.08 -1.16)	1.22 (1.19-1.26)	1.58 (1.53-1.64)
Combined	**1.31**	**2.76**	**1.06 (0.91-1.17)**	**1.17 (1.03-1.28)**	**1.59 (1.33-1.91)**

NOEC = No Observable Effects Concentration; LOEC = Lowest Observable Effects Concentration; EC5 = Effective Concentration affecting 5% of the test organism population; EC10 = Effective Concentration affecting 10% of the test organism population; EC50 = Effective Concentration affecting 50% of the test organism population.

CONCLUSIONS

The present study determined the effects of sedimentation on WFL eggs buried 48 h post fertilization under laboratory conditions. Sublethal and lethal effects were observed as sediment depth increased. When data from all three sediments was combined, a NOEC of 1.31 and LOEC of 2.76 was estimated. The expectations to assess potentially adverse effects following burial of WFL eggs are influenced by experimental conditions. It is not known whether the sediment depths studied are mimicking exposure conditions created by a dredge operation or natural exposure conditions or both. For instance, if the experimental sediment depths are several orders of magnitude higher than what a dredge operation would contribute it would be more difficult to assess adverse effects on or risks to WFL eggs. Although there is an overall lack of knowledge about the sedimentation rates in WFL spawning areas, the data from the present experiments are still needed to establish experimental rigor and to assess the effects of sedimentation.

These experiments provided insight into the importance of sedimentation on WFL eggs. To use this evidence-based information to enable better-informed EWs, additional questions that need to be addressed include: (1) What is the ambient sedimentation rate during WFL spawning? (2) What is the incremental contribution to the sedimentation rate from dredging operations? (3) What is the DO profile of the sediment in the estuary under ambient conditions and after dredging? (4) How much does sedimentation and DO levels vary in response to natural hydrodynamics? Of importance is the need to obtain reliable data of ambient conditions so that the effects caused by sedimentation due to dredging can be determined in light of the effects caused by naturally occurring sedimentation. This knowledge will provide a better exposure assessment of dredge relevant concentrations of sedimentation, which will help evaluate EWs as an effective practice for managing WFL in New England waterways.

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