Final Report: Investigation of Noise Pollution in the Hudson River at Pier 26 and Pier 97

Abstract:

Anthropogenic noise has become a growing concern for underwater wildlife. Studies show that noise pollution from human activities impacts the entire marine food web, including aquatic plants and fishes, posing a substantial threat to marine ecosystems and biodiversity. This project examines sound levels recorded in the Hudson River at Pier 26 and Pier 97. Pier 26 is a high - traffic area, with nearby ferry routes, commercial shipping, and frequent recreational boating activity, while Pier 97 experiences comparatively lower commercial and recreational activity. We recorded underwater noise using a vertical hydrophone array, and at the same time we captured videos of nearby ship traffic on the river with a camcorder. We then analyzed the collected data using customized software tools to evaluate how loud the environment was over time and how often loud sound events occurred. Measured values were then compared with acoustic thresholds provided by the National Oceanic and Atmospheric (NOAA, 2018), for three classified injuries among typical fish species (such as the oyster toadfish and black sea bass) in the region: behavioral disturbance, temporary threshold shift (TTS), and onset of auditory injury. Findings indicate excessive exceedance of recommended thresholds at Pier 26.

Introduction:

Anthropogenic underwater noise is recognized as an environmental hazard in aquatic ecosystems, influenced heavily by ship traffic, construction, and industrial activity (Hildebrand, 2009). Sound propagates about five times faster underwater and dissipates less over distance, allowing for anthropogenic noise to travel farther and remain louder. This results in aquatic life, especially coastal, being exposed to higher ambient sound levels (Slabbekoorn et al., 2010). There are multiple consequences of a higher ambient sound level, such as acoustic masking of biologically significant signals, stress induced physiological response s, as well as changes in behavior such as predator avoidance and reproductive communication (Slabbekoorn et al., 2010). Species of fish that utilize sound for communication or awareness are at particularly high risk, due to low-frequency ship noise overlapping with their hearing ranges (Popper & Hawkins, 2019). As coastal environments are subject to higher levels of human activity, it's important we understand the effects of such activity on the soundscape and environment (Halpern et al., 2008).

The Hudson River is a tidally influenced urban estuary that hosts a diverse population of biological communities while simultaneously functioning as a commercial and recreational waterway. The New York City region of the Hudson River, managed by the Hudson River Park Trust, experiences frequent boating traffic both commercial and private. Commercial and private vessels are a source of low-frequency underwater noise and increases in the quantity and size of vessels are shown to substantially change aquatic soundscapes. Global analysis from 1950 to the early 2000s reveals a three-time increase in the number of vessels, and a 3dB increase in low frequency ambient noise every decade (Hildebrand, 2009). The boating traffic causes fluctuations in the soundscape and provides a unique acoustic environment for analysis. Species such as the oyster toadfish (Opsanus tau) and black sea bass (Centropristis striata) inhabit this section of the Hudson River and are uniquely affected by the acoustics (Hudson River Park Trust, 2023; Fine & Parmentier, 2015).

According to the technical guideline established by NOAA, noise exposure of over 40 dB above the hearing threshold could lead to permanent auditory injury and behavioral abnormality in marine mammals (NOAA, 2024). Therefore, it is necessary to survey the underwater noise distribution in the Hudson River Park Estuary Sanctury (HRPES) and gauge its environmental impact. Previous research on underwater sound monitoring measures sound level with a single hydrophone (Martin & Popper, 2016, NASEM, 2017). The sound signals at different depths are not received simultaneously, and their connection to typical human activities is not specified. In particular, the environmental impact of underwater noise on the HRPES ecosystem has not been quantitatively analyzed. This study hopes to providethe framework for further investigation and monitoring of the HRPES soundscape.

The objective of this study is to measure underwater acoustic conditions at Pier 97 and Pier 26 in the Hudson River, and to assess sound level exceedances relative to the NOAA thresholds. To determine whether the potential for biological risk is present, this study utilizes an exceedance-based framework commonly used in underwater acoustics to classify sound exposure and categorize its possible effects (Southall et al., 2007; Popper et al., 2014). Furthermore, this study includes the analysis of current, as well as hydrophone depth in its analysis of the soundscape. To do so, correlation coefficients were calculated for each metric to determine the most impactful factors on the sound levels. Findings indicate excessive exceedance of recommended thresholds at Pier 26.

In linking the measured sound levels to established thresholds, this study aims to support the Trust in efforts to contribute to a better understanding of the urban estuarine soundscape and its ecosystem.

Methods:

To assess the impact of noise pollution on aquatic wildlife in HRPES, we conducted a literature review to find out the thresholds of sound levels considered to be harmful to fishes. From existing research work we identified thresholds of sound levels causing several major types of auditory injuries in different fish species. Table 1 provides a summary of the established thresholds for three types of injuries: behavioral disturbance, TTS, and onset of auditory injury (NOAA24, Popper et al., 2014). Prior research shows that RMS sound levels exceeding 150dB re 1 μPa interfere with fish communication and behavior (NOAA, 2018).

Meanwhile, we built an underwater sound recording system with a hydrophone array to sample acoustic signals at different depths in the river. After we collected the acoustic data at Pier 26 and Pier 97, we generated a noise map to reveal sound levels at different depths in the river, followed by changing to 5-min equivalent sound pressure level (L_{eq}), computing correlation coefficients, and hypothesis testing to determine if any recorded sound level at a particular spot exceeded the anchored threshold. Overall, there are three major components in this work: (1) software and hardware development for building an underwater acoustic sound recording system, (2) on-site experiment for noise and ship traffic data collection at Pier 26 and Pier 97, (3) data processing and analysis on noise level exceedance. The following passages detail the implementation procedure.

Sound recording system. Underwater sound was recorded using a Tascam Portacapture X8 digital recorder operating on 48 V phantom power with three CRT-90N hydrophones. The CRT-90N manufactured by Cetacean Research Technology are sensors designed for passive underwater acoustic monitoring with a sensitivity of –199 dB re 1 V/µPa and a depth rating suitable for our shallow water coastal deployment. Phantom power was supplied directly by the Portacapture X8, connected directly to an external battery. Recordings were made using 24-bit resolution and a sampling rate of 96kHz, each hydrophone recorded audio on its own unique input channel. The preamplifier input gain of the recorder was fixed at +4 dB, and the hydrophones +30 dB (total +34 dB) for the duration of each deployment. According to the manufacturer of the recorder, 0 dBFS corresponds to an RMS line level of +24 dBu at recorder input. This specification was used to convert normalized digital samples into physical voltages. Hydrophones were then tied in an array formation with the intention of the first hydrophone being deployed 2 meters below the water surface, and subsequent hydrophones were deployed 1 meter beneath the prior hydrophone (Figure A4 in Appendices).

On-site experiment. We selected the location for placing the hydrophone array based on the bathymetry data provided from the park. Figure 1B and 1C show the locations of the two sites in HRPES. Acoustic monitoring was conducted at the Hudson River at Pier 26 and Pier 97. Monitoring at Pier 97 took place on Sunday, July 13th, from 10:30 to 21:00 local time. Monitoring at Pier 26 took place on Friday, July 18th, from 11:00 to 21:00 local time.

Figure 1A. Equipment deployment

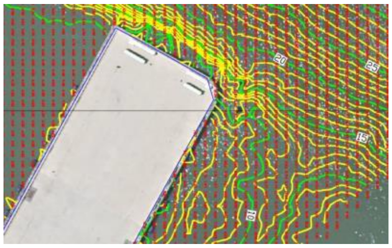


Figure 1B. Pier 97 Bathymetry

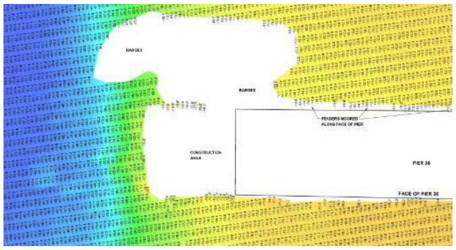


Figure 1C. Pier 26 Bathymetry

Recordings were captured on a continuous basis, with audio files segmented into 30 - minute increments for data management. We used a camcorder to record passing vessel traffic simultaneously for the duration of experiment at Pier 26, and intermittently at Pier 97 for later analysis (See figures A8.1-A8.3 & A9.1-A9.3 in Appendices). In addition, we noticed that the surface level of the river changes through the day due to tidal flowing and ebbing, and therefore the hydrophone position information needs to be updated according to the actual depths at different times. Another observation is that the sound generated by the current on the river also contributes to the recording. Based on this consideration, we collected data for three environmental factors that would impact the noise level during the experiment: hydrophone position underwater (HPU), hydrophone incline degree (HID, similar to reflect water current), and ship traffic (commercial and recreational).

During the experiment, HPU was measured by placing a marker on the hydrophone array and adjusting the depth as tidal influences changed. This resulted in a near consistent depth for each hydrophone with slight deviations throughout the day (see figures 3B & 3C); measurements were taken on an hourly basis. Current was measured through observation of the array's vertical angle in the water. When the array strongly swayed north, the corresponding angle was recorded positively and when it swayed south, there was a corresponding negative angle recorded (see figure 3A). To record the change of this angle information that reflects the current speed, multiple pictures were taken of the hydrophone angle every hour. We also measured the surface level of the river alongside every hour for later analysis.

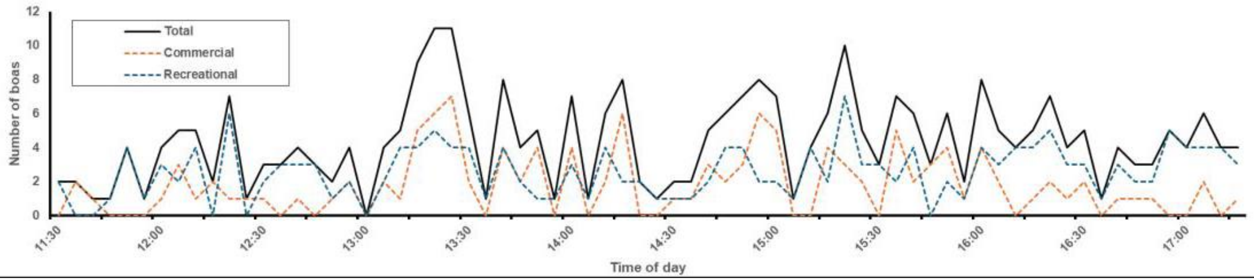


Figure 2A. Pier 26 Ship counting, 5-minute bins

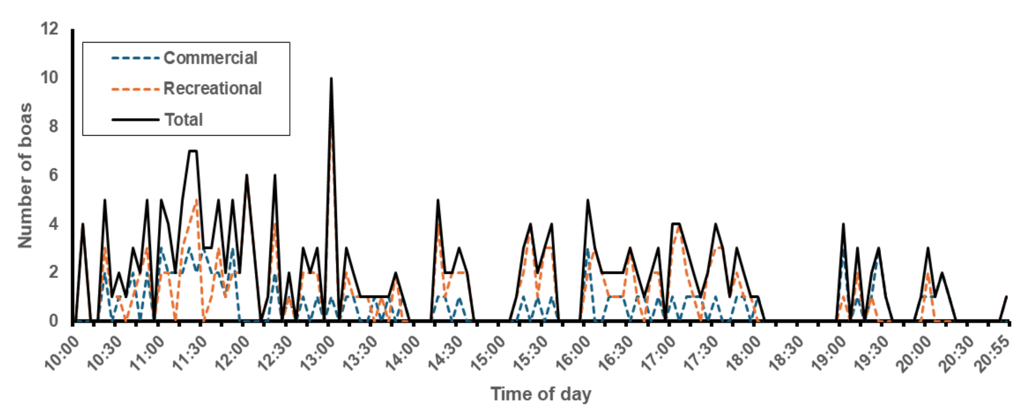


Figure 2B. Pier 97 ship counting

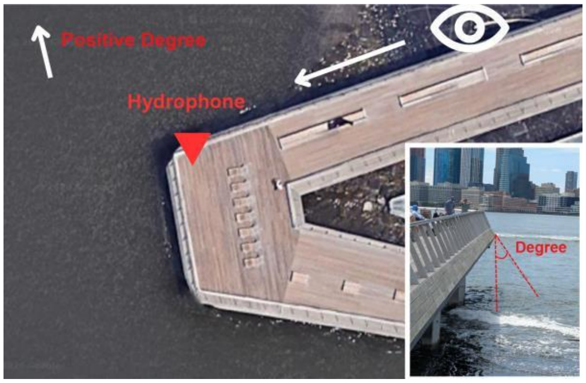


Figure 3. current measurement method

Table 1. the Hydrophone position underwater and degree in Pier 26

Time	1st HPU, top (m)	2nd HPU, middle (m)	3rd HPU, bottom, (m)	Degree
11:30	1.54	2.53	3.51	0
12:00	1.6933	2.61	3.5167	-10
1	2	2.77	3.53	40
2	2.03	2.79	3.56	40
3	2.05	2.92	3.78	30
4	2	2.5	3	30
5:15	1.97	2.97	3.97	0

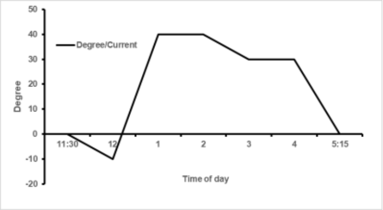


Figure 3A. Current displayed by hydrophone degree hourly

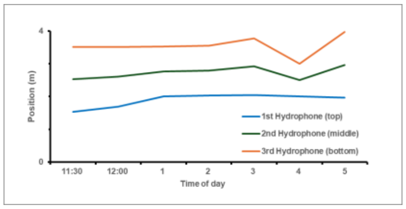


Figure 3B. Depth in meters of each hydrophone

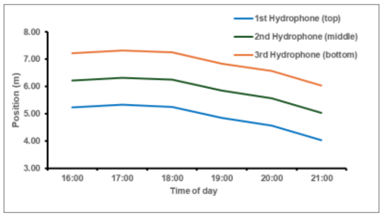


Figure 3C. Depth in meters of each hydrophone

Hypothesis testing. To quantify underwater noise exposure in biologically meaningful terms, we computed standard acoustic metrics used in underwater acoustics. For this study, normalized signals were converted to Root-Mean-Square Sound Pressure Level (L_{eq}) following standard underwater acoustic metrics (Ren et al, 2012). SPL is the instantaneous sound pressure re 1 μPa, and RMS SPL is an energy averaged over a defined time window (NOAA, 2018). To determine what extent sound levels exceeded the established behavioral disturbance thresholds, we performed one-tailed tests for each channel (Illowsky & Dean, 2013) on binned $L_{eq,5min}$ values at each hydrophone depth ($L_{eq,5min}$: frame duration = 50 ms, hop = 50 ms, bin duration = 5 min; sample size: 67 for Pier 26; 115 for Pier 97).

We set up hypotheses as follows:

Null hypothesis (H_0): $\mu \leq 150$ dB re 1 μPa (Mean $L_{eq,5min}$ of sample size does not exceed threshold)

Alternative hypothesis (H_1): $\mu > 150$ dB re 1 μPa (Mean $L_{eq,5min}$ of sample size exceeds threshold)

In practice, this was implemented in Microsoft Excel by creating a threshold vector of the same length as the observation vector and applying a paired t-test. A one-tailed p-value was compared with the significance level $\alpha=0.05$ (Illowsky & Dean, 2013).

Correlation analysis. After time alignment, Pearson correlation coefficients were computed between the recorded sound level (for each hydrophone channel) and each covariate: total ship counting, interpolated HPU change, and interpolated HID. Correlations were computed separately for each hydrophone channel to evaluate depth-dependent sensitivity. Linear interpolation was utilized when analyzing depth and current to simulate changing tidal patterns between hourly adjustments. Vessel traffic video was reviewed, and vessels were classified based on utility and size. Ships were accumulated and time stamped for the moment they crossed the hydrophone arrays' path; the counts were then accumulated in 1-minute, 5-minute, 10-minute, and hour-long bins for correlation testing. This allowed for the generation of a correlation coefficient at different resolutions for each metric.

Power Spectral Density. Broadband levels quantify overall acoustic energy but do not display how the energy is distributed across frequency. PSD mapping was utilized to visualize frequency-specific contributions. PSD was calculated using calibrated pressure time series and short time Fourier transforms with overlapping windows to preserve temporal resolution while maintaining spectral stability. All PSD values were expressed in units of dB re 1 μPa/Hz, allowing for direct comparison across time, depth, and site. Spectral maps were generated for each hydrophone (using a sample size $n = 4096$, averaged over a 30-min time window), allowing for visualization of how frequency varied with depth and environmental conditions (Jensen et al, 2011).

Results:

Pier 26. As shown in figure 4, the acoustic field exhibited a persistent vertical gradient, with the shallow hydrophone (top) recording the highest, the mid-depth hydrophone intermediate levels, and the deepest hydrophone the lowest. Also, this ordering remained stable over time and during periods of elevated vessel traffic, indicating that the dominant noise sources and/or propagation conditions impact the upper water column more. Pier 26 is located in Tribeca along the lower Hudson River within the Hudson River Estuary, where freshwater runoff mixes with ocean saltwater, making it part of a dynamic near-harbor environment. Ship noise is a dominant low-frequency anthropogenic source, primarily generated by propeller cavitation and propulsion machinery. In shallow water, repeated interactions with the sea surface and seabed create a waveguide/multipath environment that can produce depth-dependent received levels. Consistent with these mechanisms, our measurements at Pier 26 showed higher at shallower hydrophones (top > middle > bottom) (Hildebrand, 2004).

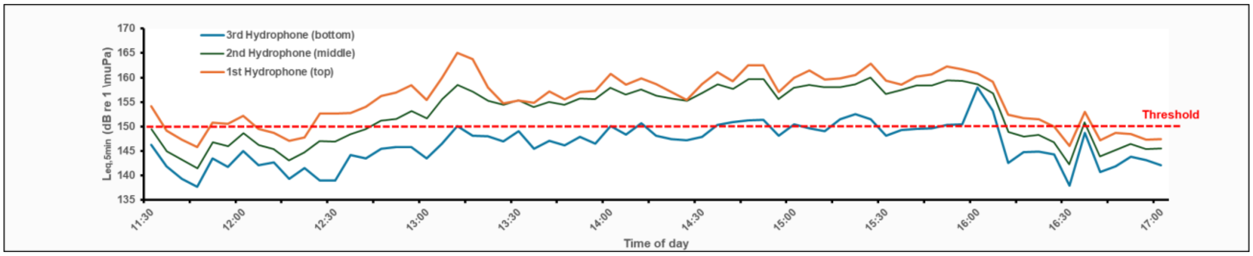


Figure 4. L_{eq}, 5min from ***1*** *:30 to 17:00 for three channels at Pier 26*

The Hypothesis test results in Figure 5 further suggest that behavioral-risk exceedance is depth-dependent. The top and middle hydrophones frequently and significantly exceeded the 150 dB re 1 μPa behavioral threshold, while the bottom hydrophone was often below threshold and showed weaker or intermittent exceedance. Ecologically, this implies that fish occupying shallower depths near Pier 26 may be exposed to more frequent and sustained stress conditions, while deeper habitats may provide partial acoustic refuge, though not complete protection during high-traffic episodes.

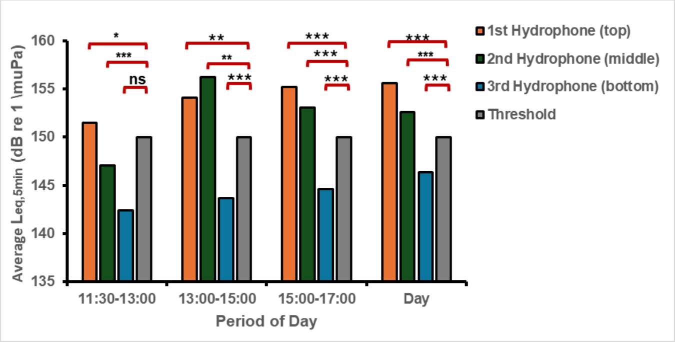


Figure 5. Hypothesis Test: Mean $L_{eq,5min}$ *vs 150 dB Threshold by Depth and Time Block, Pier 26*

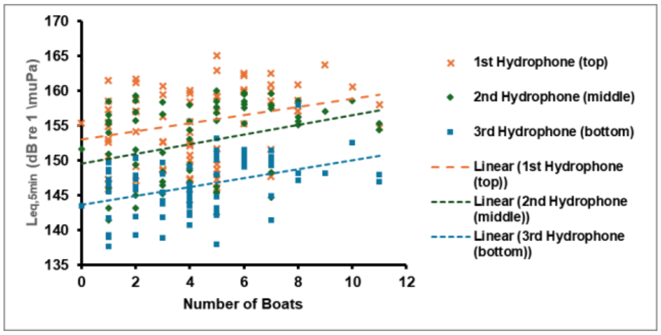


Figure 6A. Regression: $L_{eq,5min}$ *vs Total boat count, Pier 26 (r = 0.579, 0.696, 0.647）*

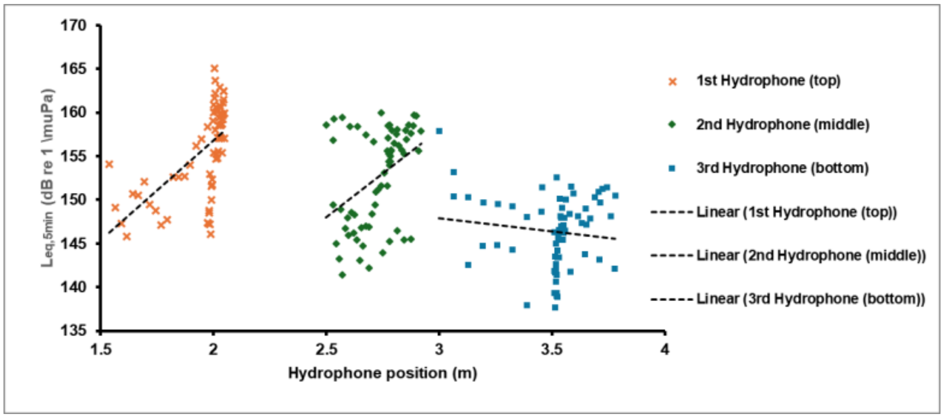


Figure 6B. Regression: $L_{eq,5min}$ *vs Hydrophone Position Underwater, Pier 26 (r = 22.832, 20.034, -2.9926)*

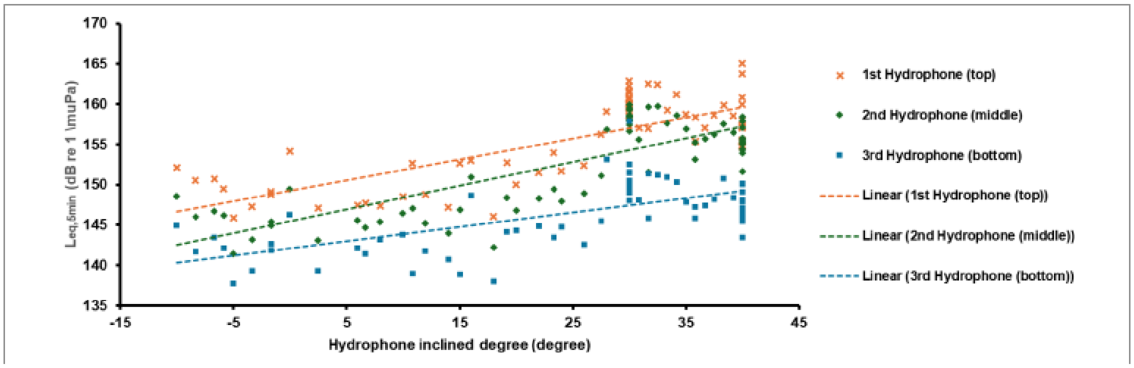


Figure 6C. Regression: $L_{eq,5min}$ *vs Hydrophone Incline Degree, Pier 26 (r = 0.259, 0.294, 0.178)*

As shown in Figure 6A, 6B, and 6C, single-factor linear regressions (Illowsky & Dean, 2013) showed that $L_{eq,5min}$ increased with (i) the number of boats, (ii) hydrophone array tilt angle, and (iii) shallower hydrophone depth. The positive regressions with total boat count and tilt were observed for all three hydrophones, with the larger sensitivity consistently at the top and middle hydrophone and the smaller at the bottom hydrophone. Different linear regression for hydrophones at different depths also show a vertical gradient in received levels, which is a negative regression in the bottom hydrophone, and others are position regression. However, substantial scatter at fixed boat count or tilt angle suggests that there are more unmodeled factors (e.g., vessel type/distance/speed and tidal state) that also contribute, and the apparent depth trends within each hydrophone likely reflect co-variation with tide/current conditions rather than a single factor alone.

Limitation: Because depth and tilt were sampled discretely and interpolated, short-term vertical fluctuations may not be fully captured. In addition, shallow hydrophones may be more susceptible to near-surface flow/wave noise, which could elevate broadband levels in some intervals. Future work could incorporate frequency -weighted or band-limited metrics aligned with fish hearing ranges and examine whether the depth gradient is driven primarily by low-frequency vessel noise or by broadband surface-related components.

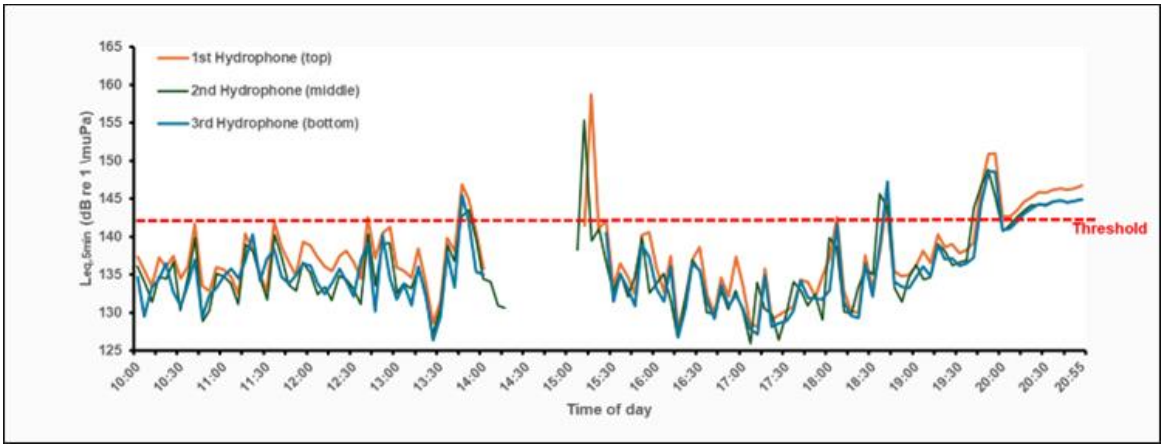


Figure 7A. $L_{eq,5min}$ *from 10:00 – 20:55 for three channels at Pier 26*

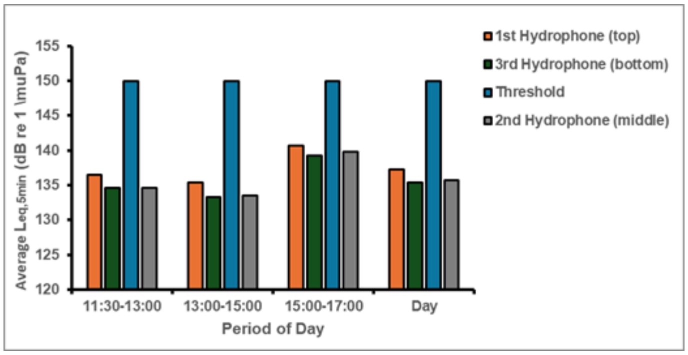


Figure 7B. Hypothesis Test Results: Mean $L_{eq,5min}$ *vs 150 dB Threshold, Depth and Time Block, Pier 97*

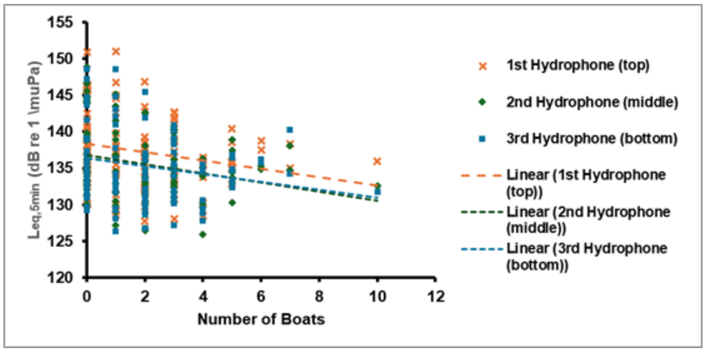


Figure 7C. Regression: $L_{eq,5min}$ *vs Total boat count, Pier 97 (r = -0.565, -0.620, -0.542)*

Pier 97. As information from Figure 7A, 7B, and 7C, Pier 97 exhibited substantially lower broadband exposure than Pier 26, with mean $L_{eq,5min}$ values remaining well below the 150 dB behavioral threshold across most of the deployment and only weak depth stratification. And the hypothesis test confirms this point again that there are significantly lower than the 150db threshold in all three channels. Unlike Pier 26, $L_{eq,5min}$ – ship count regression in Pier 97 did not show a consistent positive relationship with sound levels, suggesting that either traffic intensity was low, observations were limited, or background environmental conditions dominated the soundscape. Notably, a late-evening increase in $L_{eq,5min}$ occurred simultaneously across all positions, but according to our observations that a high-intensity on-shore anthropogenic event (amplified music and dancing) during the same time window, providing a plausible explanation for the broadband elevation in underwater levels via air–water coupling and nearshore propagation.

Pier 26 and Pier 97 showed clear contrasts in both boat count and acoustic exposure, but the comparisons need to be interpreted with time and locations: Pier 26 was monitored on a Friday (weekday), whereas Pier 97 was monitored on a Sunday (weekend). Weekday conditions near Lower Manhattan typically include stronger commuter and service traffic, while weekend patterns may shift toward lower or more intermittent traffic. Accordingly, Pier 26 exhibited frequent and sustained elevations in $L_{eq,5min}$ and sound level variability tracked vessel activity more closely, and this tells boat count as a primary factor of elevated noise at this site. In contrast, Pier 97 generally remained below the behavioral threshold for most of the day and did not show the same clear positive relationship between boat count and $L_{eq,5min}$; instead, the late-evening rise at Pier 97 coincided with a loud onshore human activity event. Probably, sources not from boats can affect more at this location, Pier 97. These differences are also affected by setting location: Pier 26's physical location in the lower Hudson near busy Lower Manhattan corridors and ferry routes likely exposes it to denser and more continuous vessel passage compared with Pier 97 that has a lower routine traffic.

Noise map at different depths. PSD analysis revealed frequency-dependent characteristics of the underwater soundscape that were not captured using broadband metrics. PSD maps generated for Pier 26 **(**Fig A5 in Appendices**)** Show elevated acoustic energy at low frequencies (<500Hz). PSD plots generated without interpolation (Fig A6 in Appendices) confirm low frequency dominance across all hydrophone recordings. The consistent frequency readings across depths display that low frequency energy was not a byproduct of interpolation. The results of PSD mapping also indicate that the low frequency energy overlaps with effective hearing range and communication range of many fish species (<1kHz) (Popper & Hawkins, 2019).

Hypothesis testing on sound level exceedance. For each hydrophone channel, RMS SPL was evaluated to see whether the mean SPL exceeded the relevant threshold. For the first hydrophone (≈2m depth), the mean RMS SPL = 156.08 dB and SD = 4.81dB (n = 71,997). Null hypothesis was rejected at providing strong evidence that mean RMS SPL exceeded disturbance criteria at 2m depth. For the second hydrophone (≈ 3m depth) the exceedance was stronger. The mean was 158.47 dB with SD = 4.83 dB (n = 71,997). This indicated the ability to reject the null hypothesis, showing threshold exceedance of mean SPL within the same time window. The third hydrophone (4m depth) shows SPL distribution below the threshold. The mean SPL was 148.25 dB with SD = 3.55 dB (n = 71,997). Null hypothesis was not rejected indicating insufficient evidence of mean RMS SPL exceeding 150 dB at 4m depth during the time interval. Hypothesis testing shows depth dependent results during the analyzed windows, where the deepest hydrophone mean RMS SPL did not exceed disturbance criteria.

Discussion:

Acoustic analysis revealed repeated exceedances of behavioral disturbance thresholds based on RMS SPL, while peak SPL values remained below thresholds associated with auditory injury or temporal threshold shift. The distinction in important behavioral thresholds is meant to capture sublethal, but ecologically significant responses, while peak based criteria are primarily used to assess acute auditory risk (Popper & Hastings 2009).

Behavioral disturbance threshold exceedance indicates impact on behaviors such as habitat displacement, altered swimming depth or orientation, interruption of feeding, masking of biologically important sounds (prey detection, communication), elevated stress responses (Popper & Hastings, 2009; Slabbekoorn et al., 2010; Kunc et al., 2016). These effects occur at sound levels well below those required to implicate hearing damage, particularly when exposure is repeated.

Despite limited sample size, one day at each pier, behavioral disturbances were observed at both sites. Importantly, recordings at Pier 97 were conducted on a Sunday, when vessel traffic was significantly reduced from weekday conditions. With behavioral thresholds exceeding consistent at Pier 97, the data suggests that reduced vessel traffic still generates moderate anthropogenic noise sufficient to raise ambient sound levels into ranges associated with behavioral disturbance. The findings are consistent with studies showing low-frequency vessel noise propagating efficiently in shallow waterways, which may elevate background sound levels over extended time (Slabbekoorn et al. 2010).

Some limitations should be considered when interpreting these results. First, the analysis relied on broadband SPL metrics without frequency weighting. Without frequency weighting or filtering, our metrics may incorporate energy outside biologically relevant hearing ranges, potentially inflating or obscuring biologically meaningful exposure data (Popper et al. 2014). Second, surface-related noise sources, such as water splashing or flowing, may have contributed to the shallowest hydrophone picking up the loudest sound. This is relevant at Pier 26 where high current and wind speeds are frequent. Additionally, because depth and tilt were sampled discretely and interpolated, short-term vertical fluctuations may not be fully captured. In addition, shallow hydrophones may be more susceptible to near-surface flow/wave noise, which could elevate broadband levels in some intervals. Lastly, this study has not yet analyzed cumulative sound exposure to determine exposure patterns and other related risks, which remains an important and relevant metric for underwater acoustics (Kunc et al., 2016). Future work could incorporate frequency-weighting or band-limited metrics aligned with fish hearing ranges and examine whether the depth gradient is driven primarily by low-frequency vessel noise or by broadband surface-related components.

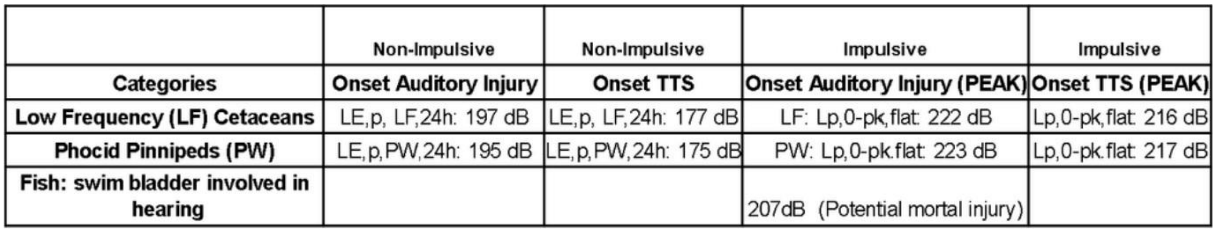
Conclusions

The audio analysis resulted in exceedance of behavioral thresholds for RMS SPL; however peak SPL did not exceed any listed thresholds, meaning there was limited risk for auditory injury on the days of recording. These findings indicate anthropogenic sound levels in the Hudson River frequently reach levels capable of interfering with fish behavior, especially when vessel traffic is active. The combined use of broadband metrics with PSD analysis shows that elevated sound levels are persistent and concentrated at low frequencies which overlaps with known fish communication ranges. While this study was limited to short term recordings at two sites, the results provided clear indication that the Hudson River soundscape is heavily influenced by anthropogenic noise. The findings demonstrate the importance of continued, long-term monitoring, and the incorporation of frequency analysis when assessing underwater noise.

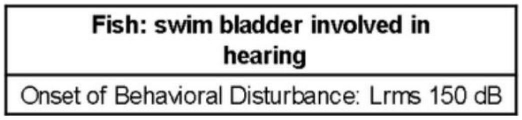
Appendices:

Tables

Table A1. *Summary of sound level thresholds causing different auditory injuries.*

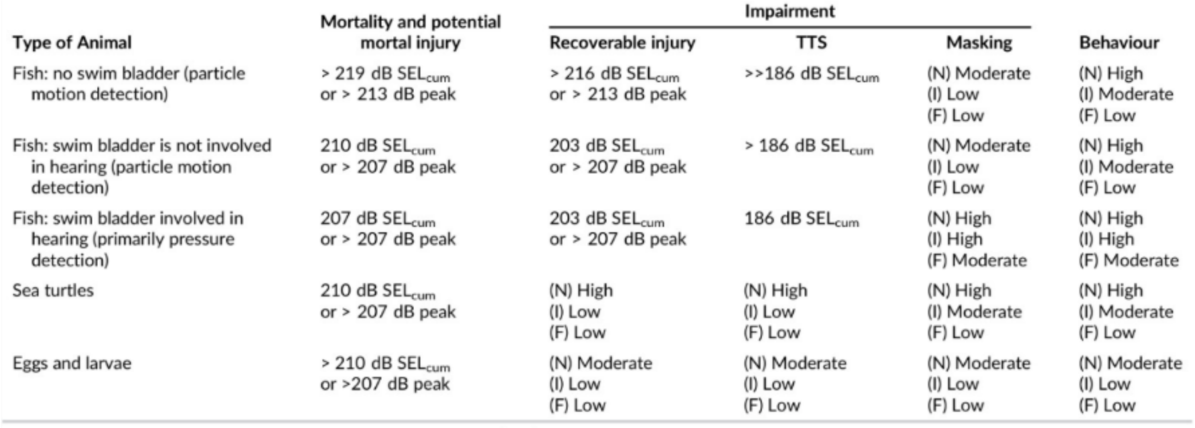


	Non-Impulsive	Non-Impulsive	Impulsive	Impulsive
Categories	**Onset Auditory Injury**	**Onset TTS**	**Onset Auditory Injury (PEAK)**	**Onset TTS (PEAK)**
Low Frequency (LF) Cetaceans	LE,p, LF,24h: 197 dB	LE,p, LF,24h: 177 dB	LF: Lp,0-pk,flat: 222 dB	Lp,0-pk,flat: 216 dB
Phocid Pinnipeds (PW)	LE,p,PW,24h: 195 dB	LE,p,PW,24h: 175 dB	PW: Lp,0-pk.flat: 223 dB	Lp,0-pk.flat: 217 dB
Fish: swim bladder involved in hearing			207dB (Potential mortal injury)	



Fish: swim bladder involved in hearing
Onset of Behavioral Disturbance: Lrms 150 dB

Table A2. *Additional thresholds for specific fish biology*



Type of Animal	Mortality and potential mortal injury	Impairment			
		Recoverable injury	TTS	Masking	Behaviour
Fish: no swim bladder (particle motion detection)	> 219 dB SEL_{cum} or > 213 dB peak	> 216 dB SEL_{cum} or > 213 dB peak	>>186 dB SEL_{cum}	(N) Moderate (I) Low (F) Low	(N) High (I) Moderate (F) Low
Fish: swim bladder is not involved in hearing (particle motion detection)	210 dB SEL_{cum} or > 207 dB peak	203 dB SEL_{cum} or > 207 dB peak	> 186 dB SEL_{cum}	(N) Moderate (I) Low (F) Low	(N) High (I) Moderate (F) Low
Fish: swim bladder involved in hearing (primarily pressure detection)	207 dB SEL_{cum} or > 207 dB peak	203 dB SEL_{cum} or > 207 dB peak	186 dB SEL_{cum}	(N) High (I) High (F) Moderate	(N) High (I) High (F) Moderate
Sea turtles	210 dB SEL_{cum} or > 207 dB peak	(N) High (I) Low (F) Low	(N) High (I) Low (F) Low	(N) High (I) Moderate (F) Low	(N) High (I) Moderate (F) Low
Eggs and larvae	> 210 dB SEL_{cum} or >207 dB peak	(N) Moderate (I) Low (F) Low	(N) Moderate (I) Low (F) Low	(N) Moderate (I) Low (F) Low	(N) Moderate (I) Low (F) Low

Figures:

Figure A1. *Test sites at Piers 97, Pier 26*

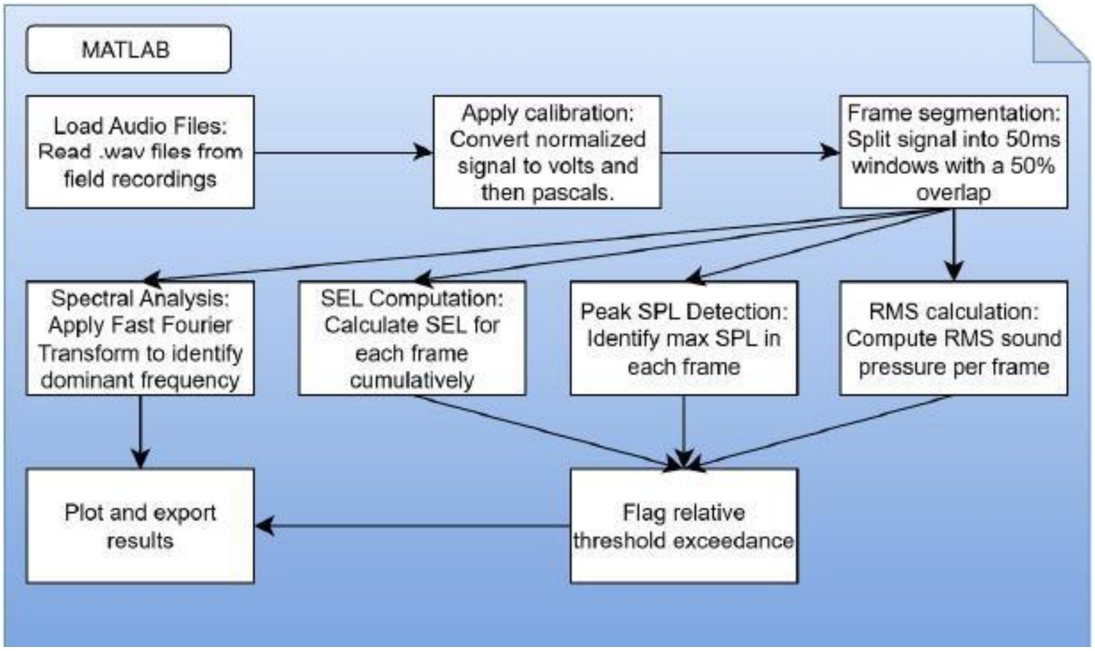


Figure A2. *MATLAB Data processing*

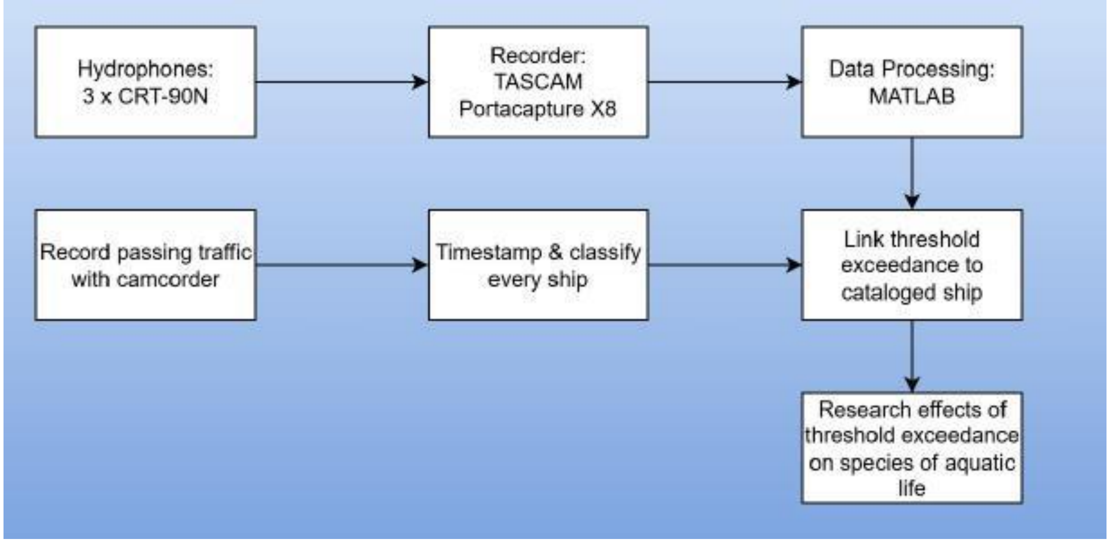


Figure A3. *Research/Experiment overview*

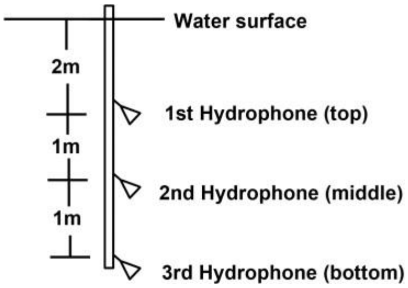


Figure A4. *Hydrophone array deployment*

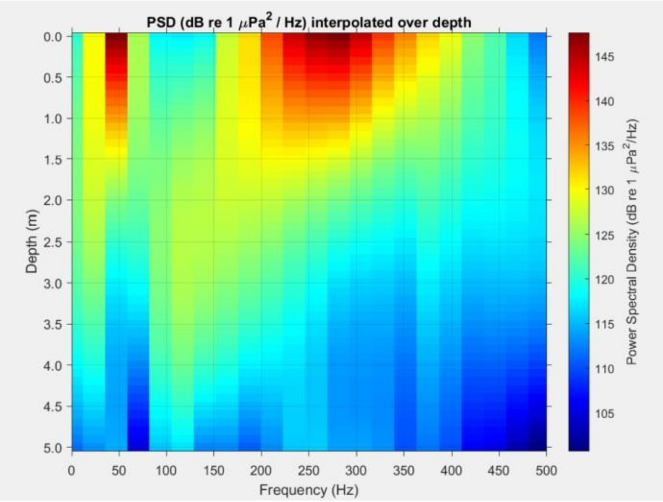


Figure A5. PSD with interpolation

Pier 26 4:20PM – 4:50PM, (n = 4096; averaged over 30 minutes; 84,000 windows)

The PSD graph demonstrates low frequency dominance of the time window analyzed.

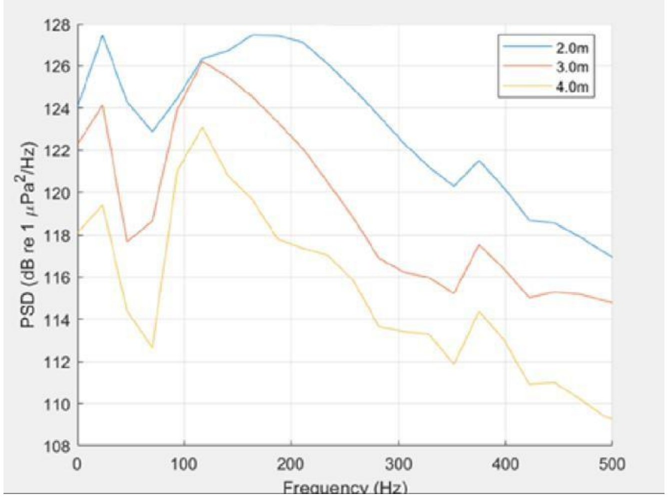


Figure A6. PSD no interpolation,

The figure demonstrates all three depths. (n = 4096; averaged over 30 minutes)

The figure represents PSD calculated independently for each hydrophone deployed.

Hypothesis testing

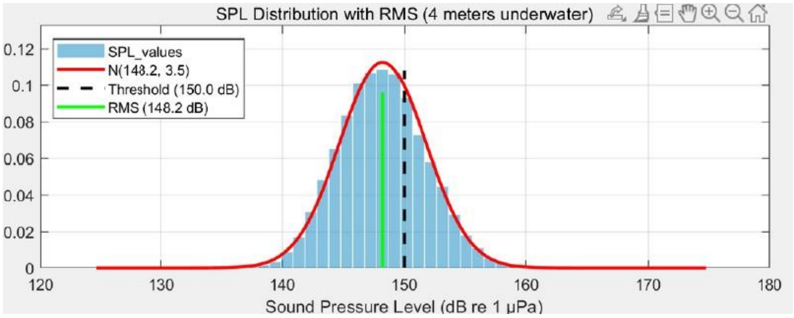


Figure A7.1. RMS SPL Distribution (4m)

This graph present RMS SPL distribution at 4 m depth. The blue bars represent the probability density of RMS SPL calculated using a fixed analysis window. The red curve shows normal distribution, and the dashed line marks the behavioral disturbance threshold. The solid green line indicates RMS SPL at this depth averaged for the entire window. A significant portion of the distribution lies beyond the threshold to indicate frequent exceedance.

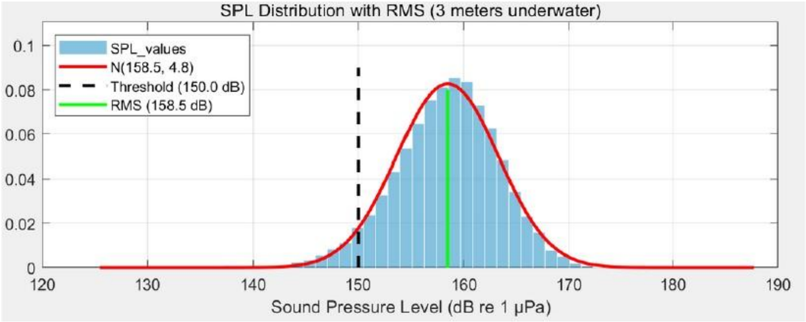


Figure A7.2. RMS SPL Distribution (3m)

This graph’s distribution is centered higher than the deeper measurement, with majority of the distribution above the behavioral disturbance threshold. The results indicate consistent exposure to anthropogenic noise in the mid water depth.

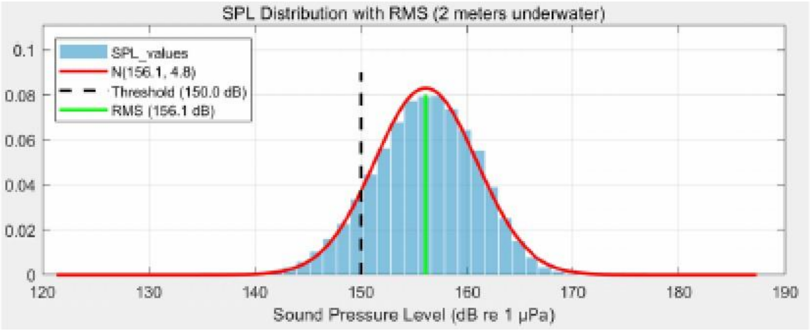


Figure A7.3. RMS SPL Distribution (2m)

This graph demonstrates the shallowest hydrophones distribution, with the majority of the distribution being above the behavioral disturbance threshold

RMS SPL

Figures A8.1-3 show the root mean square SPL for each analysis frame at all three depths, representing the time averaged acoustic energy of the soundscape rather than instantaneous pressure. RMS SPL was calculated from calibrated pressure samples within each frame and converted to dB re 1 μPa, allowing for a continuous measure of sustained sound exposure. The reported global RMS values summarize the acoustic energy over the full deployment for each depth. RMS SPL identifies periods of anthropogenic activity, such as vessel traffic, and provides a basis for evaluating threshold exceedance. The resulting RMS SPL graphs show consistent exceedance of the 150 dB re 1μPa associated with behavioral disturbance.

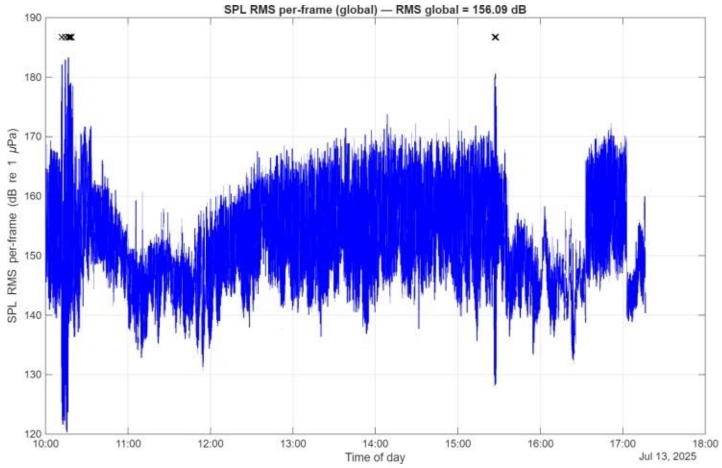


Figure A8.1. Pier 26 RMS Ch1

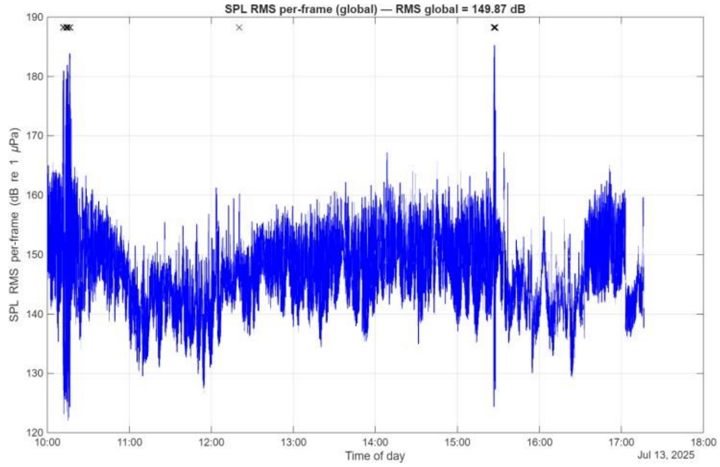


Figure A8.2. Pier 26 RMS Ch2

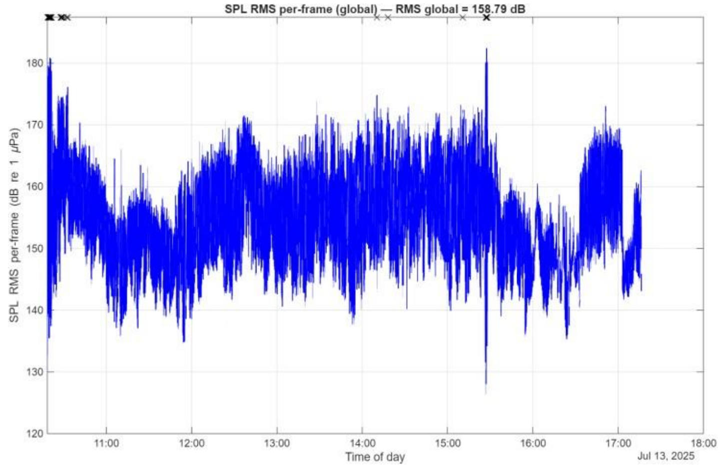


Figure A8.3. Pier 26 RMS Ch3

Peak SPL

Figures A9.1-3 display the frame based peak sound pressure level time series measured simultaneously using the array of hydrophones. For each frame (50 ms), the maximum instantaneous acoustic pressure was calculated from the waveform and converted to decibels. This metric shows short duration, high amplitude sounds that are not captured by more energy averaged measures, such as impulsive ship noise, or mechanical disturbance. The figures and audio observed from vessel traffic at Pier 26 and Pier 97 did not reach the listed thresholds for TTS injury, which indicated there was not a sound event that could cause physical damage to the aquatic habitats.

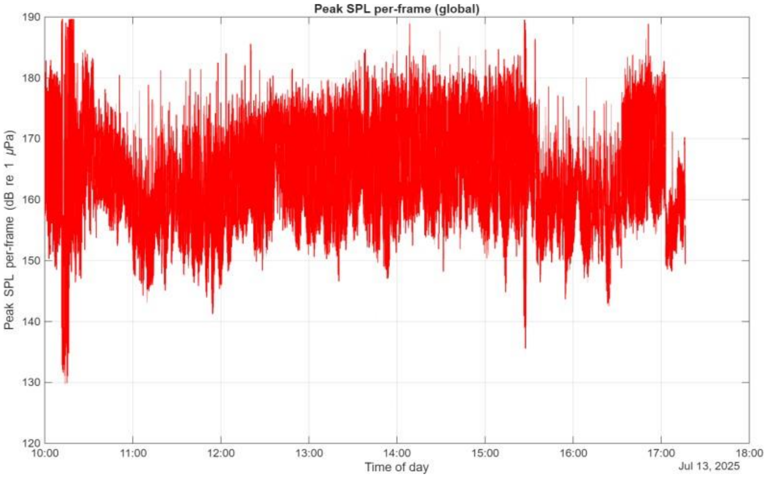


Figure A9.1. Pier 26 Peak Ch1

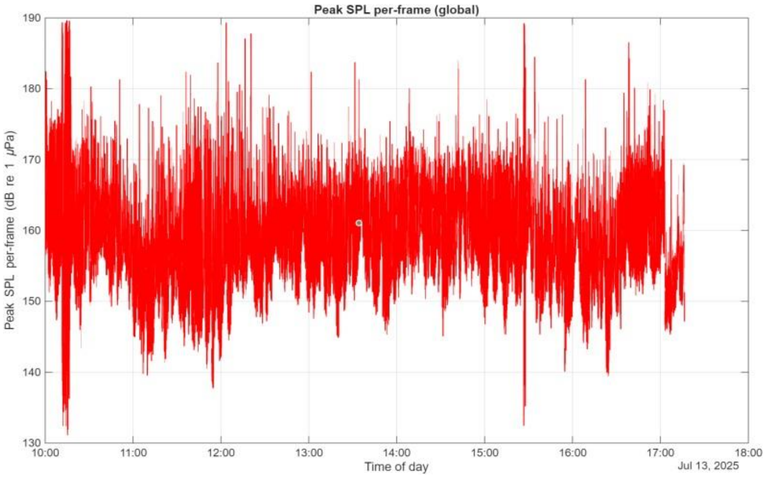


Figure A9.2. Pier 26 Peak Ch2

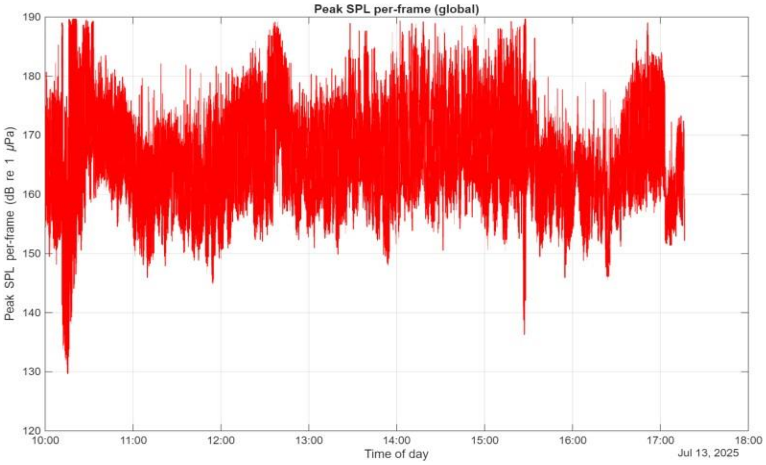


Figure A9.3. Pier 26 Peak Ch3

References:

Andrew, Rex & Howe, Bruce & Mercer, James & Dzieciuch, Matthew. (2002). Ocean ambient sound: Comparing the 1960s with the 1990s for a receiver off the California coast. Acoustics Research Letters Online-arlo - ACOUST RES LETT ONLINE-ARLO. 3.10.1121/1.1461915.

Barber, J. R., Crooks, K. R., & Fristrup, K. M. (2010). The costs of chronic noise exposure for terrestrial organisms. *Trends in Ecology & Evolution, 25*(3), 180–189. https://doi.org/10.1016/j.tree.2009.08.002

Benjamin S. Halpern et al., A Global Map of Human Impact on Marine Ecosystems.Science319,948-952(2008).DOI:10.1126/science.1149345

de Jong, K., Forland, T.N., Amorim, M.C.P. et al. Predicting the effects of anthropogenic noise on fish reproduction. Rev Fish Biol Fisheries 30, 245–268 (2020). https://doi.org/10.1007/s11160-020-09598-9

de Soto, N., Delorme, N., Atkins, J. et al. Anthropogenic noise causes body malformations and delays development in marine larvae. Sci Rep 3, 2831 (2013) https://doi.org/10.1038/srep02831

Fine, Michael & Parmentier, E.. (2015). Mechanisms of Fish Sound Production. 10.1007/978 -3-7091-1846-7_3.

Hawkins, Anthony & Pembroke, Ann & Popper, Arthur. (2014). Information gaps in understanding the effects of noise on fishes and invertebrates. Reviews in Fish Biology and Fisheries. 25. 10.1007/s11160-014-9369-3.

Hildebrand, J. (2004). *Sources of anthropogenic sound in the marine environment* [PDF]. Marine Mammal Commission. https://www.mmc.gov/wp-content/uploads/hildebrand.pdf

Hildebrand, JA. (2009). Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology-progress Series - MAR ECOL-PROGR SER. 395. 5-20. 10.3354/meps08353.

Hom, K. N., Quigley, T.D., Rodriguez, R. D., Dganski, S. G., Lazrinth, X. I., Jones, R., & Forlano, P.M. (2024). *Characterization of anthropogenic noise and oyster toadfish (Opsanus tau) calling behavior in urban and small-town coastal soundscape.* Journal of the Acoustical Society of America, *155*(2), 1230-1239. https://doi.org/10.1121/10.0024763

Hudson River Park Trust. (2023). *2022 Fish Survey Report (pp. 1-9).* Hudson River Park Trust. https://hudsonriverpark.org/app/uploads/2023/01/2022 -Fish-Survey-Report_final.pdf

Illowsky, B., & Dean, S. (2013). *Introductory Statistics*. OpenStax, Rice University, *pp. 505-430 & 679-691.* https://assets.openstax.org/oscms-prodcms/media/documents/IntroductoryStatistics-OP_i6tAI7e.pdf

Jensen, F. B., Kuperman, W. A., Porter, M. B., Schmidt, H. & Tolstoy, A. (2011), *Computational Ocean Acoustics*, Springer, 794, 65-153.

Kunc, H.P., McLaughlin, K. E., & Schmidt, R. (2016). *Aquatic noise pollution: Implication for individuals, populations, and ecosystems. Proceedings of the Royal Society B: Biological Sciences, 283*(1836), Article 20160839.https://doi.org/10.1098/rspb.2016.0839

Martin, S.B. & Popper, A.N. (2016). Short- and Long-term Monitoring of Underwater Sound Levels in the Hudson River. *J Acoust Soc Am.* 2016 Apr;139(4):1886.

Maruska, Karen & Mensinger, Allen. (2009). Acoustic characteristics and variations in grunt vocalizations in the oyster toadfish Opsanus tau. Environmental Biology of Fishes. 84. 325-337. 10.1007/s10641-009-9446-y.

National Academies of Sciences, Engineering, and Medicine (2017). Approaches to Understanding the Cumulative Effects of Stressors on Marine Mammals, *The National Academies Press*.

National Marine Fisheries Service (2018). *2018 revisions to: Technical guidance for assessing the effects of anthropogenic sound on marine mammal hearing (Version 2.0).* NOAA Technical Memorandum NMFS-OPR-59.

National Oceanic and Atmospheric Administration (2024). *2024 Updated Guidance for Assessing the Effects of Anthropogenic Sound on Marine Mammal Hearing-Underwater and In-Air Criteria for Onset of Auditory Injury and Temporary Threshold Shifts* [RTID 0648-XD212].

Popper, A.N., & Hawkins, A.D. (2018). The importance of particle motion to fishes and invertebrates. *Journal of the Acoustical Society of America*, 143(1), 470-488. https://doi.org/10.1121/1.5021594

Popper, Arthur & Hawkins, Anthony & Fay, Richard & Mann, David & Bartol, Soraya & Carlson, Thomas & Coombs, Sheryl & Ellison, William & Gentry, Roger & Halvorsen, Michele & Løkkeborg, Svein & Rogers, Peter & Southall, Brandon & Zeddies, David & Tavolga, William. (2014). Sound Exposure Guidelines. 10.1007/978 -3-319-06659-2_7.

Ren, H., Halvorsen, M. B., Deng, Z. D., & Carlson, T. J. (2012). Aquatic Acoustic Metrics Interface Utility for Underwater Sound Monitoring and Analysis. Sensors, 12(6), 7438 - 7450. https://doi.org/10.3390/s120607438

Slabbekorn, H., Bouton, N., van Opzeeland, I., Coers, A., ten Cate, C., & Popper, A. N. (2010). *A noisy spring: The impact of globally rising underwater sound levels on fish.* Trends in Ecology & Evolution, 25(7), 419-427. https://doi.org/10.1016/j.tree.2010.04.005

Southall, B.; Bowles, A.; Ellison, W.; Finneran, J.; Gentry, R.; Greene, C. Jr.; Kastak, D.; Ketten, D.; Miller, J.; Nachtigall, P.; Richardson, W.; Thomas, J.; Tyack, P. (2008). Marine Mammal Noise Exposure Criteria: Initial Scientific Recommendations. Aquatic Mammals, 33(4), 273-275. https://doi.org/10.1080/09524622.2008.9753846

Acknowledgement:

We would like to thank CUNY’s Office of Research, Hudson River Park Trust, and LaGuardia QSTEM Summer Academy for supporting this work. We couldn’t have achieved those findings without the hard work of all students involved with this research: Justen Gallagher, Jiale Lin, Melvin Schwartzbart, and many other students from previous research activities that laid solid foundation for this project. Special thanks to the Park staff Michaela Mincone, Siddhartha Hayes and Beryl Kahn for their help with the logistics for our on-site experiment and project presentation.