**INfluence of Bridge Structural Arrangement on the Noise Induced by Traffic and Its Effect on the Use of the Migration Route by Wildlife**

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**ABSTRACT**

At present, specialized structures for mitigation of habitat fragmentation caused by transport infrastructure are built on motorways in Europe on regular basis. These structures, ecological bridges, are built on locations, where the permeability of the new capacitive transport route (mainly motorway) for migration of wildlife cannot be achieved by underbridges which are built where the motorway (railway, etc.) crosses valleys, creeks, rivers or rural roads. During the past years, several sections of newly built motorways in the Czech Republic were monitored before construction, during it and after the opening. Special emphasis was put on the effect of the structural arrangement of the bridge on the usage of the migration corridor by wildlife. The effect of landscaping was studied on one hand, on the other one, the structural arrangement was noted. During inspections, the noise induced by traffic was noted. Under the bridges, the noise is produced mainly by wheels of heavy vehicles crossing the expansion joint. This noise is sharp and quite irritating even for human ears, which are much less sensitive than ears of wild animals.

According to the gathered data an original program of noise measurements was developed for quantifying the effect of noise induced by traffic on the usage of underbridges for migration of wildlife. The program covered 8 bridges along the D47 and R35 motorways in the Czech Republic and will be widened this spring. At every structure, noise was measured approximately 50m from the bridge on the ideal axis of migration and under the bridge in the ideal axis of migration; simultaneously the noise was recorded in the span adjacent to the expansion joint, where the effect of traffic was the strongest. The equivalent noise levels under the bridge were recorded and compared to the equivalent noise level 50m from the bridge, the effect of noise barriers was noted, together with the effect of landscaping close in the surroundings of the structure. The peak noise levels under the bridge were recorded and put connection to the arrangement of the expansion joints and bearings thus providing a multi-criteria approach to the issue.

The measurements provided very useful data, which helps to understand the influence of bridge structural arrangement, mainly bearings and expansion joints, and location of the bridge in the countryside on the noise induced by traffic. The peak noise levels reached up to 110 dB under selected bridges, the equivalent noise levels reached approximately 60-65dB.

The effect of noise will be incorporated into the new methodology for calculation of the migration potential of bridges in the following research.

**INTRODUCTION**

Noise induced by traffic on capacity highways or motorways influences the use of under-bridges as migration profiles. This effect has been spotted in the previous years during field observations, but it has not been quantified yet.

During the past years, several sections of newly built motorways in the Czech Republic were monitored before construction, during it and after the opening. Special emphasis was put on the effect of the structural arrangement of the bridge on the usage of the migration corridor by wildlife. The effect of landscaping was studied on one hand, on the other one, the structural arrangement was noted. As a part of the long time monitoring, a set of noise measurements was outlined and performed. The measurements were undertaken in on the D47 (now D1) and R35 motorways in the Czech Republic. These are planned, built and maintained by the Road and Motorway Directorate of the Czech Republic (RSD).

For an overview of the road and motorway network of the Czech Republic, see Figure 1; orange thick lines represent motorways under operation, grey lines represent planned motorways.
The monitored motorways, the D1 and R35 (D for motorway, R for expressway; a dual carriage-way with less strict design parameters and safety features) are connecting the cities of Ostrava and Olomouc, the 3rd and 6th biggest city in the Czech Republic. For a detailed view, see Figure 2; the green lines represent the section of the D1 motorway, which were set into operation in the previous year heading east from Ostrava. The R35 expressway provides the continuity of the capacity connection, before the red marked section (going in very sensitive landscape) is built.

The measurements were performed in the summer 2010 and were focused mainly on quantification of the effect of landscaping and the structural arrangement of the bridge on the propagation of noise induced by traffic. Special emphasis was put on the effect of used expansion joints and bearings on the noise impulse produced by heavy vehicles.

**LAYOUT OF THE NOISE MEASUREMENTS**

The motivation of the noise measurements can be summarized as follows: to quantify the effect of noise induced by traffic on capacity highways or motorways on the use of under-bridges as migration profiles. The noise further from the bridge can completely prevent the animals from using the under-bridge for migration (when passing around the fenced highway), while the noise impulse under the bridge induced by heavy traffic crossing the expansion joint can shock the migrating animal and makes it change the originally desired migration route.

Figure 3 provides a comprehensive illustration of the motivation. The arrow No. 1 represents the original migration need of the animals for crossing the motorway. If the highway is properly fenced and equipped with guiding measures, the animals are lead to migration profiles, such as overbridges and underbridges, arrow No. 2. If the noise further from the bridge, and other disturbing agents, such as traffic under the bridge, improper landscaping, etc., does not prevent the animal from using the migration route, it nears the under-bridge, arrow No. 3. Close to the bridge, the noise impact from heavy traffic crossing the expansion joint can shock the animal and make it change the original desired migration route, arrow No. 4.
Figure 2. Detailed situation of the D1 motorway and R35 expressway (www.rsd.cz).

Figure 3. Motivation of the noise measurements.
The noise further from the bridge can be minimized by noise barriers. These are very expensive and the effectiveness is questionable.

The noise from expansion joints and bearings can be mitigated by: the design of integral bridges (can be used for total bridge lengths up to 120m), and/or the design of proper landscaping under and around the bridge. The landscaping can involve e.g. noise reducing ramparts, vegetation and mainly refraining from the design of bridges with small clearance.

Regarding all the facts listed in the previous paragraphs, the measurements must be designed so that they produce data both about the noise level in the vicinity of the under-bridge and the noise impact underneath and/or close to the bridge. The arrangement of the noise measurements is explained in Figure 4.

As noted in Figure 4, the measurement point No. 1 produces the noise level in the vicinity of the under-bridge, while the measurement point No. 2 produces the noise impact underneath and/or close to the bridge. Two sound meters are used:

- No. 1 for measuring the equivalent noise level 30-50m far from the bridge, where the animal decides whether to use the migration route or not, and
- No.2 for measuring the noise impact underneath and/or under the bridge.

The measurements were performed simultaneously at both measurement points, 30 minutes at each studied bridge.

For evaluation of the noise measurement data, the methods of frequency weightenings are used. The most common weightening, the A weightening, cuts off the low and high frequencies the human ear cannot hear (the ear responds more to frequencies between 500 Hz and 8 kHz). By using this method, the noise meter measures the same what the human ear can hear. The second most used weightening method is the C weightening which does not cut the lower frequencies the same way as the previous method does. Both methods were used during the performed measurements.

The most used sound measurement parameter is the average sound level, the Leq. It is an average value of the sound over the period of the measurement, carrying the total energy of the measured noise, and thus indicating the potential hearing damages. The average sound level can be written as LAeq or LCeq according to the frequency weightening method.

The peak sound level is the maximum value produced by the sound pressure. This value is usually C-weightened and referred to as LCpeak in dB(C).

At both measurement points, the impacts caused by heavy traffic crossing the expansion joint, the peak sound level LCpeak, were measured and compared to the average sound levels LAeq and LCeq.
At the measurement point No. 2, the time distribution of the peak sound level was recorded, so the frequency of the passing heavy vehicles and its influence on the noise distribution close to the bridge could be studied.

For typical example of the measurement arrangement, noise measurement point No. 1, see Figure 5; the measurement point No. 2 is located in the middle of the second span, underneath the noise barrier.

![Figure 5. Typical example of the noise measurement arrangement, noise measurement point No. 1, bridge S0206 D4704 (with noise barriers).](image)

**CHOICE OF THE BRIDGES FOR NOISE MEASUREMENT**

The noise measurements were limited on the newly opened portion of the D1 motorway (sections D4704 and D4705), so the results would not be influenced by the effect of material deterioration of the bridges itself and their expansion joints.

The under-bridges were selected in a way that each one of the selection represents one “family” of bridges from the point of view of:

- suitability for satisfying migration needs,
- structural system,
- landscaping around the bridge (type of the landscape, proximity of other transport infrastructure and buildings, etc.),
- landscaping under the bridge (pavement of the embankments and surface under the bridge, roads, railroads, watercourse, etc.),
- noise reduction measures on the bridge or its vicinity.

As mentioned in the previous section, the noise measurements take 30 minutes at each studied bridge. Due to the great time demands, which include transfer to the next measurement point, (usually from 2-10km), choice of the measurement points, preparation of the measurement, the measurement itself and dismantling of the devices, choice of the studied bridges had to be deliberate, level-headed and based according to a large field survey.
From 20 under-bridges of the sections D4704 and D4705 (which represent 16% of the total 33km length) 8 ones were chosen:

- typical 3-span bridge, adequate clearance, noise barriers, S0206 D4704,
- buried structure, S0207 D4704,
- 2-span bridge, very small clearance, S0208 D4704,
- 4-span bridge, very small clearance, S0212 D4704,
- 10-span (317m) bridge, adequate clearance, S0220 D4704,
- 21-span (857m) bridge, adequate clearance, noise barriers, S0215 D4705
- The ecological tunnel at Dolni Ujezd (R35 expressway, section R3511) and the ecological bridge at Suchdol nad Odrou (D1 motorway, section D4705) were added to the selection because they represent specialized over-bridges designed for migration of wildlife.

**EXAMPLE OF THE MEASUREMENT ARRANGEMENT AND ITS RESULTS, THE ECOLOGICAL TUNNEL AT DOLNI UJEZD**

The ecological tunnel Dolni Ujezd was included in the measured set of bridges not only for the reason mentioned in the previous section, but also for the reason, that it is the first ecological tunnel built in the Czech Republic (1999). It serves its purpose well, but with limitations, it leads from a large forest area, in a forest spur which ends with a rock cut-off of the district road II/437. Accordingly, all migrations must take place at the east side of the south tunnel portal, where the animals run down a steep slope which leads to the meadows northern from the village of Dolni Ujezd, see Figure 6.

![Figure 6. South portal of the ecological tunnel at Dolni Ujezd, see the migration path and both measurement points.](image)

The noise measurement point No. 1 was located in the spot, where the steep slope from the tunnel embankment gets gentle; the measurement point No. 2 was located at the tunnel portal, where the exiting heavy vehicles produce large noise impacts.

While the peak sound level LCpeak reaches up to 102,5dB (e.g. air hammer, discotheque) at the tunnel portal (measurement point No. 2), 50m further from it (measurement point No. 1, at the embankment) the value is 86,5dB (e.g. very loud music, cocks crowing). The average sound level goes down from LAeq 71,03dB and LCeq 76,6dB at the portal to LAeq 62,8dB and LCeq 73,1dB at the embankment. For example of the results, see Figure 7.
The Czech hygienic limits, 60dB at day, 50dB at night, are not satisfied, but the migration route is still operational. This fact is induced by an easy accessible food source.

**SUMMARY OF THE RESULTS**

This section summarizes the measurements program. For each bridge (see Table 1), the values of $LC_{peak}$ and $LA_{eq}$ are provided at all measurement points, followed by a brief description of the expansion joint and arrangement around and underneath the bridge.

The presence of noise reduction measures is noted; the underlined numbers note noise measurement points close to the bridge. Bold indicates satisfied hygienic limits for noise at day.

According to the data presented in Table 1, the biggest role in distribution of traffic-induced noise can be attributed to the clearance of the bridge above from the ground, see SO 208 and SO 212, where the peak sound levels under the bridge reach up to 109.7 dB.

Noise barriers can reduce the average sound level, but are not effective in mitigating the sound impact produced by heavy vehicles crossing the expansion joints, compare SO 206 D4704 and SO 215 D4705 to SO 220 D4704.

As spotted on the ecological tunnel at Dolní Ujezd and the ecoduct at Suchdol nad Odrou, special emphasis has to be attributed to mitigation of sound impact at tunnel portals close to migration routes.
Table 1. Summary of the noise measurements.

<table>
<thead>
<tr>
<th>Bridge</th>
<th>Mostní závěr</th>
<th>Measurement point</th>
<th>LCpeak [dB]</th>
<th>LAeq [dB]</th>
<th>Noise barriers / arrangement underneath / remarks</th>
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<tbody>
<tr>
<td>Dolní Újezd R3511</td>
<td>buried structure</td>
<td>1</td>
<td>102,5</td>
<td>71,0</td>
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<tr>
<td></td>
<td></td>
<td>2</td>
<td>86,5</td>
<td>62,8</td>
<td></td>
</tr>
<tr>
<td>SO 206 D4704</td>
<td>1 lamella</td>
<td>3</td>
<td>95,9</td>
<td>62,1</td>
<td>Yes, stone pavement</td>
</tr>
<tr>
<td></td>
<td></td>
<td>4</td>
<td>69,8</td>
<td>51,8</td>
<td></td>
</tr>
<tr>
<td>SO 207 D4704</td>
<td>buried structure</td>
<td>8</td>
<td>95,1</td>
<td>56,6</td>
<td>No, closer measurements under the highway level, closer at it</td>
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<tr>
<td></td>
<td></td>
<td>10</td>
<td>79,9</td>
<td>58,1</td>
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<td>9</td>
<td>97,8</td>
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</tr>
<tr>
<td></td>
<td></td>
<td>11</td>
<td>94,4</td>
<td>66,4</td>
<td></td>
</tr>
<tr>
<td>SO 208 D4704</td>
<td>elastic</td>
<td>5</td>
<td>105,9</td>
<td>64,9</td>
<td>No, measurement No. 7 under the uncovered median</td>
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<tr>
<td></td>
<td></td>
<td>6</td>
<td>84,7</td>
<td>62,6</td>
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<td></td>
<td>7</td>
<td>106,3</td>
<td>67,6</td>
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<tr>
<td>SO 212 D4704</td>
<td>5 lamellas</td>
<td>12</td>
<td>109,7</td>
<td>72</td>
<td>No, very small clearance</td>
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<td></td>
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<td>14</td>
<td>92,8</td>
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<td>13</td>
<td>103,3</td>
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<td></td>
<td></td>
<td>15</td>
<td>90</td>
<td>64,7</td>
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<tr>
<td>SO 220 D4704</td>
<td>3 lamellas</td>
<td>21</td>
<td>93,3</td>
<td>64,4</td>
<td>No, bridge ca 7m above the ground</td>
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<td>22</td>
<td>87,1</td>
<td>61,2</td>
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<td>16</td>
<td>93,4</td>
<td>60,5</td>
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<td></td>
<td>19</td>
<td>81,3</td>
<td>59,6</td>
<td></td>
</tr>
<tr>
<td>Suchdol nad Odrou D4707</td>
<td>buried structure</td>
<td>24</td>
<td>97,1</td>
<td>68,2</td>
<td>No, measurement at the tunnel portal and at the centre-line</td>
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<td>25</td>
<td>81,2</td>
<td>58,7</td>
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</table>
CONCLUSIONS

This paper presented outcomes of a newly developed noise measurement program which was performed in year 2010 at the D1 motorway and R35 expressway. The results showed that improper landscaping, bad structural arrangement and missing noise barriers can greatly influence the noise induced by traffic passing an overbridge. This can then result into lesser effectively of the under-bridge as a migration profile. The biggest role in distribution of traffic-induced noise can be attributed to the clearance of the bridge above from the ground and the type of the expansion joints used. The effect of noise will be incorporated into the new methodology for calculation of the migration potential of bridges in the following research.

ACKNOWLEDGEMENTS

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BIOGRAPHICAL SKETCH

Marek Foglar, born 1980, graduated at the Czech Technical University in Prague (CTU), Faculty of Civil Engineering in 2005. During his doctoral studies, he focused on fatigue performance of pre-stressed and reinforced concrete. His Ph.D. thesis “Strain development under cyclic loading” was awarded the first prize in the Czech Concrete Society (CBS) contest of an outstanding dissertation in 2008. Since 2008 he works at the Department of Concrete and masonry structures of the CTU as assistant professor. Besides from his academic career, he is a chartered engineer for bridges and is active in bridge design. He was the chief engineer for the bridges of Grade-separated connection of road II/468 and the industry area in Trinec-Baliny which was awarded the first prize in the contest of the Czech Chamber of Certified Engineers and Technicians in 2010. He runs a research project “Theoretical investigation of motorways and roads environmental impacts” supported by the Czech Science Foundation.
TRAFFIC NOISE DISTURBANCE IN IMPORTANT BIRD AREAS IN SWEDEN

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ABSTRACT

Previous research has pointed out the negative impact of traffic noise on wildlife adjacent to major infrastructure corridors. The effects of traffic noise on birdlife are well documented, but effects on other taxa are also described in literature. In a similar manner, traffic noise decreases the value of human recreation in natural environments. Noise emissions from roads can hence be seen as a considerable problem for nature conservation and outdoor recreation. Despite this strong scientific evidence, the impact of traffic noise in natural environments is rarely assessed, and even more rarely mitigated, in Swedish road planning. We propose a method for assessing the traffic noise impact on areas of importance for nature conservation, with special emphasis on important bird areas. The method is based on effect levels presented in literature, available habitat data, bird observation data, road data and a simplified model for noise distribution. We applied the method in two Swedish regions with different traffic intensity, in order to estimate the impact of traffic noise on birds on a larger geographic scale, and to identify conflict points at which mitigation efforts should be directed. According to the method, the impact zone covers only 3 % of the total land area in the region with relatively high traffic, but affects as much as 15.8 % of all bird lakes, 15.0 % of all high nature value grasslands, 8.7 % of all large open bogs, and 17.4 % of the region´s all high nature value deciduous woodland. The corresponding figures from the low-traffic region are; < 1 % of the total land area, 12.3% of bird lakes, 6.9% of high nature value grasslands, and 1.5 % of large open bogs (no measure was given for high nature value deciduous woodland in this region). Although most sites were only in part affected by the noise effect zone, the study still indicates that traffic noise may have a disproportionate impact on some important bird habitats, thereby making noise impact a more serious conservation issue than first expected. Because bird areas are often rich also in other taxa, and in addition tend to be important areas for outdoor recreation, we argue that the traffic noise may have a broad impact on nature conservation, and that mitigation efforts should be made to minimize this impact. The method presented here can be further developed into a tool for prioritizing locations for such efforts.

INTRODUCTION

One of the many adverse effects of traffic and roads on wildlife is noise disturbance. Previous studies have shown that several bird species occur in lower numbers in the vicinity of high traffic roads (reviewed by Reijnen and Poppen 2006, Benítez-López et al. 2009), and that a major cause of this effect is likely to be traffic noise (Reijnen et al. 1995, Forman and Alexander 1998, Reijnen and Poppen 2006, Parris and Schneider 2009, Barber et al. 2010, Kociolek et al. 2011).

Many animal species including birds, mammals, frogs, and insects use acoustic signals to attract mates, to defend territories, to maintain group cohesion, to hunt, and to warn from predators (Marler and Slabbekoorn 2004, Brumm and Slabbekoorn 2005, Warren et al. 2006). Anthropogenic noise compromises the function of such signals. For example, high levels of traffic noise may lead to difficulties for birds and frogs to attract mates (Reijnen and Poppen 1994, Bee and Swanson 2007), and may result in reduced foraging efficiency in bats (Schaub et al. 2008, Siemers and Schaub 2011).

Behavioral responses to traffic noise may in birds and frogs include singing/calling at higher pitch (to reduce the masking from low-frequency noise; Slabbekoorn and Peet 2003, Slabbekoorn and den Boer-Visser 2006, Wood and Yezrinac 2006, Parris and Schneider 2009, Parris et al. 2009) or at higher volume (Brumm 2004), altering the time spent singing/calling (Sun and Narins 2005, Díaz et al. 2011), and shifting to nocturnal singing (Fuller et al. 2007). Such adaptations may reduce the problem with masking but still involve costs in the form of physiological and energetic stress. Also, not all species have the possibility to behavioral adaptation. Hence, traffic noise may lead to reduced
reproductive success, increased mortality risk, and emigration, resulting in decreased population densities (Fletcher and Busnel 1978, Reijnen and Foppen 1994, Rabin et al. 2003, Patricelli and Blickley 2006, Lengagne 2008, Barber et al. 2010).

Along busy-traffic roads, more than half of the bird species may be negatively affected, and the effects are particularly strong in species of special conservation concern (Forman and Deblinger 2000, Reijnen and Foppen 2006). In a comparison among habitat types, wetlands, grassland and natural woodland appeared to have a larger proportion of affected species than arable land or urban habitats (Reijnen et al. 1996, Reijnen and Foppen 2006).

Traffic noise also decreases the value of human recreation in natural environments, urban green areas as well as more remote wilderness. Tranquility is increasingly perceived as an important landscape value (Shaw 1996, Health Council of the Netherlands 2006, National Board of Housing Building and Planning 2007). Technical noise causes an array of physiological and psychological effects in humans, such as raised stress levels, disturbed conversation and sleep, and increased ill-health (WHO 2000). Outdoor environments provide important opportunities for human physical exercise and psychological restoration (Grahn and Stigsdotter 2003, Ottosson 2007, National Board of Housing Building and Planning 2007), but technical noise has a negative impact on the value of outdoor recreation (Nilsson and Berglund 2006).

Interestingly, some of the most comprehensive studies on the effects of noise on outdoor recreation (Nilsson 2007) and wild bird fauna (Reijnen and Foppen 1995, Reijnen et al. 1996) show strikingly similar dose–response relationships, which reveals a promising prospect for a coordinated treatment of noise disturbance in environmental assessment and mitigation. The first discernible effects occur at noise levels of 42-47 dB $L_{Aeq}$, at 48-49 dB $L_{Aeq}$ the environmental quality (measured as perceived soundscape quality for people and breeding population density for birds, respectively) has dropped to 80% of that in the undisturbed surroundings, and at 55 dB $L_{Aeq}$ the environmental quality is halved. At higher noise levels, environmental quality further decreases asymptotically towards 10-20% of undisturbed surroundings.

Due to its impact on wildlife and outdoor recreation, noise emissions from transport infrastructure can be seen as a considerable problem for nature conservation. With the ongoing urbanization and the rise in motorization, and the resulting increase in anthropogenic noise in the landscape, the demand for efficient noise mitigation measures will further increase. However, until now, the impact of traffic noise in natural environments has rarely been addressed in Swedish transport infrastructure planning. Present noise regulations in Sweden (Swedish Government 2004) and the European Union (European Parliament and Council 2002) require traffic noise to be mitigated only in residential areas. Existing general (non-binding) advices on noise prevention in recreational areas, issued by Swedish transport authorities (Swedish Road Administration 2001), are rarely followed, even by the authorities themselves.

The lack of a practical method to assess the impact of traffic noise in natural environments may be one reason to this shortcoming. It has been argued that without a baseline assessment, the obstacle to implementation is twofold: i) there are no results to illustrate the overall severity of the problem, and ii) there are no means to identify priority sites for mitigation (Anders Sjölund, Swedish Transport Administration, pers. comm.).

In this paper, we propose a method for a regional scale assessment of road traffic noise impact on important bird areas. The method uses bird richness as a proxy for a more general conservation value of habitats. We present results derived from the method applied in two Swedish regions with different traffic densities, giving estimates of the impact of traffic noise on birds on a larger geographic scale, and indicate the locations of conflict points at which mitigation efforts should be directed. The significance of the results in a broader nature conservation perspective is discussed. We outline how the method will be further developed.

THE METHOD

In one sentence, the method maps the overlap between expected detrimental traffic noise levels and important bird areas. The method relies on a number of assumptions described as follows, and uses data derived from existing data bases.

Identification of the Noise Effect Zone

Traffic noise levels around larger roads were calculated using the so-called Nordic prediction method (Jonasson et al. 1996, Bendtsen 1999), the present standard mathematical model for noise prediction in Sweden. In order to make the model operational on a regional scale, we simplified it by assuming no topography, road surface in level, and soft ground surface surrounding the road. We obtained data on traffic densities (separated between light and heavy vehicles) and signed speed limits from the Swedish Data Base for Road Traffic (administered by the Swedish Transport...
Administration). Roads <70 km/h and <3000 vehicles/day were excluded, because no significant noise impact on birds has been proven for minor roads (Reijnen and Foppen 2006). Based on results from previous research (see Introduction), the noise effect zone was defined as the zone with traffic noise ≥45 dB L_aeq,24.

### Selection of Important Bird Areas

In order to identify bird areas of conservation concern, we selected the following habitat types, based on assumed proportion of bird species affected (following Reijnen et al. 1996, Reijnen and Foppen 2006), bird richness, or importance for particular species of conservation concern. Here we also describe the selection criteria and data bases used for each habitat type.

- **Bird lakes (Fig. 1a)**

These are shallow lakes with generally high biodiversity and bird density, hosting a large number of wetland bird species, many of which are red-listed. We defined bird lakes as areas where at least 4 of 8 selected indicator species were observed during breeding period in the last 5 years. The indicator species were pochard (*Aythya ferina*), shoveler (*Anas clypeata*), horned grebe (*Podiceps auritus*), coot (*Fulica atra*; only breeding birds), black-headed gull (*Larus ridibundus*; only breeding birds), grasshopper warbler (*Locustella naevia*), reed warbler (*Acrocephalus scirpaceus*), and sedge warbler (*Acrocephalus schoenobaenus*). We derived bird observation data from the national on-line report system for birds at the Species Gateway ([http://www.artportalen.se/birds](http://www.artportalen.se/birds)), administered by the Swedish Species Information Centre. In larger lakes where the selection criteria were only valid for parts of the lake, we arbitrarily delimited the bird lake part on the basis of bird observations. We also included a 100 m buffer around the actual water surface, in order to include the valuable riparian habitat.

- **High nature value (HNV) grasslands (Fig. 1b)**

These are semi-natural grasslands – pastures and meadows – that host a large proportion of the declining agricultural bird guild. Occurrence of this habitat type was derived from the Swedish grassland data base TUVA ([https://etjanst.slv.se/tuva2/site/index.htm](https://etjanst.slv.se/tuva2/site/index.htm)), administered by the Swedish Board of Agriculture. Because these grasslands in Sweden are often small in size, we lumped areas within <500 m from each other, included a 250 m buffer zone around each grassland, and defined these as systems of HNV grasslands. We excluded systems <50 ha.

- **Large open bogs (Fig. 1c)**

Generally poor in species and individuals, but hosting a number of bird species that are characteristic for the boreal region. Occurrence of this habitat type was derived from the Swedish Wetland Survey, with data presented at the Environmental Data Gateway ([http://gpt.vic-metria.nu/GeoPortal/#/startMenu](http://gpt.vic-metria.nu/GeoPortal/#/startMenu)), administered by the Swedish Environmental Protection Agency. We excluded bogs <30 ha.

- **High nature value (HNV) deciduous woodlands (Fig. 1d)**

Woodlands dominated by what in Scandinavia is referred to as “noble” desiduous trees (*Quercus, Ulmus, Fagus, Tilia, Acer, Fraxinus, and Carpinus*). Such woodlands generally have a high biodiversity and bird density, and host several bird species on the red-list. Occurrence of this habitat type was derived from the Swedish forest data base Skogens Källa ([http://www.skogsstyrelsen.se/Aga-och-bruka/Skogsbruk/Karttjanster/Skogens-Kalla](http://www.skogsstyrelsen.se/Aga-och-bruka/Skogsbruk/Karttjanster/Skogens-Kalla)), administered by the Swedish Forest Agency. We selected only areas with special protection value according to the data base. Because such woodlands in Sweden are typically small in size, we lumped areas within <500 m from each other, included a 250 m buffer zone around each woodland, and defined these as systems of HNV deciduous woodland. We excluded systems <50 ha.
Figure 1. Habitat types with assumed large importance to birds, and therefore selected for the method: a) bird lake, b) high nature value grassland, c) large open bog, and d) high nature value deciduous woodland. Photo: Jan Olof Helldin.

The Case Study Regions

We applied the method in two Swedish regions (Fig. 2) with different traffic intensity:

- **Region Mid-Sweden (ca 117,000 km²)**

  This part of Sweden is sparsely populated (7.8 inhabitants/km²), with a density of the state road net of 18 km road/100 km². The natural environment in the region is typically boreal; hilly terrain covered by coniferous forest, and with a high frequency of bogs. The few agricultural areas, villages and towns are generally concentrated in narrow river valleys. HNV deciduous woodlands are largely lacking.

- **Region West Götaland (ca 25,000 km²)**

  This region is more densely populated with Swedish standards (66.1 inhabitants/km²), and include one of Sweden’s larger cities, Gothenburg. The density of the state road net in this region is 62 km road/100 km². Also in this region, hilly terrain with coniferous forest of boreal type is a dominant trait, but with nemoral/continental elements; agricultural land makes up a larger proportion, at some places concentrated in agricultural plains, and HNV deciduous woodlands are not rare.

It should be noted that neither of these regions can be considered densely populated, nor densely roaded, in comparison with e.g. western and central Europe, were road densities in most parts are considerably higher (Jaeger et al. 2011).
RESULTS

In the region Mid-Sweden, the total length of the roads included in the method was 1,570 km, and the width of the noise effect zone around these roads varied between 110 and 350 m on each side of the road. The noise effect zone covered 735 km$^2$, or 0.63 % of the total area of the region. The noise effect zone affected 12.3% of the bird lakes (Fig. 3), 6.9% of HNV grassland systems (Fig. 4), and 1.4% of the large open bogs in the region (Fig. 5; all percentages derived from figures in Table 1 row la-b). In most cases the noise effect zone overlapped with only part of the respective bird site, and the proportion of the total area of each habitat type within the noise effect zone was therefore considerably lower; 0.8% of the total bird lake area, 2.0% of HNV grassland, and 1.2% of large open bog (percentages derived from figures in Table 1, row lc-d). Due to the low occurrence of HNV deciduous woodland in the region, this habitat type was excluded from the analysis.

In the region West Götaland, the total length of the included roads was 2,624 km, and the width of the noise effect zone 95-1,410 m on each side. The zone covered 1,133 km$^2$ in total, or 3.28 % of the region’s total area. The noise effect zone affected 15.8% of the bird lakes (Fig. 3), 15.0% of HNV grassland systems (Fig. 4), 8.7% of the large open bogs (Fig. 5), and 17.4% of the region’s HNV deciduous woodland systems (Fig. 6; percentages derived from figures in Table 1 row IIa-b). The proportion of the total area of each habitat type overlapping with the noise effect zone was 2.5% of the total bird lake area, 4.8% of HNV grassland, 1.4% of large open bog, and 7.9% of HNV deciduous woodland (percentages derived from figures in Table 1, row Iic-d).

Table 1. Number of sites and area of selected habitat types in the two case study regions, and in the noise effect zone of each region.

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<th>Table 1. Number of sites and area of selected habitat types in the two case study regions, and in the noise effect zone of each region.</th>
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<td><strong>I. Region Mid-Sweden</strong></td>
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<td>a) Total no. of sites</td>
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<td>b) No. of sites affected by noise</td>
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Figure 3. Bird lakes affected by the noise effect zone (red dots) and outside the noise effect zone (dark blue dots) in Mid-Sweden (above) and West Götaland (below). The noise effect zone is shown in light blue.
Figure 4. High nature value grasslands affected by the noise effect zone (red areas) and outside the noise effect zone (yellow areas) in Mid-Sweden (above) and West Götaaland (below). The noise effect zone is shown in light blue.
Figure 5. Large open bogs affected by the noise effect zone (red areas) and outside the noise effect zone (brown areas) in Mid-Sweden (above) and West Götaland (below). The noise effect zone is shown in light blue.
DISCUSSION

The study shows that in these Swedish regions, also the relatively more densely roaded region, the predicted noise effect zone covers only a minor part (ca 1-3 %) of the total area. Still a considerably larger proportion of the important bird areas (e.g. 13.5 % of all bird areas identified with the proposed method in West Götaland) are to some degree affected by the noise effect zone. Because of the often interlinked ecology within habitat units, noise effects in only part of a larger area may create secondary effects on the area as a whole, e.g. by an increased pressure on the non-disturbed parts, or by reducing the total population size of certain species below a critical threshold.

Even if only considering the area actually in the noise effect zone, important bird areas may to some degree be more affected than the average landscape. In the present study, HNV deciduous woodland systems in West Götaland deviated most in this respect, with almost 8 % of the area covered by the noise effect zone. The reason to this pattern is probably to be found in landscape structure; both some bird rich habitats (semi-natural grasslands, HNV deciduous woodlands) and human settlements with associated roads tend to be concentrated in agricultural areas and river valleys. In effect, traffic noise may have a disproportionate impact on some important bird habitats.

Previous attempts to estimate the proportion of habitats ecologically affected by roads have presented larger figures than in the present study. Reijnen and Foppen (2006) concluded that between 8 and 20 % of bird habitat area in the Netherlands falls within a road effect zone, where noise is expected to be the dominating road-induced factor affecting bird density. Forman (2000) estimated that at least 20% of the land area in the United States is affected, but in that assessment a variety of taxa and effects were included (in addition to noise, also mortality and barrier effects on deer, hydrology, spread of exotic plants etc.), so the study from the Netherlands makes a better comparison to our study. Obviously, the Netherlands has on average a much denser road net and higher traffic densities than Sweden, which explains the higher proportion. However, our study indicates that also in regions with low or moderate road densities, a disproportionate impact on certain habitats can make noise disturbance a more serious issue than first expected.

Clearly one can argue that it is still only a few percent of the important bird habitat that is affected by the noise effect zone in the present two Swedish regions. Therefore, other pressures (in Sweden mainly habitat changes due to agriculture and forestry) probably have a larger general impact on bird populations. But just as obvious is the counter-argument: the noise effect may well be additive to other impacts, and therefore critical to bird conservation. Furthermore, according to principles stated in article 6(b) of the Convention on Biological Diversity (United Nations 1992), each sector is responsible for its own impact on biodiversity. Particularly where traffic noise affects the most important bird areas, it is of a potentially high conservation importance that noise emissions are mitigated.
This it turn points at the importance to locate the major conflict points, at which mitigation efforts should be directed. The maps in Figure 3-6 show the locations of the important bird areas identified with the proposed method, thereby giving a first and general picture of where to direct mitigation measures.

Although the habitat types in the present study were selected on the basis of value for birds, they are often also important habitats for other taxa, such as amphibians and bats (possibly with the exception of mires; Ahlén et al. 1995, Ahlén 2006), i.e. also for other taxa that may be negatively affected by traffic noise (see Introduction). Similarly, these habitat types are also expected to be of value for outdoor recreation, because of their accessibility, high biodiversity and wilderness qualities (National Board of Housing Building and Planning 2007). Hence, we argue that birds are reasonable indicators of nature conservation value in a broader perspective, and particularly in relation to noise disturbance. Although the species-rich and important bird taxon may well be reason enough for substantial conservation efforts, noise mitigation in important bird areas will probably have an even more multi-faceted conservation value.

In our future work, the method will be further developed into a tool for prioritizing locations for mitigation efforts. Based on conservation value or protection assigned by authorities, some additional habitat characteristics, and further elaboration of bird observation data – everything derived from the aforementioned data bases – we will calculate a standardized conservation value for each site. In addition, the distance to the noise source will be included in calculations of affected areas, to achieve a better-tuned estimate of relative habitat quality loss. For each site, the product of the standardized conservation value and relative habitat quality loss can be interpreted as the loss of conservation value, and be used to rank sites according to necessity for mitigation.

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Anna Jangius is an ecologist and GIS-specialist working as a consultant at Calluna AB, Sweden.

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MINIMIZING IMPACT PILE DRIVING HYDROACOUSTIC EFFECTS ON 14 SPECIES OF ENDANGERED SPECIES ACT LISTED FISH: TALES FROM A MEGA-PROJECT

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ABSTRACT

The Columbia River Crossing (CRC) is a large, complex project with multiple years of in-water construction to replace a bridge spanning 1.1 kilometer (0.7 mile) of the Columbia River. The Columbia River is a migratory corridor for 14 fish species listed under the Endangered Species Act (ESA), including two endangered salmon species. Year-round impact driving to install 2.4 meter (8’) diameter steel pile for bridge pier foundations was originally proposed, but would have produced noise levels resulting in exposure to critically imperiled interior basin listed fish creating the possibility of the project being deemed to jeopardize the continued existence of a species (i.e. a jeopardy opinion). An alternative, limiting in-water work to a 4-month period to avoid key interior basin migration, would concentrate the impact on less imperiled, but still federally listed lower river fish for several consecutive years, again raising the specter of a jeopardy opinion. In addition, this timing would extend the overall construction timeline to almost 14 years, impacting contract viability and dramatically increasing construction cost.

To develop a viable construction timeline while balancing impact minimization, the project team facilitated a solution through collaboration with regulatory agencies and internal project staff, and implementation of a test pile study to verify assumptions. Pier design and construction methods were modified based on regulatory agency input for prioritizing impact minimization to various fish species. Project timing was modified based on a project sequencing and exposure model that determined the optimal timing for in-water impact pile driving. Exposure modeling was based on conservative estimates of predicted noise levels and propagation. The analysis included weekly fish abundance and timing modeled by species and life-stage. Estimates were often problematic; especially for certain lower river fish populations and data inputs involved a considerable collaborative effort by experts from several agencies. A test pile study was implemented to obtain baseline data on pile installation noise levels, noise propagation, effectiveness of proposed noise attenuation, and constructability of proposed installation methods. The test pile study provided valuable information regarding the spatial extent of noise, as well as important installation timing and engineering information. The results of the study will be used to validate the conservative estimates of fish exposure and further refine estimates of impacts to listed fish species.

The reality of multiple imperiled listed fish species led the project to revise design and construction approaches to avoid or minimize impacts. Through extensive collaborative efforts internally and with resource agencies, an in-water impact pile driving work window was identified. The results of the modifications and collaboration were: 1) Avoidance of a jeopardy opinion, 2) A viable construction timeline, and 3) Receipt of a biological opinion in less than 7 months.

Permitting success for other projects with unknown, but potentially high impacts, may hinge on a high level of intra-project and inter-agency collaboration. Modeling potential exposure and pre-project studies can be effective tools to advance collaboration by obtaining site specific information that can improve constructability assumptions and significantly improve regulatory approval uncertainties, timelines, and the need for overly conservative assumptions regarding potential impacts.
INTRODUCTION

The Columbia River Crossing (CRC) project will replace the Interstate-5 (I-5) bridges over the Columbia River between Portland, Oregon, and Vancouver, Washington, as well as upgrade multiple interchanges and add light rail (Figure 1). Two new bridges will span 1.1 kilometer (0.7 mile) of the Columbia River mainstem and five new bridges will span a side channel, the North Portland Harbor. Multiple years of in-water work are required to construct the new bridge foundations and superstructures, and remove the existing structures.

![Figure 1. Project area map showing location of replacement bridges.](image)

Design of the bridges is constrained by several factors including the need to minimize intrusion into the airspace for two airports while maintaining clearance for shipping traffic, minimize intrusion into cultural and historic resources, minimize impacts to downtown Vancouver, maintain a usable grade for light rail, pedestrians and bicycles, tie into the existing roadway infrastructure, and minimize the size of the permanent in-water footprint. In addition, in-water construction activity is constrained by a requirement to maintain at least one open navigation channel. The project is located in the lower Columbia River near river kilometer 171 (river mile 106). The Columbia River contains 13 species of federal Endangered Species Act (ESA) listed salmon and eulachon (Thaleichthys pacificus). Two species are listed as endangered and the long-term (100-year) extinction risk for many of the listings is rated as high or very high.
All species of listed fish are present in the lower river during at least a portion of the year, including the November through February established in-water work window for the lower river. Most juvenile outmigration between Bonneville Dam (river kilometer 146 [river mile 91]) and the mouth of the river occurs outside this work window between March and October, with peaks at various times within this period, depending on species and run type (Carter et al. 2009). However, for eight of the listed species, migration timing extends outside of the March through October period and into the established window for in-water work construction.

In-water work for mainstem bridge construction was initially estimated to take 3.75 years. To accomplish this timeline, 96, 8-foot-diameter steel piles for the mainstem bridge foundations would need to be impact driven for a 2- to 3-year period with no in-water work restrictions. Impact pile driving produces impulsive sounds that can elevate underwater sound to levels that may cause behavioral disturbance, injury, or mortality in fish (Popper and Hastings 2009). The level of effect depends on numerous factors, including the intensity, frequency, and duration of sound and the size of the fish exposed (Popper et al. 2006). Because large pile sizes were proposed for long durations and the in-water construction timing would overlap with listed fish species, potential project impacts from pile driving were of particular concern to NMFS and the U.S. Fish and Wildlife Service (USFWS), who have regulatory oversight over federally listed fish. These impacts were also of concern to the Oregon Department of Fish and Wildlife (ODFW) and Washington Department of Fish and Wildlife (WDFW), who issue recommendations or permits for the timing of in-water work. Conducting all impact pile driving work utilizing the existing 4-month window was estimated to extend bridge construction to 7 to 7.5 years and the total project timeline to almost 14 years. From a project perspective, the extended timeline was considered unworkable due to the increased construction costs and associated construction contracting issues.

The CRC project team was faced with the challenge of obtaining an in-water work window (IWWW) that the four resource agencies would approve, while maintaining a construction timeline that would result in a viable project. This paper focuses on the collaborative inter-agency effort to address listed fish exposure to impact pile driving noise levels, including changes to bridge design and construction techniques, as well as the timing of in-water impact driving.

**INTERAGENCY WORKGROUP**

Early in the design phase of the project, CRC developed the Interstate Collaborative Environmental Process (InterCEP) for the project’s NEPA review with the goal of early agency coordination (CRC 2006). As part of the InterCEP process, a subgroup consisting of CRC environmental and engineering staff and NMFS, USFWS, ODFW, and WDFW representatives was formed to collaborate on in-water work timing and minimization of impacts to fish and fish habitat, especially in regard to those listed under the ESA. The workgroup met regularly over a 2.5-year period to discuss several aspects of the project. The primary impact of concern was potential exposure of fish to harmful levels of underwater noise from impact pile driving.

To evaluate the risk of exposure, the CRC project team working with NMFS modeled the expected extent of underwater sound levels over the established thresholds for fish (FHWG 2008) for stationary and moving fish. The variables needed for this analysis are fish speed, sound source levels, pile strike numbers, a transmission loss coefficient, and attenuation levels to model the distance to the threshold for the potential injury for impulse noise. Attenuation of impact pile driving noise would be achieved through use of a noise attenuation device, such as a bubble curtain or cofferdam. No data were available for pile installation source levels, pile strike numbers, transmission loss, and achievable attenuation levels in the Columbia River, so conservative assumptions were used to estimate all input variables using available data from other projects. The extent of the exposure was overlaid with coarse estimates of fish abundance and timing in the project area and projected project sequencing.

Based on the initial modeling results and feedback from the InterCEP workgroup, additional measures were needed to avoid or further minimize potential hydroacoustic impacts to listed fish species. NMFS discussed the relationship of the CRC project’s potential impacts to those where incidental take was already allotted in the Federal Columbia River Power System Biological Opinion (NMFS 2008) analyses and conclusions. Specifically, NMFS did not want to authorize any incidental take of interior basin listed species due to the level of incidental take already allotted in the Columbia River Basin. Impacts to particularly sensitive interior basin species could potentially result in NMFS issuing a jeopardy opinion under the ESA. A jeopardy opinion would result in an alternative to the current project design and significantly delay the project timeline.

To address the high level of potential hydroacoustic impacts and avoid interior basin threatened or endangered species, the project team 1) Re-evaluated the project design to significantly reduce overall project impacts, and 2) Re-evaluated...
the proposed IWW with no restrictions to a more restrictive IWW for impact pile driving. In order to accomplish these tasks, the project team produced a more sophisticated analysis of fish exposure and conducted a test pile study to provide more confidence in model assumptions.

PIER DESIGN AND CONSTRUCTION REVISIONS

To reduce the hydroacoustic impacts associated with impact driving 96, 2.4 meter (8') diameter piles, the CRC engineering team revised the pier foundation design and the pile installation method. The revised design contains 88, 3 meter (10') diameter drilled shafts for the main channel bridge foundations. The design change results in the elimination of impact driven piles for bridge foundations; however, temporary work platforms for support equipment necessary to install the foundation shafts and construct the superstructures are needed. The temporary work platforms require the installation of up to 1,200 temporary hollow steel piles ranging in size from 0.61 to 0.91 meters (24" to 48") in diameter.

Based on output of the hydroacoustic exposure model, further reducing potential hydroacoustic exposure to listed fish from the installation of the temporary piles was necessary. The result was a measure incorporated into the project requiring piles to be placed with a vibratory driver or pressed-in; using an impact hammer only to verify each pile's load bearing capacity as necessitated by project specifications. Vibratory pile driving produces less intense sound levels with the expectation that there is less potential for fish injuries (Carlson et al. 2001 and Nedwell and Edwards 2002). Incorporation of this measure, in addition to the change to drilled shafts for pier foundations, resulted in reduction of daily impact pile strikes by over 90 percent from the original impact only driving of the 2.4 meter (8') diameter piles.

IMPACT PILE DRIVING IN-WATER WORK TIMING

To avoid impacting interior basin species, the project team evaluated potential species exposure iteratively using variations of the in-water work window for impact pile driving. This window is referred to as the hydroacoustic in-water work window (HIWWW). This evaluation occurred for bridge construction occurring in the mainstem river and not for bridge construction in North Portland Harbor. Overall project duration was evaluated with each variation.

Hydroacoustic Exposure Analysis

To evaluate a given HIWWW, the potential number, species, and life stage potentially impacted, as well as the extent, duration and timing of the impact, was needed. The CRC team conducted the following three part analysis to model potential hydroacoustic impacts to the 14 listed fish species.

1. Model listed species timing and abundance.
2. Model projected impact pile driving and duration by day, by week, and for each year of in-water construction for a given HIWWW.
3. Model potential exposure above the potential injury threshold for each species, at both juvenile and adult stages, by week, by year, and cumulatively over the life of in-water construction.

Each of the three steps occurred concurrently and each analysis evolved and was refined as more data or information contributing to each element became available. Each part of the analysis is outlined in this paper along with a comparison of HIWWWs evaluated. Detailed information on each step, data sources and corresponding references can be found in the CRC Biological Assessment (CRC 2010).

1. Fish Timing and Abundance Modeling.

Although the Columbia River Basin has a long history of research, management, and commercial fishing, compiling datasets of fish timing and abundance presented the project team with a challenge. Not all species or life stages had a dataset available, and many of the datasets had variable formats and periods of record. Most datasets are from Bonneville Dam or Willamette Falls (Sullivan Dam) and consist of daily or weekly counts. However, dam counts are done at a species or run level and ESA listed fish are listed at a Distinct Population Segments (DPS: steelhead [anadromous Oncorhynchus mykiss] and eulachon) or Evolutionarily Significant Units (ESU: Pacific salmon) level. Datasets were not readily available for lower Columbia River ESUs and DPS that do not pass the dam, or do not pass the dam in entirety. When counts at Bonneville Dam were limited or potentially represented patterns not typical of fish most likely to pass through the project area, data from tributaries (e.g., coho [O. kisutch] counts at Marmot Dam on the Sandy River) were assessed. In addition, some DPSs or ESUs are further divided into natural origin and hatchery components and others are further divided into management units by the state (e.g. Group A and B steelhead). Eulachon presented an additional challenge because only data from landings or date of first entry are available.
Figure 2 depicts groupings within the ESU/DPS level. Where timing is primarily uniform at the ESU/DPS level, datasets were modeled at this level (e.g., eulachon and chum salmon [O. keta]). Where timing varied among runs in an ESU or DPS, datasets were modeled for each subcategory that had prominent differences. For example, the Lower Columbia River steelhead DPS has summer and winter runs, as well as natural and hatchery runs that vary in timing; therefore, each run was modeled.

Working in collaboration with the InterCEP workgroup, and with considerable contributions and support from staff of WDFW, ODFW, NMFS, and Portland General Electric, the CRC project team evaluated available data for all 14 listings at the DPS/ESU level, as well as their associated subgroups, for both adults and juveniles (Figure 2). Project-specific fish timing curves for the analysis were compiled from existing data at the ESU or DPS level. The analysis approach initially focused on exploration of the available data and preliminary estimates of the maximum likelihood of species timing and duration at the CRC project area. Maximum likelihood, as used here, means that estimates are based on patterns that represent a better than average or typical year. These initial estimates were based on an average (typical) value for a week during the species migration and on the variability of this value for the period of record. Where possible, the most recent 10-year period was examined. Weekly abundance was translated into run indexes, where run indexes were defined as the percent of the annual total count occurring each week. A set of weekly values was estimated by adding one standard deviation ($\sigma$) to the weekly average value across years ($\bar{x}$). The relationship between weeks was modeled to estimate duration and timing patterns by fitting a polynomial curve to $\bar{x} + \sigma$. This standard deviation represents the variation within a given week (across years) for the period of record. In this way, weeks with highly variable run indexes are represented as having greater run indexes than the statistical average for that week. For example, Weeks 21 and 26 could both have average run indexes of 0.400 but generate different weekly values for modeling if $\sigma$ was 0.002 and 0.120, respectively.

Preliminary curves were reviewed by agency biologists for general agreement on the approach described above. Further refinements were made to the curves with supplemental data sources. Additionally, NMFS was interested in an approach that estimated timing and duration based on weekly maximum abundance in the period of record in lieu of using an average plus one standard deviation. Therefore, the analysis approach was revised to further emphasize the maximum historic abundance by week using the greatest (most abundant) year in the period of record as the basis for calculating weekly run indexes. Raw weekly abundance was transformed as a fraction of the greatest annual abundance for the period of record. This treatment gives weight to weeks of relatively greater abundance in years with the largest populations. It also minimizes the effect of high weekly abundance in years of low annual abundance on curve fit and overall strength of estimates.
Information on weekly variation was used to model timing duration and patterns. A polynomial was fit to the weekly $\bar{x}$ plus one, two, or three standard deviations as needed to best fit weighted weekly maximum values for the period of record. Values were normalized to $\bar{x} + \sigma$ to provide a comparison of values and curve fit. An example of relationship between weighted and normalized values fit with a polynomial curve is provided in Figure 3.

Steelhead and Chinook (O. tshawytscha) were the most complex groups. For illustration of their complexity, timing estimates for all steelhead DPSs are grouped in Figure 4 and spring and fall runs of Lower Columbia River (LCR) Chinook are depicted in Figure 5. Both figures represent the grouping levels used in the hydroacoustic analyses. Unique management references, such as Group A and B summer-run steelhead (see WDFW 2009), and distinctions between natural origin and hatchery populations were assessed with each analysis, as appropriate.

Once the curves were developed for juveniles and adults, estimates could be made for the proportion of each ESU/DPS that could occur any week of the year. This weekly ESU/DPS proportion was an input variable in the hydroacoustic exposure analysis. For each curve developed, agency personnel reviewed and agreed on the data.

In addition, the relative abundance of salmon and steelhead at the basin scale was portrayed for all analyzed groups (Figure 6). Timing is depicted as 52 weeks and shown as radials. Abundance is the percentage of each unit that may be expected in any week and is depicted as distance from the center of the figure. At this level, timing patterns indicate weeks of minimal presence, weeks of maximum abundance, and the complexity of time-clustered migrations of fish. Overlaid with a proposed in-water work window (green), this diagram provided a view of major groups most affected by a proposed in-water work period and proved a useful visual aide and discussion tool. The 31-week in-water work window depicted in the figure shows interior basin groups are not likely to be impacted and the lower Columbia group will be the most impacted by the window.
Figure 4. Estimated timing of all ESA-listed steelhead adults and juveniles passing through the CRC project area.

Figure 5. Estimated timing of LCR Chinook adults and juveniles at the CRC project area.
Note: Weeks are radials; abundance is the percent of listed fish present each week as distance from the center. All ESA-listed salmon and eulachon and life stages are represented. Green shading depicts proposed impact driving period of September 15 through April 15.

**Figure 6. Estimated timing of listed fish species in the CRC project area by basin.**

2. **Projected Project Sequencing**

For each proposed HIWWW, a projected construction sequence for the mainstem bridges was modeled with the various construction elements linked logically in a master schedule. For example, a barge placement for a tower crane would not be sequenced into the end of the HIWWW if there was insufficient time to complete the associated construction of a temporary support structure within the HIWWW.

A total of thirteen different schedules were compiled for each proposed HIWWW to account for variation in potential bridge construction contract award dates. For the purpose of analysis, these award dates occurred approximately 1 month apart between February 5, 2013 and February 1, 2014. If the contracts are awarded earlier or later than the scheduled dates, impact pile driving schedules and impacts would not be likely to change substantially.

Although items were linked in a master schedule, a considerable review of the linking logic occurred by engineering expert opinion for each of the 13 schedules for each HIWWW analysis. Each scheduling and modeling effort required several weeks to complete and involved creating a master project schedule and engineer review prior to input into the hydroacoustic exposure analysis.

3. **Hydroacoustic Exposure Model**

Potential impacts to listed salmonids and eulachon for the mainstem bridge construction were estimated based on magnitude of exposure in relation to the estimated proportion of the ESU/DPS in the project area during impact pile driving. An exposure factor was calculated for each week of the year based on the NMFS moving fish hydroacoustic impact model and the projected construction sequence. The weekly exposure factor and weekly percentage for a given ESU/DPS and life stage were calculated with the following equation for each week of construction:

\[
\text{Weekly Fish Exposure} = \text{Weekly Proportion of ESU/DPS} \times \text{Weekly Exposure Factor}
\]
The same weekly exposure factor for each week of construction was used to calculate proportions of the other ESU/DPSs exposed in the project area. It is important to note that for the lower river ESU/DPS, such as the Columbia River chum ESU, this impacted value includes only the proportion of the ESU that spawns upriver of the project and not the entire ESU.

Exposure factors were calculated for daily, weekly, annual, and project-life periods. For example, to obtain annual exposure factors and annual percentage of ESU/DPS impacted, the model repeats the calculation for each week of construction each year. In weeks with no pile driving, the exposure factor is zero. To find which year of construction has the biggest impact to a listed entity, the calculations are repeated for each construction year using the appropriate weekly exposure factor for each week and each year. To obtain the cumulative percentage value, the percentages of an ESU/DPS potentially impacted each year were added together. By dividing the cumulative percentage by the construction period, one can obtain an average for the construction period.

In addition, because the exact date in-water construction would begin is unknown, this analysis was repeated for 13 separate construction schedules that varied with the potential bridge construction contract award dates. For the purpose of analysis, these award dates occurred approximately 1 month apart between February 5, 2013 and February 1, 2014. If the contracts are awarded earlier or later than the scheduled dates, impact pile driving scenarios and impacts would not be likely to change substantially.

Several modeling assumptions were made in concurrence with representatives from NMFS, ODFW, and WDFW. In these instances, conservative assumptions were used in the absence of site-specific, species-specific data to allow regulatory agencies to err on the side of caution when analyzing impacts.

**Evaluation of Hydroacoustic In-Water Work Windows**

A total of three variations on the HIWWW were modeled using the design and construction changes described previously; a 20-week HIWWW, a 23-week HIWWW, and a 31-week HIWWW. Potential impacts to all ESU/DPSs and life stages for all 13 pile-driving scenarios were calculated. Cumulative impacts for all four construction years, in addition to the mean, minimum, and maximum proportion of an ESU/DPS exposed under any of the 13 construction sequences for each scenario were calculated. The outcome of each modeled HIWWW is summarized below. Exact comparisons of model output are not available, because throughout the modeling process the CRC team continued to refine the project sequencing information with the engineering team as the design progressed, and the fish timing and abundance information as agency feedback and additional data were available.

**Scenario #1: 20-Week HIWWW From November 1 Through February 28**

This scenario used the established 20-week IWWW for in-water construction and applied it to impact pile driving. Almost all potential hydroacoustic exposure to interior basin ESU/DPSs was eliminated with this scenario: however, exposure was concentrated on the lower river ESU/DPSs (LCR Chinook, Columbia River chum, LCR coho, and Columbia River steelhead). The overall length of the main channel bridge construction was estimated to be 54 to 63 months.

**Scenario #2: 23-Week HIWWW From October Through February 15**

This scenario extended the established IWWW by just over 2 weeks. Like scenario #2, almost all potential hydroacoustic exposure to interior basin ESU/DPSs was eliminated with this scenario and exposure was concentrated on the lower river ESU/DPSs. The overall length of the main channel bridge construction was estimated to be 52 to 63 months.

**Scenario #3: 31-Week HIWWW From September 15 Through April 15**

This scenario extended the established in water work window by approximately 13 weeks. The timeline for the main channel bridge construction was estimated to be 48 to 56 months. Although the project would range from 4 to almost 5 years, potential exposure was spread over a longer time period and; therefore, impacts to the lower river ESU/DPSs were below those calculated for Scenario #2.

For all 13-construction sequences modeled in this scenario, the impact on any ESA-listed ESU/DPSs in any given year would be between zero and 0.5 percent, with average impacts below 0.06 percent for all ESU/DPSs per year. A comparison of mean cumulative percent of potential impacts for each ESU/DPS and life stage is provided in Figure 7. Nine of the 14 adult ESU/DPSs in this scenario are estimated to experience less than a mean 0.2 cumulative percent of potential impact and 11 of the 14 juvenile fish ESU/DPSs are estimated to have less than a mean 0.1 cumulative...
percent of potential impact. The exceptions include Columbia River chum, Upper Willamette River Chinook, and LCR coho juveniles, with Columbia River chum being the most impacted.

**TEST PILE STUDY**

A number of assumptions are used in the fish data analyses and exposure model. These assumptions were intentionally biased to provide additional protection to species when data were lacking. To provide additional confidence in the input variables for the noise exposure analysis, a test pile study was proposed early in project design, but due to funding, it was not conducted until after the exposure analysis modeling was conducted. With the help of the InterCEP workgroup, the study was permitted on an accelerated timeline. The study measured pile source noise levels, strike numbers, and noise transmission loss rates. Information from the study will be incorporated into the hydroacoustic exposure model. Preliminary results indicate that assumptions used in the model were valid.

**Figure 7. Mean cumulative percent impact by ESU/DPS and life stage.**

**DISCUSSION**

The InterCEP workgroup was vital to the project successfully avoiding a jeopardy opinion and obtaining construction timing for impact pile driving. By changing the foundation design to drilled shafts and adding a requirement for initial non-impact pile placement, a significant reduction in the amount of impact pile driving, the size of piles, and amount of in-water noise expected during project construction resulted. Direct communication between agency representatives and the engineering and environmental team was essential to achieve this outcome.

The hydroacoustic exposure model proved a valuable tool for analyzing variations in the HIWWW by producing an analysis of impacts for each ESU/DPS and life stage by construction year and project-life period. The final HIWWW decision was based on striking a balance between the length of in-water construction extending over multiple years and...
overall minimization of exposure to interior basin ESU/DPSs. Results demonstrated that shortening the HIWWW to too few weeks, as in the 20- and 23-week periods in Scenarios #1 and #2, resulted in potential hydroacoustic exposure being heavily concentrated on lower river ESUs/DPSs. These two scenarios were also the least favorable from a constructability standpoint. Conversely, lengthening the HIWWW beyond the 31-week period modeled in Scenario #3 would result in a shorter project timeline, but also in greater potential impacts to interior basin ESU/DPSs. Scenario #3, the 31-week HIWWW from September 15 through April 15, was ultimately chosen as a viable alternative for balancing potential impacts to individual ESU/DPSs while maintaining a viable project timeline.

To further ensure that the allowable exposure from impact pile driving will not be exceeded during construction, a performance measure related to maximum daily, weekly, annual, and project-wide exposure factors was developed. The performance measure limits impacts to any single ESU/DPS and life stage of listed salmonids and eulachon to be no more than 0.5 percent in a given year and no more than 1 percent (cumulative) over the entire in-water construction period. This performance measure provides NMFS with a conservative, but quantifiable means to evaluate potential project impacts into the broader viability assessment for each ESU/DPS in the basin. Addition of this measure means that project construction may need to adaptively manage impact pile driving to ensure this benchmark is not exceeded.

The hydroacoustic exposure analysis was also used to quantify potential impacts to the Chinook forage base (listed and unlisted ESU/DPSs) of the Southern Resident killer whale as requested by NMFS. Results indicated the project would have an insignificant effect on the Chinook prey base of killer whales.

As part of the ESA consultation, a biological opinion was issued for this large, multi-year project in less than seven months. NMFS concluded in its biological opinion that impacts were likely to be too low to reduce the abundance or productivity of any affected population and they do not expect the project to exceed a reasonable level of mortality for Columbia River species when added to take in other biological opinions (NMFS 2011). The addition of the performance measure, and associated adaptive management to achieve that measure, provided added assurance that estimates of hydroacoustic impacts will not be exceeded.

The overall time estimated for bridge construction changed appreciably with the requirement for drilled shaft pier foundations, the addition of a vibratory pile installation measure, and a HIWWW. The originally project proposal, driving 96-inch piles year-round with no in-water work window would have resulted in impact pile driving occurring throughout a 2- to 3-year period and an estimated construction timeline of 3.75 years for the mainstem bridges. The final accepted 31-week HIWWW for the modified design with the addition of a non-impact pile installation measure has an estimated construction timeline of 4- to 4.67-years for the mainstem bridges.

The information from the test pile study will allow further refinement of fish exposure estimates. The project team has also committed to monitor current and future research in the Columbia River for best available science regarding modeling assumptions. For example, more refined data on fish travel rates in the lower mainstem from ongoing fish tagging efforts may refine the model further.

Other projects with potentially high, but unknown impacts may benefit from a collaborative pre-permitting process both internally with engineers and environmental staff and externally with resource agency staff. Where data on potential exposure is lacking, impact modeling with conservative assumptions, incorporation of a performance measure, and the use of pre-project studies to reduce uncertainty can improve regulatory approval uncertainties, timelines, and the need for overly conservative assumptions regarding potential impacts.

**CONCLUSIONS**

The members of the InterCEP workgroup were an integral part in the development and agreement of project minimization of hydroacoustic impacts. A high level of cooperation occurred among members to issue permits for a test pile study in an accelerated timeline. Future continued coordination with InterCEP members will occur as various permits advance and is expected to continue throughout project construction.

The project team developed a hydroacoustic exposure model to estimate weekly, yearly, and cumulative estimates of hydroacoustic impacts for each juvenile and adult listed ESU/DPS in the project area. This analysis utilized fish timing information and expert opinion from resource agency staff and extensive coordination with these staff to develop timing curves and weekly abundance estimates for each listed fish ESU or DPS. This unique analytical model was used to analyze HIWWWs based on an assessment of risk to specific species.

Potential project hydroacoustic impacts to listed fish have been reduced through minimization measures related to project design (pier foundations constructed with drilled shafts versus 2.4 meter (8’) diameter piles), project...
construction methods (minimizing the impact pile driving and use of a noise attenuation device), and limits on the
timing of impact pile driving and the duration and extent of potential impacts to any listed ESU or DPS (through use of a
performance measure). Through impacts to listed fish are unavoidable, the final project HIWWW limits impacts to any
single ESU/DPS and life stage of listed salmonids and eulachon to no more than 0.5 percent in a given year and no
more than 1 percent (cumulative) over the entire construction period. Most potential impacts will occur to fish that
originate and return to the lower Columbia River, rather than those ESU/DPSs that migrate into the middle or upper
Columbia River and its tributaries.

Uncertainty, of modeling assumptions is reduced through a test pile study used to validate some input variables.
Confidence in the analysis can be further increased through re-evaluation of model assumptions if additional data
regarding those assumptions becomes available.

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Matt Deml, provided valuable input on engineering and construction sequencing. CRC scheduler, Fred Bullen, patiently
made over 39 iterations of proposed project sequences used in the analysis.

BIOGRAPHICAL SKETCHES

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Cindy Callahan is an Environmental Specialist/Biologist with the Federal Highway Administration Washington Division.
Prior to joining FHWA, she spent 16 years in the private sector as a senior project manager. Cindy has completed or
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Frank Green, PE, is the structures engineering manager for the Columbia River Crossing.

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Steve Morrow is an ODOT Environmental Coordinator with 20 years’ experience in the Pacific Northwest working on
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Michael Parton is an aquatic ecologist and senior consultant with Parametrix. Mike served as the principal scientist in
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conservation projects throughout the NW.

Heather Wills is the Environmental Manager for the Columbia River Crossing Project.
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FHWG (Fish Hydroacoustics Working Group). 2008. Agreement in principal for interim criteria for injury to fish from pile driving activities. Memorandum between FHWA, NOAA Fisheries NW and SW Region, USFWS Regions 1 and 8, Caltrans, and ODOT. June 12, 2008.


**Puget Sounds: A Summary of Recent Underwater and Airborne Noise Measurements from Marine Waters of Washington State**

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**ABSTRACT**

Compliance with the Endangered Species Act of 1973 and the Marine Mammal Protection Act of 1972 are the primary drivers for establishing noise thresholds and wide zones of potential harm and monitoring for marine mammals in Puget Sound, Washington. These zones are established conservatively as there are no site-specific data to better understand the transmission of sound through the air and marine waters. The establishment of these zones has a direct impact on ferry terminal projects as noise-generating work is curtailed when marine mammals are observed within the zone. Underwater and airborne sound level data for both vibratory and impact driving of steel piles were collected at three ferry terminal locations in Puget Sound to address the sound impact thresholds and provide site specific information. Vibratory sound levels ranged between 158 dB and 176 dB RMS, and between 194 dB and 195 dB RMS at the near field locations for impact driving of steel piles of different sizes. Sound levels measured near the pile as well as simultaneous measurements collected approximately ½ a mile or more away are used to calculate the transmission loss over that distance. The source sound levels and transmission loss values are used in three standard spreading loss models (Practical, Spherical, and Cylindrical) to determine which model fits most closely to the measured data. This site specific information was then used to reduce the monitoring area for marine mammals in Puget Sound based on a 120 dB RMS threshold for vibratory driving. It was found that, primarily due to differences in bathymetry, each site corresponded to a different spreading model. Vibratory pile driving results indicate that site specific data can reduce the biological monitoring area by half or more, and inclusion of sound level frequencies above 10 kHz does not alter the results in any meaningful way. Results indicate that a new underwater sound spreading model is needed for each specific project site with the possibility of a new generalized model for Puget Sound. Background sound levels were measured at four ferry terminals in Puget Sound following the NOAA guidance for collection, analysis, and reporting of underwater background sound levels using only those frequencies corresponding to marine mammal functional hearing groups. Results show that underwater sound levels at these terminals range between 100 dB RMS and 130 dB RMS. The differences in these sound levels are likely correlated to the frequency of ferry traffic at the terminals because the sound levels are dominated by ferry traffic. Future research will include possible methods to mitigate noise transmitted through the substrate and then into the water column.

**INTRODUCTION**

Recent advances in the understanding of noise impacts to marine mammals have created underwater sound regulatory thresholds as low as 120 dB and have required Washington State Department of Transportation (WSDOT) projects to monitor very large areas in Puget Sound, Washington (Figure 1), for Killer Whales and stellar sea lions. Underwater sound levels dissipate with distance from the source; however, the current use of the standard practical spreading model (Davidson 2004, Thomsen et al., 2006) to assess the area of effects to marine mammals appears overly conservative with such low regulatory thresholds of 120 dB RMS as described by NMFS (2005).

The installation of steel piles with an impact or vibratory hammer to support ferry docks and bridge piers in nearshore marine and aquatic habitats produces high sound levels in air and underwater. The WSDOT has been concerned since they became aware of the effects of pile driving on fish since 2002. Concerns regarding the potential effects of pile driving activities on marine and aquatic animals have grown since WSDOT first became aware of potential harm to fish from noise. Sounds from pile driving may cause effects to marine and aquatic animals resulting in barbotrauma, temporary or permanent loss of hearing, and/or altered behavior leading to greater risk to predators or reduced feeding (Schwarz 1985, Pearson et al. 1992, McCauley et al. 2003, Smith et al. 2004, Popper et al. 2005). Several marine and anadromous fish in Puget Sound are protected under the Endangered Species Act (Stadler 2003, NMFS 2007) providing federal regulatory agencies a means to control noise-generating projects.

In this paper, vibratory and impact pile driving results will be presented from the Vashon Ferry Terminal test pile project and vibratory driving results from Keystone and Port Townsend Ferry terminals in Puget Sound, Washington. Results of underwater noise level monitoring will be presented and compared to existing spreading models. Unweighted airborne noise levels from the Keystone Ferry Terminal will be presented and compared to the current airborne thresholds for pinnipeds. Additionally, results from underwater background sound level measurements will be presented from the Port Townsend, Anacortes, Edmonds and Seattle Ferry Terminals.
CHARACTERISTICS OF UNDERWATER SOUND

Several descriptors are used to describe underwater noise. One common descriptor used in this paper is the Root Mean Square (RMS) pressure level; sometimes referred to as the RMS level. The RMS level is the square root of the energy divided by the impulse duration. This level, presented in dB re: 1 microPascal (μPa), is the mean square pressure level of the signal. It has been used by National Oceanographic and Atmospheric Administration (NOAA) in criteria for determining impacts to marine mammals from airborne and underwater anthropogenic sources. Underwater sound levels reported in this report are expressed in dB re: 1 μPa. Airborne sound levels are reported in dB re: 20 μPa. The equation to calculate the RMS sound level is:

$$f_{rms} = \sqrt{\frac{1}{T_2 - T_1} \int_{T_1}^{T_2} [f(t)]^2 \, dt},$$

Where $T = T_2 - T_1$ is the total time interval from start ($T_1$) to finish $T_2$, and $t = t_1$ are the periods within the larger interval. A second impacts descriptor for fish is the Cumulative Sound Exposure Level (SELCum). The SELcum is calculated by first determining the 1-second SEL for a single impact pile strike, which is typically the highest SEL of the entire drive for one pile. The single strike SEL is then plugged into the following formula to calculate the SELcum.

$$SELCum = \text{Single Strike SEL} + 10 \times \text{LOG(total number of strikes)}$$

The calculation of the SELcum assumes that all pile strikes start with the highest single strike SEL to provide a conservative estimate of the SELcum.
MATERIALS AND METHODS

Underwater sound levels were measured at mid-water level near the pile using a Reson TC 4013 hydrophone deployed on a weighted nylon cord. The distance between 6 to 16 meters from the individual pile being monitored is referred to as the near field measurements. The measurement system includes a Brüel and Kjær Nexus type 2692 4-channel signal conditioner, which kept the high underwater sound levels within the dynamic range of the signal analyzer. The output of the Nexus signal conditioner is received by a Dactron Photon 4-channel signal spectrum analyzer that is attached to an Itronix GoBook II laptop computer. The operation of the near field hydrophones were checked daily in the field using a GRAS type 42AC high-level pistonphone with a hydrophone adaptor. The signal levels produced by the pistonphone system were within 1 dB and the operation of the system was judged acceptable over the study period. Signal analysis software provided with the Photon was set at a sampling rate of one sample every 41.7 μs (9500 Hz). No frequency weighting (e.g., A-weighting or C-weighting) was applied to the underwater acoustic measurements presented in this report. Near field hydrophones were located where there was a clear line of sight between the pile and the hydrophone, with no other structures within 3 meters of the hydrophone. The distance from the pile to the hydrophone location was measured using a Bushnell Yardage Pro rangefinder.

In addition to the near field measurements, far field measurements were also collected at distances of approximately 800 meters from the piles using a Jasco Research Ltd. Autonomous Multi-Channel Acoustic Recorder (AMAR mini) anchored and retrieved using an acoustical release system. The AMAR is used to collect background sound levels to calculate a site specific transmission loss value and the distance to the NOAA underwater threshold of 120 dB RMS. Each AMAR consists of one GeoSpectrum M15 hydrophone with a filter/amplifier board, 0.1 to 20 kHz bandwidth, and a sensitivity of -160 dB re 1 μPa. The AMAR has one channel of 16-bit, 1 MS/s, solid state storage with 128 GB base, Waveform Audio File (wav) formatted recordings. The location of the far field hydrophone was determined by deploying the AMAR approximately 800 meters from the piles and allowing a clear line of sight between the pile and the hydrophone and out of the direct path of ferry vessel traffic. The distance from the pile to the hydrophone location was measured using GPS coordinates and GIS. For underwater background measurements, the AMAR was deployed approximately 800 meters from the ferry terminal and a nylon sock was placed over the metal cage surrounding the hydrophone to protect it from flow noise caused by tidal currents greater than one meter per second.

Noise attenuation methods were not used as part of these vibratory measurements. Broadband Root Mean Square (RMS) noise levels are reported in terms of the 30-second averages and have been computed from the Fourier transform of pressure waveforms in 30-second time intervals.

Pile Driving

Vashon
Use of a vibratory hammer produces continuous sounds for extended periods that may disturb whales and seals when they exceed a criterion level of 120 dB RMS (NMFS, 2005). Four 30-inch diameter steel piles were monitored at the Vashon Ferry Terminal as they were driven with a vibratory hammer, and then with an impact hammer in approximately 11 meters of water. Near field measurements were collected within 11 to 16 meters of the piles. The AMAR was deployed at two separate locations to collect vibratory sound levels at distances of 790 meters for Deployment Site 1 for piles 1 and 2, and 805 meters for Deployment Site 2 for piles 3 and 4 (Figure 2).

Port Townsend
The piles were driven near a floating dolphin on the northeast side of the ferry terminal with a vibratory hammer. When fully driven, the 30-inch pile was in 9.0 meters of water; and the 36-inch pile was in 9.5 meters of water. Both piles were driven in sequence, approximately 3.0 meters apart, 12 meters into the substrate. Water depth differences are the result of tidal changes.

Three AMAR were used during the Port Townsend test pile project. The near field hydrophone was located at 10 meters from the piles and two of the three AMARs were in fixed far field locations at 3200 and 6400 meters, respectively. A small boat was used to deploy the near field hydrophone, and a tug to deploy the AMARs. The third AMAR was moved between 100, 200, 400, 800 and 1600 meters (Table 1, Figure 3).
Figure 2: Location of near field and AMAR vibratory monitoring locations at the Vashon Ferry Terminal.

Table 1: Hydrophone distances and depths

<table>
<thead>
<tr>
<th>Hydrophone Distance from Piles (meters)</th>
<th>Depth (meters)</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>8.2</td>
</tr>
<tr>
<td>100</td>
<td>8.2</td>
</tr>
<tr>
<td>200</td>
<td>12.8</td>
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<tr>
<td>400</td>
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<tr>
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<tr>
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</tr>
<tr>
<td>3200</td>
<td>10.0</td>
</tr>
<tr>
<td>6400</td>
<td>54.9</td>
</tr>
</tbody>
</table>

Six separate pile driving measurements are associated with the 30-inch pile and five pile driving measurements associated with the 36-inch pile. Each measurement was collected at different ranges, with some overlap. In each measurement set there was a hydrophone collecting data at 10 meters from the pile and two AMARs located nominally at 3200 meters and 6400 meters that were taking data during all measurements. Due to some malfunctioning of the third AMAR, the intermediate measurement is not available for every set. Also, due to a very low signal to noise ratio at 6400 meters, no measurements from that AMAR site are used in this paper.

Due to the widely varying source level over the duration of the pile driving events, a different analysis method was used. A 30-second averaging window was used to find the maximum 30-second RMS pressure level during the vibratory pile driving. This was determined to be a better way to quantify the sound levels and the transmission loss than an overall average RMS measurement. The data used in this analysis was also filtered through a high pass filter which results in only the frequencies between 1 kHz and 10 kHz, following guidance from NOAA (2009) on how to characterize background and vibratory sound levels in Puget Sound. The transmission loss is given as $20 \log_{10}$ of the maximum 30-second RMS value at the near field hydrophone minus $20 \log_{10}$ of the maximum 30-second RMS value at the far field hydrophones.
Figure 3: Approximate location of the AMAR deployment sites along a transect moving away from the pile for the Port Townsend Test Pile Project.

Keystone
Near field measurements were taken six meters from the pile in 9.0 meters of water. Far field measurements were taken 546 meters from the pile in 29 meters of water (Figure 4, Location #2). The far field location was just south of the strong current area just outside of the harbor mouth.

Background Sound Levels

Background sound levels were measured near the Port Townsend, Anacortes, Edmonds and Seattle Ferry Terminals in an effort to determine site specific underwater background sound levels to assist in the determination of the zone of influence for marine mammals. Seven days of data were collected at each site. Three full 24-hour cycles or 72-hours (e.g., 6AM to 6AM) were analyzed as part of this analysis. The data was analyzed using a high pass filter at 1000 Hz and 150 Hz for Port Townsend, and 75 Hz and 150 Hz for Anacortes, Edmonds, and Seattle, in addition to the full frequency bands between 20 Hz and 20 kHz to approximate the functional hearing group frequency ranges for pinnipeds and Killer Whales (Southall, et al., 2007). A 30-second RMS value was calculated for each 30-second period for the entire 72-hour duration. These 30-second RMS values were plotted on a Cumulative Distribution Function (CDF) plot (NOAA, 2009) and the 50th percentile value was used to approximate the average background sound level. In addition a sliding bootstrap analysis was conducted on the Port Townsend data to determine a statistically significant sample size for collection of underwater background data.
RESULTS AND DISCUSSION

Pile Driving

Vashon

Average RMS values ranged from 160 to 169 dB RMS at the near field location with an overall average RMS value of 164 dB RMS (Table 2), with most RMS values around 160 dB RMS in the near field.

Table 2: Summary of Underwater Vibratory Monitoring Results at the Near Field and Far Field Locations.

<table>
<thead>
<tr>
<th>Pile #</th>
<th>Near Field Distance To Pile (meters)</th>
<th>Near Field Absolute Peak (dB)</th>
<th>Near Field Average RMS Value (dB)</th>
<th>Far Field Distance To Pile (meters)</th>
<th>Far Field Absolute Peak (dB)</th>
<th>Far Field Average RMS Value (dB)</th>
<th>Transmission Loss1</th>
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</thead>
<tbody>
<tr>
<td>1</td>
<td>11</td>
<td>180</td>
<td>169</td>
<td>790</td>
<td>168</td>
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<td>16</td>
<td>187</td>
<td>160</td>
<td>806</td>
<td>155</td>
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</tr>
<tr>
<td>Average</td>
<td>187</td>
<td>164</td>
<td>164</td>
<td>129</td>
<td>34</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

1 - Transmission loss is a complicated function of local bathymetry, sound-speed profile, range, source frequency, absorption, and scattering (Medwin and Clay, 1998). However, if it is possible to measure both the source and received sound pressure levels, the equation $TL_{DB} = SL_{DB} - RL_{DB}$ may be used to calculate the transmission loss (Carr et al., 2006), where $SL_{DB}$ is the measured source level and $RL_{DB}$ is the measured received level.
The far field RMS measurements collected using the AMAR ranged from 126 to 131 dB RMS with an overall average of 129 dB RMS, most likely due to differences in transmission loss over that distance. Calculating the simple transmission loss (TL), or the reduction in sound levels over distance, is accomplished by subtracting the sound level measured at approximately 800 meters (0.5 miles) from the sound level near each pile measured. These TL values range between 29 and 43 dB with an overall average TL of 34 dB (Table 2).

NOAA NMFS and the U.S. Fish and Wildlife use the conservative practical spreading model to determine the distance to the threshold boundary for marine mammals and fish. Other spreading models such as the spherical and cylindrical spreading models are used under special circumstances. The formulae for the three spreading models are:

Practical Spreading Model: \( R_1 = R_0 \times 10^{(TL/15)} \)
Spherical Spreading Model: \( R_1 = R_0 \times 10^{(TL/20)} \)
Cylindrical Spreading Model: \( R_1 = R_0 \times 10^{(TL/10)} \)

Comparing the highest measured AMAR sound levels at 806 meters against the calculated distance to the measured far field sound level using all three spreading models (practical, spherical and cylindrical) it appears, that on average, the spherical model is more accurate at modeling the actual distance to the measured RMS level for each pile (Table 3).

### Table 3: Comparison of Different Spreading Models Using Actual Vashon Vibratory Measured Data.

<table>
<thead>
<tr>
<th>Spreading Model</th>
<th>Distance From Pile (meters)</th>
<th>Pile #</th>
<th>Transmission Loss(^1)</th>
<th>Meters To Measured dB RMS</th>
<th>Miles To Measured dB RMS</th>
<th>Measured Distance at 131 dB RMS (miles)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Practical</td>
<td>11</td>
<td>1</td>
<td>43</td>
<td>8092</td>
<td>5.0</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>30</td>
<td></td>
<td>1100</td>
<td>0.7</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>33</td>
<td></td>
<td>1743</td>
<td>1.1</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>29</td>
<td></td>
<td>944</td>
<td>0.6</td>
<td>0.5</td>
</tr>
<tr>
<td>Average</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1.9</td>
</tr>
<tr>
<td>Spherical</td>
<td>11</td>
<td>1</td>
<td>43</td>
<td>1554</td>
<td>1.0</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>30</td>
<td></td>
<td>348</td>
<td>0.2</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>33</td>
<td></td>
<td>491</td>
<td>0.3</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>29</td>
<td></td>
<td>310</td>
<td>0.2</td>
<td>0.5</td>
</tr>
<tr>
<td>Average</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.4</td>
</tr>
<tr>
<td>Cylindrical</td>
<td>11</td>
<td>1</td>
<td>43</td>
<td>219479</td>
<td>136</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>2</td>
<td>30</td>
<td></td>
<td>11000</td>
<td>6.8</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>3</td>
<td>33</td>
<td></td>
<td>21948</td>
<td>13.6</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>4</td>
<td>29</td>
<td></td>
<td>8738</td>
<td>5.4</td>
<td>0.5</td>
</tr>
<tr>
<td>Average</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>40.4</td>
</tr>
</tbody>
</table>

\(^1\) \( TL = SL_{\text{dB}} - RL_{\text{dB}} \), where \( SL_{\text{dB}} \) is the measured source level and \( RL_{\text{dB}} \) is the measured received level

There is additional support for the use of the Spherical Model. Carr et al., (2006) found that at the Cacouna LNG terminal in Haro Straight, British Columbia, the sound levels from a vibratory hammer drop below 120 dB for ranges greater than 1.6 Km. These results are consistent with the data we collected for the Vashon Ferry Terminal.

In addition to the vibratory measurements, impact driving of steel piles was measured as part of the Vashon test pile project. Table 4 summarizes the results of the impact driving of steel piles at Vashon.

Average peak values near the pile were fairly consistent ranging from 215 to 217 dBpeak while the RMS values ranged from 194 to 195 dB RMS near the pile with an overall average RMS value of 195 dB RMS. The results of Table 4 show that RMS values were within 1 dB for all piles measured.
In addition to the near field noise measurements, far field measurements were taken at a location of 754 meters from the piles using the AMAR for all piles driven with an impact hammer (Figure 5). The impact pile driving sounds at the far field location were still well above background noise levels (Table 4), but 30 to 35 dB lower than the near field location. The peak levels ranged between 168 and 174 dB peak while the RMS levels ranged between 159 dB RMS and 165 dB RMS with an overall average of 163 dB RMS. Transmission loss ranged between 30 and 35 dB with an overall average of 32 dB.

Table 4: Summary of Unmitigated Underwater Impact Sound Levels for the Vashon Test Pile Project, Steel Piles.

<table>
<thead>
<tr>
<th>Pile #</th>
<th>Near Field Distance To Pile (meters)</th>
<th>Near Field Absolute Peak (dB)</th>
<th>Near Field Average RMS Value (dB)</th>
<th>Far Field Distance To Pile (meters)</th>
<th>Far Field Absolute Peak (dB)</th>
<th>Far Field Average RMS Value (dB)</th>
<th>Transmission Loss (dB)</th>
</tr>
</thead>
<tbody>
<tr>
<td>P-14</td>
<td>7</td>
<td>215</td>
<td>194</td>
<td>754</td>
<td>168</td>
<td>159</td>
<td>35</td>
</tr>
<tr>
<td>P-16</td>
<td>15</td>
<td>217</td>
<td>195</td>
<td>754</td>
<td>174</td>
<td>165</td>
<td>30</td>
</tr>
<tr>
<td>P-8</td>
<td>16</td>
<td>215</td>
<td>195</td>
<td>754</td>
<td>174</td>
<td>165</td>
<td>30</td>
</tr>
<tr>
<td>Overall Average</td>
<td>216</td>
<td>195</td>
<td>172</td>
<td>163</td>
<td>32</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

1 - Transmission loss is a complicated function of local bathymetry, sound-speed profile, range, source frequency, absorption, and scattering (Medwin and Clay, 1998). However, if it is possible to measure both the source and received sound pressure levels, the equation $TL_{dB} = SL_{dB} - RL_{dB}$ may be used to calculate the transmission loss (Carr et al., 2006), where $SL_{dB}$ is the measured source level and $RL_{dB}$ is the measured received level.

Figure 5: Location of the AMAR deployment for the Vashon test pile project impact driving of steel piles.
Based on our measurements, the practical spreading model appears overly conservative since it predicts that the measured sound level would occur almost 7998 meters further out. Comparing the measured AMAR results at 754 meters using all three spreading models (practical, spherical and cylindrical) it appears that, on average, the spherical model is more accurate at modeling the actual distance of the measured RMS level for each pile (Table 5).

<table>
<thead>
<tr>
<th>Spreading Model</th>
<th>Distance From Pile (meters)</th>
<th>Pile #</th>
<th>Transmission Loss 1</th>
<th>Calculated Meters To Measured RMS</th>
<th>Calculated Miles To Measured RMS</th>
<th>Measured Distance at Received RMS (miles)</th>
<th>Distance to measured RMS Level (Miles)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Practical</td>
<td>7</td>
<td>P-14</td>
<td>35</td>
<td>1508</td>
<td>0.94</td>
<td>0.46</td>
<td>2.6</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>P-16</td>
<td>30</td>
<td>1500</td>
<td>0.93</td>
<td>0.46</td>
<td>6.6</td>
</tr>
<tr>
<td></td>
<td>16</td>
<td>P-8</td>
<td>30</td>
<td>1600</td>
<td>0.99</td>
<td>0.46</td>
<td>7.0</td>
</tr>
<tr>
<td></td>
<td><strong>Average</strong></td>
<td></td>
<td></td>
<td><strong>0.95</strong></td>
<td></td>
<td><strong>5.4</strong></td>
<td></td>
</tr>
<tr>
<td>Spherical</td>
<td>7</td>
<td>P-14</td>
<td>35</td>
<td>394</td>
<td>0.24</td>
<td>0.46</td>
<td>0.22</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>P-16</td>
<td>30</td>
<td>474</td>
<td>0.29</td>
<td>0.46</td>
<td>0.52</td>
</tr>
<tr>
<td></td>
<td>16</td>
<td>P-8</td>
<td>30</td>
<td>506</td>
<td>0.31</td>
<td>0.46</td>
<td>0.56</td>
</tr>
<tr>
<td></td>
<td><strong>Average</strong></td>
<td></td>
<td></td>
<td><strong>0.28</strong></td>
<td></td>
<td><strong>0.43</strong></td>
<td></td>
</tr>
<tr>
<td>Cylindrical</td>
<td>7</td>
<td>P-14</td>
<td>35</td>
<td>22,136</td>
<td>13.75</td>
<td>0.46</td>
<td>10.9</td>
</tr>
<tr>
<td></td>
<td>15</td>
<td>P-16</td>
<td>30</td>
<td>15,000</td>
<td>9.32</td>
<td>0.46</td>
<td>29.5</td>
</tr>
<tr>
<td></td>
<td>16</td>
<td>P-8</td>
<td>30</td>
<td>16,000</td>
<td>9.94</td>
<td>0.46</td>
<td>31.4</td>
</tr>
<tr>
<td></td>
<td><strong>Average</strong></td>
<td></td>
<td></td>
<td><strong>11.0</strong></td>
<td></td>
<td><strong>23.9</strong></td>
<td></td>
</tr>
</tbody>
</table>

1 - TL\text{dB} = SL\text{dB} - RL\text{dB}; where SL\text{dB} is the measured source level and RL\text{dB} is the measured received level. The highest transmission loss for each pile is used here to represent the most conservative scenario.

**Port Townsend**

The calculated transmission loss based on measured data is graphically compared to the three geometrical spreading models in Figure 5.

Acoustic signals are nominally expected to undergo spherical spreading in shallow water at ranges on the order of a few near-range depth scales (Figure 5). As the range increases, this loss is expected to be more closely approximated by cylindrical spreading. Cylindrical and spherical spreading should represent the lower and upper bounds (respectively), on the transmission loss. The transmission loss at Port Townsend is approximated reasonably well by the practical spreading model for both the 30-inch and 36-inch piles. Variability of the data is likely due to differences in transmission loss over these longer distances (Figure 6).

The maximum RMS levels are graphed (Figure 7) as a function of range for multiple tests on the 30-inch and 36-inch piles in the band from 1 to 10 kHz with lines representing practical spreading from each source level (Dahl et al., 2010). These data indicate a wide variety of RMS levels at range of 10 meters, likely due to differences in substrate composition and hammer energy supplied to the pile driver. This suggests that if the spreading continues to be accurately modeled by the practical spreading model, the RMS values will reach the nominal background threshold of 120 dB RMS between three and 10 kilometers, with a central tendency value at approximately 6800 meters for both the 30-inch and 36-inch piles.
Figure 6: The measured transmission loss to each AMAR hydrophone location for each test on the 30-inch and 36-inch piles.

Figure 7: The maximum 1 – 10 kHz band 30-second RMS levels for multiple tests on the 30-inch and 36-inch piles. The dashed lines represent practical spreading based on the level at 10 meters for each test. The arrow is the central tendency range where the level of vibratory pile driving is expected to reach the nominal background.
Keystone

Average RMS values ranged from 164 to 176 dB RMS at the near field location with an overall average RMS value of 171 dB RMS. Distances from hydrophone to pile ranged between six and 11 meters (Table 6).

Table 6: Summary Table of Underwater Monitoring Results at the Near Field Location for Keystone Ferry Terminal.

<table>
<thead>
<tr>
<th>Pile #</th>
<th>Date</th>
<th>Hydrophone Depth (meters)</th>
<th>Distance To Pile (meters)</th>
<th>Absolute Peak (dB)</th>
<th>Average RMS Value (dB)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1/9/10</td>
<td>4.6 (midwater)</td>
<td>10</td>
<td>195</td>
<td>164</td>
</tr>
<tr>
<td></td>
<td></td>
<td>8.2 (bottom)</td>
<td>10</td>
<td>195</td>
<td>165</td>
</tr>
<tr>
<td>2</td>
<td>1/17/10</td>
<td>8.8 (bottom)</td>
<td>11</td>
<td>195</td>
<td>176</td>
</tr>
<tr>
<td>3</td>
<td>2/8/10</td>
<td>4.6 (bottom)</td>
<td>6</td>
<td>200</td>
<td>176</td>
</tr>
<tr>
<td>4</td>
<td>2/8/10</td>
<td>4.6 (midwater)</td>
<td>6</td>
<td>176</td>
<td>165</td>
</tr>
<tr>
<td>Average</td>
<td>196</td>
<td>171</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

In addition to the near shore noise measurements, analysts measured far field sound levels at 546 meters (Figure 4, Deployment Site 2) using an AMAR. The vibratory pile driving sounds at the far field location were 7.0 to 20 dB lower (Transmission Loss) than the near field location (Table 7), with an overall average of 13.5 dB. The peak levels were consistently 168 dB while the RMS levels ranged between 156 dB RMS and 158 dB RMS with an overall average of 157 dB RMS.

Table 7: Summary table of underwater AMAR monitoring results at the far field locations.

<table>
<thead>
<tr>
<th>Pile #</th>
<th>Hydrophone Depth(^1) (meters)</th>
<th>Date</th>
<th>Distance To Pile (meters)</th>
<th>Absolute Peak (dB)</th>
<th>Average RMS Value (dB)</th>
<th>Transmission Loss(^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>3(^3)</td>
<td>25.9</td>
<td>2/8/10</td>
<td>546</td>
<td>168</td>
<td>156</td>
<td>20</td>
</tr>
<tr>
<td>4(^3)</td>
<td>25.9</td>
<td>2/8/10</td>
<td>546</td>
<td>168</td>
<td>158</td>
<td>7</td>
</tr>
<tr>
<td>Average</td>
<td></td>
<td></td>
<td></td>
<td>168</td>
<td>157</td>
<td>13.5</td>
</tr>
</tbody>
</table>

1 - Depth represents depth as measured from the surface. In all locations the hydrophone was deployed approximately four meters above the bottom.

2 - Transmission loss (TL) is a complicated function of local bathymetry, sound-speed profile, range, source frequency, absorption, and scattering (Medwin and Clay, 1998). However, if it is possible to measure both the source and received sound pressure levels, the equation below may be used to calculate the transmission loss (Carr et al., 2006).

3 - A larger vibratory hammer was used for this pile than for Pile 1.

Note: TL\(_{db}\) = SL\(_{db}\) - RL\(_{db}\), where SL\(_{db}\) is the measured source level and RL\(_{db}\) is the measured received level.

The relatively low transmission loss at this site is primarily due to the very flat bottom and consistent depths throughout Keystone harbor. These conditions create a wave-guide which directs the sound energy out of the harbor with relatively small changes.
Based on measurements at the Keystone terminal (Table 7), we used the transmission loss values to calculate the distance in meters to the measured sound level (dB RMS) using the practical, cylindrical and spherical spreading models.

The practical spreading model appears to under-predict the actual measured values the least amount since it predicts that the measured sound level would occur at 0.08 km instead of 0.55 km (Table 8).

### Table 8: Comparison of Different Spreading Models at Keystone Using Measured Data.

<table>
<thead>
<tr>
<th>Spreading Model</th>
<th>Distance From Pile (meters)</th>
<th>Pile #</th>
<th>Transmission Loss</th>
<th>Calculated Meters To Measured dB RMS</th>
<th>Calculated Kilometers To Measured dB RMS</th>
<th>Measured Distance at Measured dB RMS (kilometers)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Practical</td>
<td>6.0</td>
<td>3</td>
<td>20</td>
<td>129</td>
<td>0.13</td>
<td>0.55</td>
</tr>
<tr>
<td></td>
<td>6.0</td>
<td>4</td>
<td>7</td>
<td>30</td>
<td>0.02</td>
<td>0.55</td>
</tr>
<tr>
<td>Average</td>
<td>0.08</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.55</td>
</tr>
<tr>
<td>Spherical</td>
<td>6.0</td>
<td>3</td>
<td>20</td>
<td>60</td>
<td>0.06</td>
<td>0.55</td>
</tr>
<tr>
<td></td>
<td>6.0</td>
<td>4</td>
<td>7</td>
<td>13</td>
<td>0.02</td>
<td>0.55</td>
</tr>
<tr>
<td>Average</td>
<td>0.04</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.55</td>
</tr>
<tr>
<td>Cylindrical</td>
<td>6.0</td>
<td>3</td>
<td>20</td>
<td>600</td>
<td>0.60</td>
<td>0.55</td>
</tr>
<tr>
<td></td>
<td>6.0</td>
<td>4</td>
<td>7</td>
<td>30</td>
<td>0.03</td>
<td>0.55</td>
</tr>
<tr>
<td>Average</td>
<td>0.32</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.55</td>
</tr>
</tbody>
</table>

1. $TL_{dB} = SL_{dB} - RL_{dB}$ where $SL_{dB}$ is the measured source level and $RL_{dB}$ is the measured received level

### Background Sound Levels

**Port Townsend**

Background underwater sound levels were collected over seven days representing 20160 data points. The data was then post-processed by applying a high pass filter at 1 kHz to follow guidelines set forth by NOAA (2009) for analysis of underwater background sound levels. To determine if the data was approximately log-normally distributed (Guassian) the data was plotted as a Probability Density Function (PDF) plot and compared to the nominal Gaussian distribution function (Figure 8). Once it was determine that the data was approximately log-normally distributed, it was plotted as a Cumulative Distribution Function (CDF) following the NOAA (2009) guidelines. The 50% CDF was determined to represent the average background sound levels.

The second goal of analyzing the background data from Port Townsend was to determine how large a sample size would need to be collected to get a statistically significant result. A sliding bootstrap analysis was conducted on the seven days of data and it was determined that after three days of data the increase in confidence becomes negligible, therefore, three 24-hour cycles of data in the presence of ferry traffic is the minimum sample size needed for background data collection.

The data was initially analyzed with a high pass filter at 1 kHz (Dahl et al, 2010) and then later with a high pass filter at 150 Hz and the full band width between 20 Hz and 20 kHz (Dahl, pers. Comm., 2011) (Table 9). The 50% CDF ranges between 100 dB RMS and 104 dB RMS, which is well below the 120 dB threshold for marine mammals during vibratory driving.
Figure 8: Probability Density Function Plot for Port Townsend Background Data Collected for Seven Days.

Table 9: Summary of Port Townsend Background Sound Levels (Dahl, pers. Comm. 2011).

<table>
<thead>
<tr>
<th>Frequency Range</th>
<th>Functional Hearing Group¹</th>
<th>50% Cumulative Distribution Function (dB)</th>
</tr>
</thead>
<tbody>
<tr>
<td>150 Hz to 20 KHz</td>
<td>Mid Frequency Cetaceans</td>
<td>103</td>
</tr>
<tr>
<td>1000 Hz to 20 KHz</td>
<td>NOAA Guidance (2009)</td>
<td>100</td>
</tr>
<tr>
<td>20 Hz to 20 KHz</td>
<td>N/A - Broadband</td>
<td>104</td>
</tr>
</tbody>
</table>

¹ – Southall et al., 2007 describes the functional hearing groups for marine mammals and the frequency ranges of each group.

Anacortes
Background underwater sound levels were collected near the Anacortes ferry terminal over a seven day period. The data was then post-processed by applying a high pass filter at 75 Hz and 150 Hz to represent the functional hearing groups of pinnipeds and Killer Whales, respectively (Southall et al, 2007). The data was determined to be approximately log-normally distributed. The 50% CDF ranges between 119 dB RMS and 130 dB RMS which is very near or somewhat higher than the 120 dB RMS threshold (Table 10).

Table 10: Summary of Underwater Background Noise Levels for the Anacortes Ferry Terminal.

<table>
<thead>
<tr>
<th>Frequency Range</th>
<th>Functional Hearing Group¹</th>
<th>50% Cumulative Distribution Function (dB)</th>
</tr>
</thead>
<tbody>
<tr>
<td>75 Hz to 20 KHz</td>
<td>Pinnipeds</td>
<td>124</td>
</tr>
<tr>
<td>150 Hz to 20 KHz</td>
<td>Mid Frequency Cetaceans</td>
<td>119</td>
</tr>
<tr>
<td>20 Hz to 20 KHz</td>
<td>N/A - Broadband</td>
<td>130</td>
</tr>
</tbody>
</table>

¹ – Southall et al., 2007 describes the functional hearing groups for marine mammals and the frequency ranges of each group.
Edmonds

Background underwater sound levels were collected near the Edmonds ferry terminal over a seven day period. The data was then post processed by applying a high pass filter at 75Hz and 150 Hz to represent the functional hearing groups of pinnipeds and Killer Whales, respectively (Southall et al., 2007). The data was determined to be approximately log-normally distributed. The 50% CDF ranges between 118 dB RMS and 123 dB RMS which is very near or slightly higher than the 120 dB RMS threshold (Table 11).

<table>
<thead>
<tr>
<th>Frequency Range</th>
<th>Functional Hearing Group(^1)</th>
<th>50% Cumulative Distribution Function (dB)</th>
</tr>
</thead>
<tbody>
<tr>
<td>75 Hz to 20 KHz</td>
<td>Pinnipeds</td>
<td>121</td>
</tr>
<tr>
<td>150 Hz to 20 KHz</td>
<td>Mid Frequency Cetaceans</td>
<td>118</td>
</tr>
<tr>
<td>20 Hz to 20 KHz</td>
<td>N/A - Broadband</td>
<td>123</td>
</tr>
</tbody>
</table>

\(^1\) - Southall et al., 2007 describes the functional hearing groups for marine mammals and the frequency ranges of each group.

Seattle

Underwater background sound levels were collected near the Seattle ferry terminal over a seven day period. The data was then post processed by applying a high pass filter at 75Hz and 150 Hz to represent the functional hearing groups of pinnipeds and Killer Whales, respectively (Southall et al., 2007). The data was determined to be approximately log-normally distributed. The 50% CDF ranges between 123 dB RMS and 128 dB RMS which slightly higher than the 120 dB RMS threshold (Table 12).

<table>
<thead>
<tr>
<th>Frequency Range</th>
<th>Functional Hearing Group(^1)</th>
<th>50% Cumulative Distribution Function (dB)</th>
</tr>
</thead>
<tbody>
<tr>
<td>75 Hz to 20 KHz</td>
<td>Pinnipeds</td>
<td>126</td>
</tr>
<tr>
<td>150 Hz to 20 KHz</td>
<td>Mid Frequency Cetaceans</td>
<td>123</td>
</tr>
<tr>
<td>20 Hz to 20 KHz</td>
<td>N/A - Broadband</td>
<td>128</td>
</tr>
</tbody>
</table>

\(^1\) - Southall et al., 2007 describes the functional hearing groups for marine mammals and the frequency ranges of each group.

CONCLUSIONS

Results from the Vashon and Port Townsend test pile projects and the Keystone vibratory pile installation projects was used to calculate a site specific transmission loss and then compared against the commonly used spreading models (Spherical, Practical, and Cylindrical) to see which is the best predictor for estimating the measured sound level at distance. It was determined that the conditions at each site resulted in a different spreading model for each. The spherical model worked best at Vashon, the practical model at Port Townsend, and the Cylindrical model at Keystone.

The underwater background noise levels ranged between 100 dB and 130 dB RMS based on the methodologies outlined in NOAA (2009) guidelines. These background levels can be useful to reduce the biological monitoring area for the species of concern. The background sound levels are dependent upon the frequency of ferry traffic at each location so that the higher the ferry traffic the higher the average sound levels. A minimum of three 24-hour cycles of data must be collected to get a statistically significant sample size.
BIOGRAPHICAL SKETCH

Jim Laughlin is the technical manager for the Air Quality, Acoustics and Energy section for the Washington State Department of Transportation. He oversees a wide range of research and projects related to airborne and underwater noise, provides technical expertise in the field of acoustics to projects throughout the state and is chairing a committee under the National Cooperative Research Program (NCHRP).

REFERENCES


