Marine Pollution Bulletin 64 (2012) 1320–1329

Contents lists available at SciVerse ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Assessing sound exposure from shipping in coastal waters using a single hydrophone and Automatic Identification System (AIS) data

Nathan D. Merchant [a,*], Matthew J. Witt [b], Philippe Blondel [a], Brendan J. Godley [c], George H. Smith [d]

[a] Department of Physics, University of Bath, Bath BA2 7AY, UK
[b] Environment and Sustainability Institute, College of Life and Environmental Sciences, University of Exeter, Penryn, Cornwall TR10 9EZ, UK
[c] Centre for Ecology and Conservation, College of Life and Environmental Sciences, University of Exeter, Penryn, Cornwall TR10 9EZ, UK
[d] Renewable Energy Research Group, College of Engineering, Mathematics and Physical Sciences, University of Exeter, Penryn, Cornwall TR10 9EZ, UK

ARTICLE INFO

Keywords:
Ship noise
Noise assessment
Coastal management
Continuous monitoring
Acoustic impact
Marine protected area

ABSTRACT

Underwater noise from shipping is a growing presence throughout the world's oceans, and may be subjecting marine fauna to chronic noise exposure with potentially severe long-term consequences. The coincidence of dense shipping activity and sensitive marine ecosystems in coastal environments is of particular concern, and noise assessment methodologies which describe the high temporal variability of sound exposure in these areas are needed. We present a method of characterising sound exposure from shipping using continuous passive acoustic monitoring combined with Automatic Identification System (AIS) shipping data. The method is applied to data recorded in Falmouth Bay, UK. Absolute and relative levels of intermittent ship noise contributions to the 24-h sound exposure level are determined using an adaptive threshold, and the spatial distribution of potential ship sources is then analysed using AIS data. This technique can be used to prioritise shipping noise mitigation strategies in coastal marine environments.

1. Introduction

Anthropogenic underwater noise can have deleterious effects on a variety of marine organisms, including mammals (Richardson et al., 1995; Nowacek et al., 2007), fish (Popper and Hastings, 2009a; Slabbekoorn et al., 2010) and cephalopods (André et al., 2011). High-intensity, short-term events such as seismic surveys, pile-driving operations and military sonar activities have been the focus of considerable attention due to their potential to cause physical injury and temporary or permanent loss of hearing sensitivity in marine mammals (e.g. Evans and England, 2001; Lucke et al., 2009; Bailey et al., 2010). Less intense sources can also elicit behavioural responses: boat noise, for example, has induced avoidance reactions in several cetacean species (Richardson and Würsig, 1997).

However, there is also growing recognition of the potential for long-term exposure to anthropogenic noise to induce chronic effects in marine species (Tyack, 2008; Slabbekoorn et al., 2010). These effects may occur at levels below those necessary to induce short-term behavioural responses, and through mechanisms which are more difficult to observe. They include masking of biologically significant sounds (Clark et al., 2009; Popper and Hastings, 2009b), chronic stress (Wright et al., 2007; Rolland et al., 2012), subtle long-term behavioural responses (Picciulin et al., 2010) and shifts in attention (Chan et al., 2010; Purser and Radford, 2011). *In situ* measurements of long-term exposure to anthropogenic noise both in absolute terms and relative to background levels are needed to inform further investigation in this area (Ellison et al., 2012).

Noise from shipping is pervasive throughout the marine environment, especially at low (<300 Hz) frequencies (Richardson et al., 1995; Chapman and Price, 2011), and is therefore a key concern regarding the effects of chronic noise exposure on marine species (Slabbekoorn et al., 2010). Deep water observations have shown that ambient noise levels have been rising since at least the 1960s due to increases in shipping traffic and tonnage (Andrew et al., 2002; McDonald et al., 2006; Chapman and Price, 2011). Ambient noise levels in shallower coastal waters are more difficult to characterise as they exhibit much higher spatiotemporal variability (Urick, 1983). This is partly due to the greater dependence of acoustic propagation on local environmental factors such as the sound speed profile and seabed composition (Jensen et al., 2011). Significantly, variability is also caused by a higher concentration of shipping, industrial activity, and biological noise sources: it is this combination of potentially conflicting acoustic interests that necessitates the development of noise assessment methodologies applicable to coastal environments. To be meaningful, these methodologies must incorporate metrics relevant to the assessment of impacts on marine life.

* Corresponding author. Tel.: +44 1225 385543.
E-mail address: n.d.merchant@bath.ac.uk (N.D. Merchant).

0025-326X/$ - see front matter © 2012 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.marpolbul.2012.05.004

For non-pulse sounds such as ship noise, sound exposure level (SEL) has been suggested as a suitable noise assessment metric for marine mammals (Southall et al., 2007) and fish (Popper and Hastings, 2009b). SEL is a cumulative measure of the acoustic energy of a sound throughout its temporal extent. Since coastal shipping noise is both persistent and dynamic (due to the presence of nearby vessels and more distant shipping), reliable measurement of sound exposure requires continuous monitoring. Previously, the large volumes of data accrued by such monitoring have rendered it impractical. However, advances in passive acoustic monitoring (PAM) technology and data processing capabilities are making measurement and analysis of continuous, long-term deployments feasible.

Hatch et al. (2008) made an extensive study of the Stellwagen Bank National Marine Sanctuary using 9 autonomous PAM devices over a 27-day period. The acoustic data were combined with Automatic Identification System (AIS) vessel tracking data, enabling analysis of the relationship between vessel movements and ambient noise levels. The purpose of the present study is to explore the efficacy of a similar approach using a single PAM device to assess long-term sound exposure from shipping. This would have clear benefits over a more complex experimental apparatus (ease of deployment, cost reduction, quantity of data) and could make more sophisticated analysis techniques accessible to a broader range of investigators.

2. Materials and methods

2.1. Deployment location

Falmouth Bay (Fig. 1) is a large and deep natural harbour at the western entrance to the English Channel. The Channel is one of the busiest seaways in the world with around 45,000 ship transits annually (McQuinn et al., 2011). Traffic within the Bay consists of commercial shipping into Falmouth Harbour to the north, recreational boating, and activity related to bunkering (refuelling) of large vessels. The Bay is located just outside the western boundary of the North Sea Sulphur Emission Control Area (SECA), which came into effect in August 2007 (European Commission, 2005). This led to an increase in demand for low sulphur fuel at Falmouth, such that by 2008 commercial shipping traffic in the Bay had doubled (Dinwoodie et al., 2012). Published figures from 2009 show total annual cargo ship arrivals to Falmouth of 1309 (Department for Transport, 2010).

2.2. Acoustic data

An Autonomous Multichannel Acoustic Recorder (AMAR; Jasco Applied Sciences Ltd.) was deployed in the Bay for 20 days between July 24 and August 13, 2010. It was positioned on a seabed of sand to muddy sand, 1.8 km offshore from Nare Head in waters ~30 m deep. The AMAR was mounted on a custom-fabricated frame containing an acoustically triggered pop-up buoy system, and was programmed to record continuously in 30-min blocks, sampling at 16 kHz and 24 bits, using a GeoSpectrum M8E-132 hydrophone (effective bandwidth 5 Hz–150 kHz). The frequency bandwidth of the recordings was therefore 5 Hz–8 kHz.

Acoustic data were calibrated via the hydrophone sensitivity (−165 dB re 1 V μPa^{-1}) and the AMAR pre-amplifier gain (0 dB), then processed using custom-written MATLAB scripts. The power spectral density (PSD) was calculated using a 1-s Hann window with 50% overlap for each 30-min measurement. 172 short (<1 s) bursts of system noise with exceptionally high amplitudes below 10 Hz were detected. These were purged using a frequency-sensitive noise gate. To reduce storage space, the mean PSD was then calculated in 60-s windows. The files were then concatenated to form a master file. This was used as the source file for the subsequent calculations of SPL and SEL (see below).

A 9 day period from 16:30 on July 24 to 16:30 on August 2 was selected for analysis. The remaining data were discarded since the signature of a single vessel dominated the acoustic spectrum from around 17:00 on August 2 onward, precluding analysis of surrounding shipping. The vessel was identified from AIS data as a 55-m tug within ~1 km of the deployment site throughout the period from August 2 to (at least) August 13. Its presence may have been related to bunkering or other industrial activities in the Bay. This feature was considered anomalous and of limited relevance to other coastal areas.

2.3. Ancillary data

The Automatic Identification System (AIS) is a vessel-tracking system which operates on VHF radio bandwidth and can be detected by land-based receivers. AIS transceivers are compulsory for vessels exceeding 299 GT (gross tonnes) according to the International Convention for the Safety of Life at Sea (SOLAS) (IMO, 1974). AIS data for the duration of the deployment period were provided by a Web-based ship-tracking network (http://www.shipais.com/). This covered the area 48.0–51.0N/1.0–7.0W, and included good coverage of Falmouth Bay and the surrounding area (see below). Hourly wind speed and rainfall data from the Culdrose weather station, 14 km to the west of the deployment location, were provided by the UK Met Office.

2.4. Calculation of sound pressure level and sound exposure level

Sound pressure level (SPL) is the mean square pressure expressed in decibels relative to a reference pressure. The mean square pressure, Q, is given by,

$$Q = \frac{1}{T}\int_0^T q^2(t)dt \tag{1}$$

where T is the integration time (the time over which the mean is calculated), and $q(t)$ is the instantaneous acoustic pressure at time t (Ainslie, 2010). The SPL is then

$$\mathrm{SPL} = 10\log_{10}\left(\frac{Q}{p_{\mathrm{ref}}^2}\right) \tag{2}$$

In underwater acoustics, p_{ref} is a reference pressure of 1 μPa at a distance of 1 m. The units of SPL are then dB re 1 μPa^2. Note that some authors express SPL in dB re 1 μPa; the levels are numerically equivalent (TNO, 2011).

An integration time of 300 s was used to calculate the SPL over a frequency bandwidth of 0.01–1 kHz. This bandwidth covers the nominal frequency range of commercial shipping noise (Tasker et al., 2010), and allowed comparison of recorded levels with relevant studies (e.g. Hatch et al., 2008; McKenna et al., 2012). The integration time was chosen such that the SPL varied over a similar timescale to the transmission rate of the AIS data (typically around 600 s). Reducing the time resolution of the acoustic data from 60 s to 300 s also reduced the temporal variability of the signal (smoothing). Consequently, ship passages were more likely to appear as unique local maxima in the SPL, rather than multiple maxima in the case of finer temporal resolution. This made it easier to identify ship passages from maxima in the SPL (see below).

The sound exposure level (SEL) is a cumulative measure of acoustic energy which allows the energy radiated by sounds of differing duration to be compared. It is a summation of multiple mean square pressures (consecutive or not) expressed in dB re 1 μPa^2 s:

$$\mathrm{SEL} = 10\log_{10}\left(\frac{\int Q(t')dt'}{p_{\mathrm{ref}}^2}\right) \tag{3}$$

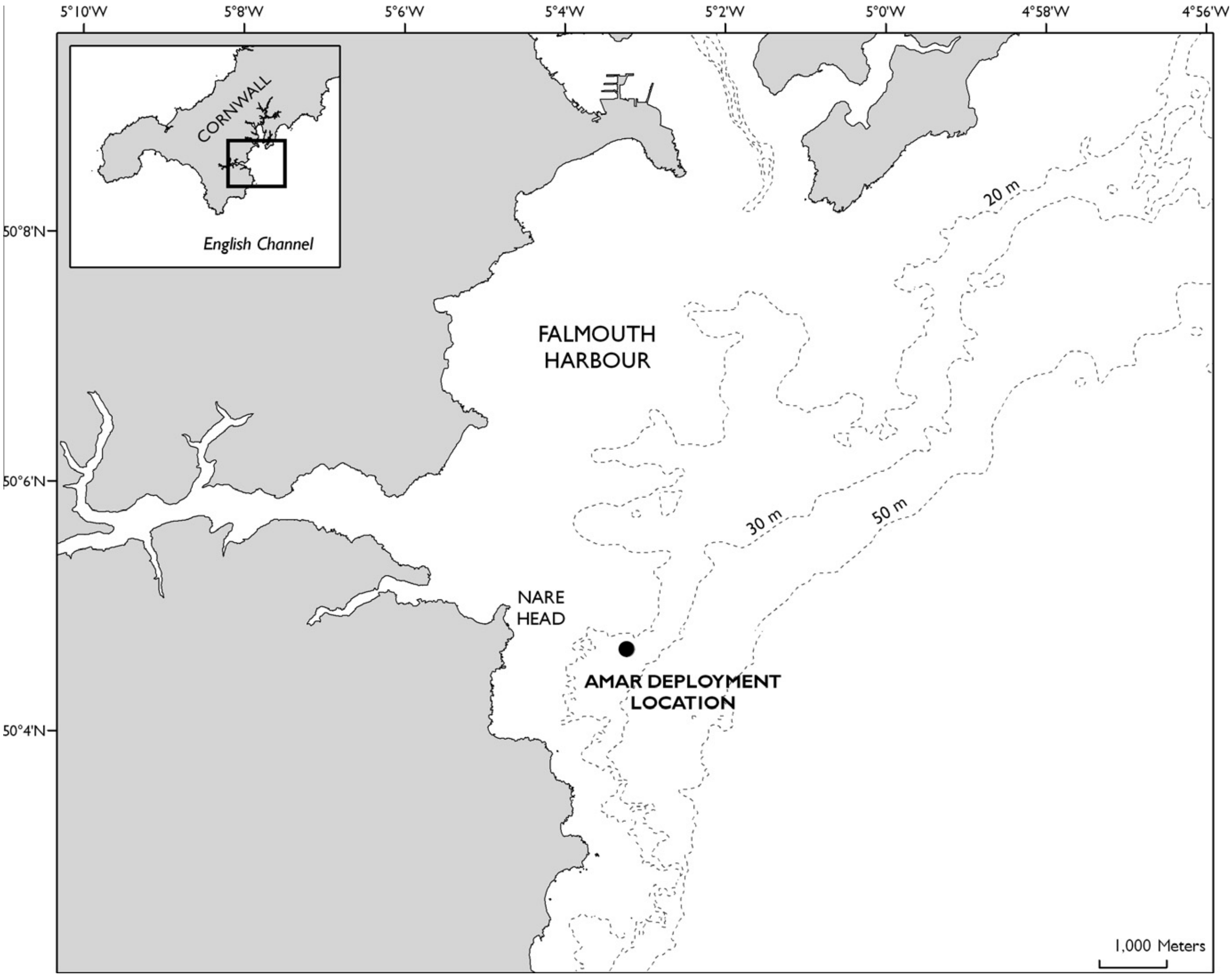

Fig. 1. Deployment location: Falmouth Bay, UK.

where Q is the mean square pressure at time t', and p_{ref} is as above. The SEL for each 24-h period was calculated using an integration time of 300 s over the nominal frequency bandwidth of shipping (0.01–1 kHz) and the full recorded bandwidth (5 Hz–8 kHz). The latter bandwidth was included to assess the effect of higher frequency components on sound exposure levels.

2.5. M-weightings

The M-weighted SEL for each 24-h period was also calculated. M-weightings are frequency weightings that can be applied to the SEL to adjust for the likely hearing sensitivity of marine mammals to high-amplitude acoustic sources (Southall et al., 2007). They are analogous to C-weightings used in terrestrial noise impact assessment for humans, and give an indication of the relative impact of noise sources on four broad functional hearing groups of marine mammals. The application of M-weightings to lower amplitude, chronic sources of noise is questionable since it is likely they overestimate the sensitivity of hearing (McQuinn et al., 2011). In this study, they are used as a notional indication of the relative impact of shipping noise on different marine mammal groups.

The M-weighting group most receptive to the nominal frequency range of shipping noise (0.01–1 kHz) is low-frequency cetaceans (baleen whales), followed by pinnipeds, mid- and high-frequency cetaceans (Fig. 2). Boats can emit significant levels of underwater noise above 1 kHz, particularly small vessels with outboard motors (Au and Green, 2000; McQuinn et al., 2011). To assess the contribution of these higher frequency components, the M-weighted levels over the full recorded bandwidth (5 Hz–8 kHz) were also calculated.

2.6. Separation of intermittent ship noise from background

Intermittent ship noise was identified using an adaptive threshold. The threshold adapts to long-term variations in the broadband SPL while distinguishing short-term, relatively high-amplitude events. This enables the relative level of shipping noise exposure above the background to be determined. This was considered preferable to a fixed threshold, which would be insensitive to the temporal variability of ambient (background) noise and would have to be adjusted for different study areas due to the spatial heterogeneity of ambient noise. Another consideration is that ambient noise characteristics affect the degree of auditory masking (Clark et al., 2009) and are likely to influence behavioural responses to anthropogenic noise (Southall et al., 2007). The relative level of anthropogenic noise exposure is therefore a key metric in acoustic impact assessment (Ellison et al., 2012).

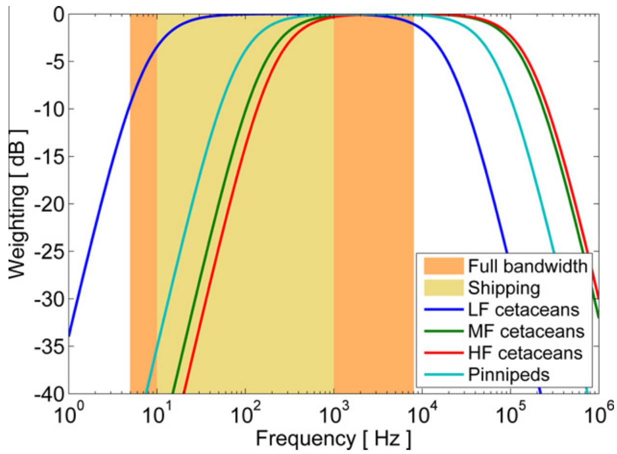

Fig. 2. M-weightings for low-, medium- and high-frequency cetaceans and pinnipeds (in water) (Southall et al., 2007). The shaded areas indicate the frequency bandwidth of the recordings ('Full Bandwidth'; 5 Hz–8 kHz) and the nominal frequency bandwidth of shipping noise ('Shipping'; 10 Hz–1 kHz) used in this study for the calculation of SPL and SEL.

The adaptive threshold works on the assumption that the minimum recorded SPL over a given period is representative of the background noise level within that period. This period is the background window duration, W, which is chosen to be long enough that each window has data free from the noise source, and short enough to adapt to more gradual variations in ambient noise level. A tolerance above the minimum SPL, the threshold ceiling, C [dB], is then defined. As for W, C may be tailored for the application. The time-dependent adaptive threshold level, ATL(t), for a time-dependent SPL, SPL(t), is then:

$$\mathrm{ATL}(t) = \min[\mathrm{SPL}(t)]_{t-W/2}^{t+W/2} + C \tag{4}$$

where ATL(t) has units of dB re 1 μPa^2. In other words, ATL(t) is C decibels above the minimum recorded SPL within a rolling time window of duration W centred on time t.

In this study, W was set to 3 h and C to 6 dB (i.e. double the minimum level). This value of W was necessary because of sustained periods of local shipping noise with durations approaching 3 h. C was selected by experimentation and for simplicity: it was found to effectively distinguish background and intermittent contributions to the 24-h SEL (see below).

Data above the threshold were classed 'intermittent', data below the threshold 'background'. Maxima in the intermittent SPL data were detected for subsequent comparison to AIS data (see below). The intermittent and background SELs were then calculated for each 24-h period. An estimate of the SEL in the absence of intermittent data was also made. This was calculated by substituting the intermittent data with the median background level computed with a rolling 3-h window.

2.7. Spatial distribution of peak-generating ships

To assess the spatial distribution of ships generating intermittent peaks in the SPL, a graphical user interface (GUI) was designed in MATLAB. The GUI allows the operator to analyse each peak in the intermittent SPL with reference to figures displaying the tracks of AIS transmissions, the calibrated spectrogram, and the broadband SPL for a two-hour window centred on the SPL peak.

Firstly, the distance of each AIS transmission from the deployment location was calculated from its latitude and longitude coordinates. Transmissions within 50 km were plotted against time, linking data points from the same vessel (identified in the AIS log by a unique Maritime Mobile Service Identity (MMSI) number). The closest points of approach (CPAs) of each vessel were then computed geometrically, assuming each vessel maintained a direct course and constant speed between AIS transmissions (the transmission rate is typically around 10 min, although this can vary).

For each peak in the intermittent SPL, CPAs within a 15-min window centred on the peak (i.e. ±7.5 min) were considered. This assumes that CPAs coincide with peak SPLs, allowing a tolerance of ± 1 SPL data point (each of which comprises 5 min). Since acoustic propagation loss generally increases with distance (Urick, 1983) and the horizontal directionality of radiated ship noise appears maximal at broadside aspect (Arveson and Vendittis, 2000; Trevorrow et al., 2008), this was considered a reasonable assumption.

Finally, the spectrogram was consulted to confirm whether SPL peaks were due to ship signatures and not, for example, wind noise. These are readily distinguished by the tonal components present in ship noise signatures. Each SPL peak was then categorised as being uniquely identified (one CPA), due to multiple possible sources (more than one CPA), or unidentified (no CPA). The coordinates of each uniquely identified CPA were then recorded.

3. Results

3.1. Ambient noise spectrum and weather data

The ambient noise field was punctuated by wide bands of intermittent noise, some of which spanned the entire frequency range (Fig. 3). These were attributable to shipping (see below). The spectral energy of intermittent noise events was concentrated in the frequency range 0.01–1 kHz, which supports the use of this nominal bandwidth for shipping noise assessment.

Mean hourly wind speeds at the Culdrose weather station ranged from 2 to 17 knots (1.0 – 8.7 m s^{-1}), with a maximum hourly increase of 6 knots (3.1 m s^{-1}). Wind speeds in this range have been associated with variations of up to around 20 dB in shallow water ambient noise levels (Urick, 1983). Spectra characteristic of wind noise did not feature in the frequency spectrum of the intermittent component, which was reviewed visually. This implies that either the wind-generated noise was below the adaptive threshold, meaning that the rate of increase in broadband (0.01–1 kHz) SPL due to wind did not exceed 6 dB per 1.5 h (4 dB per hour), or that any rapid increases in wind speed were masked by local vessel activity. Rainfall was recorded at Culdrose in 12 h of data over the 9 day period, with a maximum rate of 0.8 mm per hour. Since rain generates noise at frequencies above 1 kHz (Nystuen, 2001), it was not considered to contribute to the broadband (0.01–1 kHz) levels used for noise classification.

3.2. Sound pressure levels

Overall, broadband (0.01–1 kHz) SPLs ranged from 86.1 to 148.6 dB re 1 μPa^2. The SPL was above the threshold level ('intermittent') 29% of the time, and below 71% ('background'). SPLs from a representative day are presented in Fig. 4.

The median threshold level was 96.2 dB re 1 μPa^2, with a range of 10.6 dB. Intermittent peaks in the SPL ranged from 92.8 to 148.6 dB re 1 μPa^2, and exceeded the threshold by a median of 6.4 dB. In total, there were 314 peaks in the intermittent SPL data (mean: 34.9 per day).

3.3. Sound exposure levels

The broadband SEL for each 24-h period between 16:30 on July 24 and 16:30 on August 2 was calculated over the frequency ranges 0.01–1 kHz (nominal shipping bandwidth) and 5 Hz–8 kHz (full bandwidth). The median and maximum SELs are

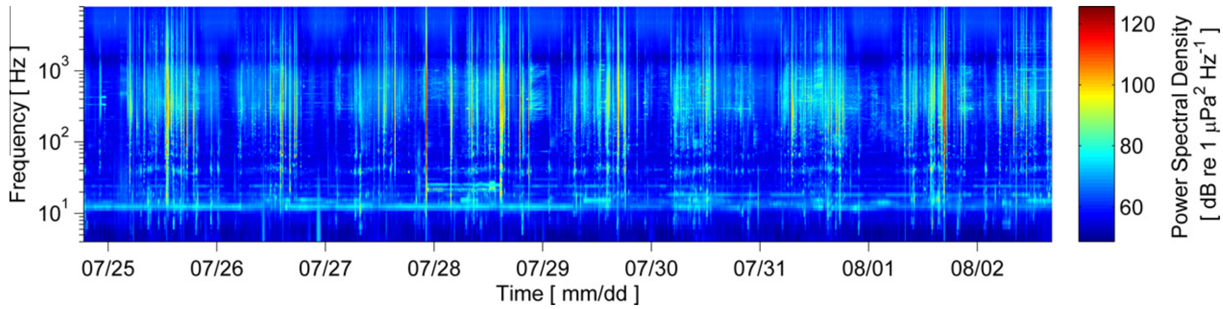

Fig. 3. Power spectral density for 9 days of continuous monitoring. Frequency bandwidth 5 Hz–8 kHz, integration time 300 s.

presented in Table 1. Over both frequency ranges, the total SEL was dominated by the contribution of the intermittent component. This was especially the case over the nominal shipping bandwidth, where the median total SEL was 14.4 dB greater than the estimated level in the absence of the intermittent events ('24-h background'). In the 24-h period with maximal total SEL (27–28 July), the intermittent component (27% of the time series in this period) raised the SEL in this frequency range by 28.9 dB above the 24-h background level.

The median 24-h SEL was concentrated above ~100 Hz, with a broad peak at 315 Hz (Fig. 5a). The intermittent component was most dominant between around 30 and 2,000 Hz (Fig. 5a). The variability of the intermittent data (Fig. 5c) appears to account for the variability of the total 24-h SEL (Fig. 5b) above ~30 Hz. In contrast, the 24-h SEL of the background component was comparatively stable at all frequencies (Fig. 5d).

Above around 2 kHz, the median background levels rose (Fig. 5d). Consequently, the background SELs across the two frequency bandwidths differed by ~5 dB (since only the full bandwidth SEL included this component) (Table 1). This high frequency component was the least variable part of the background sound exposure (Fig. 5d), and consisted of impulsive noise exhibiting a diurnal periodicity with maxima during the night (Merchant et al., 2011). It is probable that this noise was produced by snapping shrimp: these decapods generate characteristic impulses with peak frequencies in this range (Au and Banks, 1998; Radford et al., 2008). Two species of snapping shrimp have been documented in coastal waters to the east of the deployment site: *Alpheus glaber* near Plymouth (Holme, 1966) and *Alpheus macrocheles* further east around Weymouth (Holme, 1966; Hinz et al., 2011). There have also been unpublished reports of *A. macrocheles* caught by fishermen in Falmouth Bay.

As expected, the M-weighting for low-frequency cetaceans yielded the highest SELs, followed by pinnipeds, mid- and high-frequency cetaceans (Table 1). The M-weighted SEL for low-frequency cetaceans was equivalent to the unweighted level: this weighting is flat in the range 0.1–1 kHz (Fig. 2) where the SEL was concentrated (Fig. 5a). The M-weighted full-bandwidth total SELs were only marginally higher (1.2–1.8 dB) than for the nominal shipping bandwidth (Table 1), reflecting the concentration of shipping noise between 0.1 and 1 kHz in this study. In contrast, the full-bandwidth background SELs were 6.3–7.4 dB higher due to the high frequency contribution of impulsive noise.

In summary, the 24-h SEL comprised a stable background component (71% of the time series) and a more variable intermittent component (29%). The SEL of this intermittent component determined the magnitude and variability of the total SEL.

3.4. Spatial distribution of peak-generating ships

Peak-generating ships were identified manually using a GUI which displayed the AIS and acoustic data as shown in Fig. 6. Each of the peaks in the broadband SPL was categorised as uniquely identified, due to multiple ship sources, or unidentified, based on the number of CPAs within ±7.5 min of the peak. For example, in Fig. 6 the intermittent peak at 01:50 was classed as uniquely identified and attributed to the vessel 212032000. The previous peak at 01:30 was unidentified as there were no CPAs within its 15-min window.

The AIS coverage of the Falmouth Bay area was not continuous throughout the deployment, and data were unavailable for 126 of the 314 peaks recorded. Of the remaining 188 peaks, 59 (31%) were classed as uniquely identified, 61 (32%) as due to multiple possible sources, and 68 (36%) as unidentified. Visual inspection of each plot suggested that 18 of the uniquely identified peaks could not unambiguously be attributed to individual CPAs, and were instead attributed to multiple ship sources. These 'false positives' were typically due to substantial shipping activity closer to the deployment than the identified vessel. A further 5 peaks having two CPAs in the 15-min window were clearly attributable to one of the CPAs. All 5 cases involved large (>77-m length) commercial vessels close to the deployment. Figures showing these 5 cases and 3 examples of false positives are presented in Supplementary data. The classification of peaks was then 46 (24%) uniquely identified, 74 (40%) due to multiple ship sources, and 68 (36%) unidentified.

Of the uniquely identified vessels, 24 were cargo ships, 13 were tankers and the remaining 9 consisted of 3 fishing boats, 2 military vessels, a research vessel, a pilot vessel, a recreational craft and an icebreaker. Peak broadband (0.01–1 kHz) SPLs attributed to these vessels ranged from 92.8 to 148.6 dB re 1 μPa^2, with CPAs between 0.18 and 34.1 km from the hydrophone. Potential sources of the unidentified peaks include vessels <300 GT not transmitting AIS signals, ship noise unrelated to the passage of ships (engine

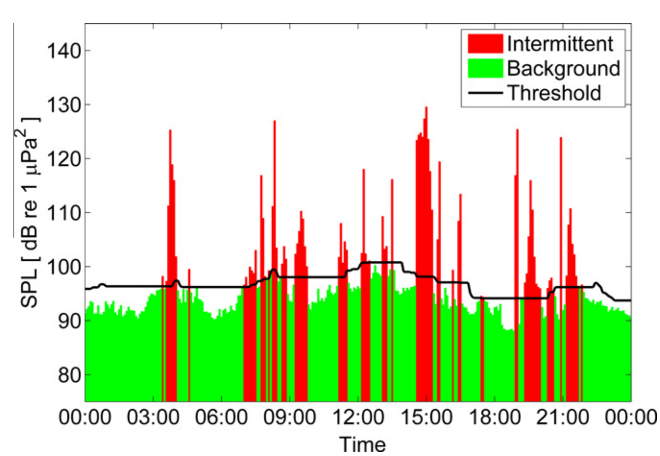

Fig. 4. Broadband (0.01–1 kHz) SPL for a representative 24-h period (28 July) showing classification of 'intermittent' and 'background' data. Integration time: 300 s. The solid line is the adaptive threshold level.

Table 1
Median and maximum 24-h SELs, calculated from 9 consecutive 24-h periods. '24-h background' is the estimated 24-h SEL in the absence of intermittent noise events.

	Nominal shipping bandwidth (10 Hz–1 kHz)		Full bandwidth (5 Hz–8 kHz)	
	Median 24-h SEL (±range) (dB re 1 μPa^2 s)	Maximum 24-h SEL (dB re 1 μPa^2 s)	Median 24-h SEL (±range) (dB re 1 μPa^2 s)	Maximum 24-h SEL (dB re 1 μPa^2 s)
Unweighted				
24-h total	157.0 ± 19.1	173.9	158.3 ± 17.9	174.3
Intermittent	156.9 ± 19.4	173.9	157.9 ± 18.5	174.3
Background	141.1 ± 3.9	143.5	147.4 ± 1.8	148.5
24-h background	142.6 ± 3.4	145.0	149.1 ± 1.4	150.2
Low-frequency cetaceans				
24-h total	157.0 ± 19.1	173.9	158.2 ± 17.9	174.3
24-h background	142.5 ± 3.5	145.0	148.8 ± 1.5	150.0
Mid-frequency cetaceans				
24-h total	155.2 ± 18.7	171.7	156.9 ± 17.0	172.1
24-h background	141.6 ± 3.6	144.0	148.8 ± 1.3	149.8
High-frequency cetaceans				
24-h total	154.5 ± 18.5	170.8	156.3 ± 16.5	171.3
24-h background	141.2 ± 3.6	143.6	148.6 ± 1.3	149.6
Pinnipeds				
24-h total	156.3 ± 19.1	173.1	157.7 ± 17.6	173.5
24-h background	142.1 ± 3.6	144.6	149.0 ± 1.4	150.0

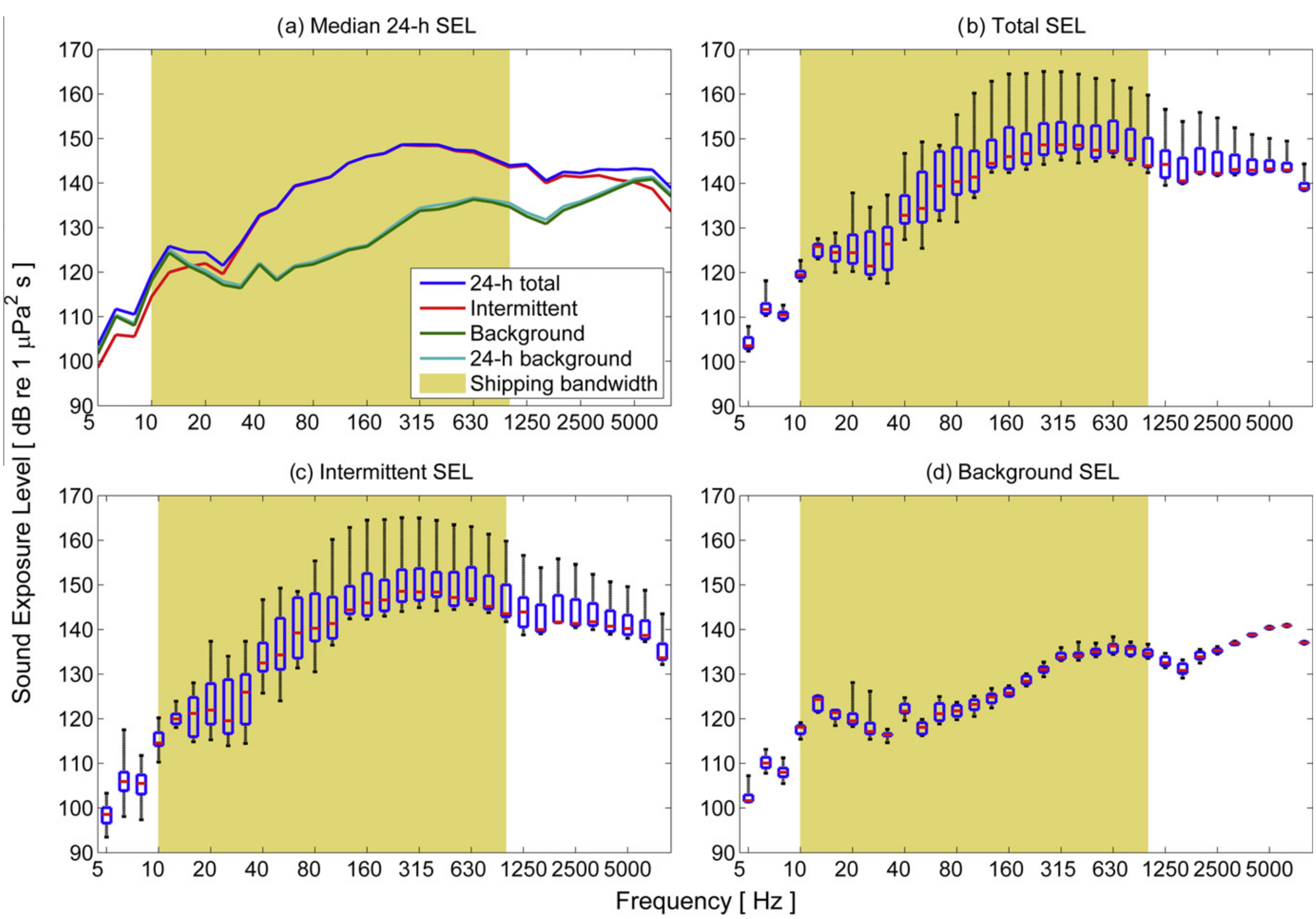

Fig. 5. (a) Median 24-h SEL in third-octave bands, calculated from 9 consecutive 24-h periods. (b–d) Total SEL and SEL due to intermittent and background components. The centre lines of the boxes denote the median and the box limits indicate the first and third quartiles. The whiskers are the maximum and minimum values recorded. The shaded areas indicate the nominal bandwidth of shipping noise (0.01–1 kHz).

activity, manoeuvring, bunkering operations, etc.), and vessels outside the 50 km range considered.

The coordinates of uniquely identified CPAs were distributed within Falmouth Bay and further south into the English Channel (Fig. 7). The largest cluster of CPAs to the east of the deployment corresponds to the paths of vessels entering and leaving Falmouth Harbour and the Bay. A second cluster ~15 km south of the deployment site corresponds to paths of vessels navigating along the coast past the headland at Lizard Point. Small vessels were distributed within the Bay close to the deployment site, while the main shipping routes were populated by tankers and cargo ships. The tanker furthest east in the English Channel appears to have been falsely identified as the coast obscures the line of sight to the hydrophone. Error in the position of the CPA could also be the cause, since these were calculated assuming constant speed and direct trajectories between AIS transmissions.

4. Discussion

The assessment of shipping noise in coastal waters is complicated by the presence of both intermittent noise from local vessel traffic and ambient noise from distant shipping. We have shown that these two components are clearly distinguished by the nature of their contribution to the 24-h SEL, and can be separated by applying an adaptive threshold to the sound pressure level. Intermittent ship noise produced a variable, high amplitude component (Fig. 5c) which determined the magnitude and variability of the total 24-h SEL (Fig. 5b, Table 1). A lower amplitude 'background' component remained stable over the 9 days analysed (Fig. 5d).

Analysing the sound exposure in this way makes it possible to assess both the absolute sound exposure at the deployment location and the contribution of intermittent shipping noise relative to background levels. In the nominal frequency range of shipping noise (0.01–1 kHz), we recorded a median 24-h SEL of 157.0 dB re 1 μPa^2 s compared to an estimated 142.6 dB re 1 μPa^2 s in the absence of intermittent shipping noise. Both elements are necessary to inform the investigation of chronic noise exposure on marine species (Ellison et al., 2012). Absolute SELs in representative marine habitats can be used in controlled studies of noise exposure (e.g. Codarin et al., 2009; Purser and Radford, 2011), while relative levels are needed to understand the relative impact of anthropogenic sources on the marine acoustic environment.

It is important to note that background levels are likely to be heightened by shipping noise below the level of the adaptive threshold applied to the SPL time series. The background level should therefore be understood as the estimated level in the absence of significant local shipping activity, not in the absence of shipping noise *per se*. In this study the 24-h SEL was determined by the intermittent component which constituted 29% of the time series. The intermittent component may be less dominant in coastal areas with a lower density of local shipping, and where there are fewer large commercial vessels.

By relating the acoustic data to the CPAs of AIS-transmitting vessels, it was possible to account for 64% of peaks in the intermittent SPL for which AIS data were available as being due to shipping. Twenty-four percent of peaks appeared to be uniquely attributable to individual vessel passages. The spatial distribution of uniquely identified vessels (Fig. 7) indicates that the majority were large commercial vessels transiting either along the northern side of the English Channel or into Falmouth Bay. Although relatively

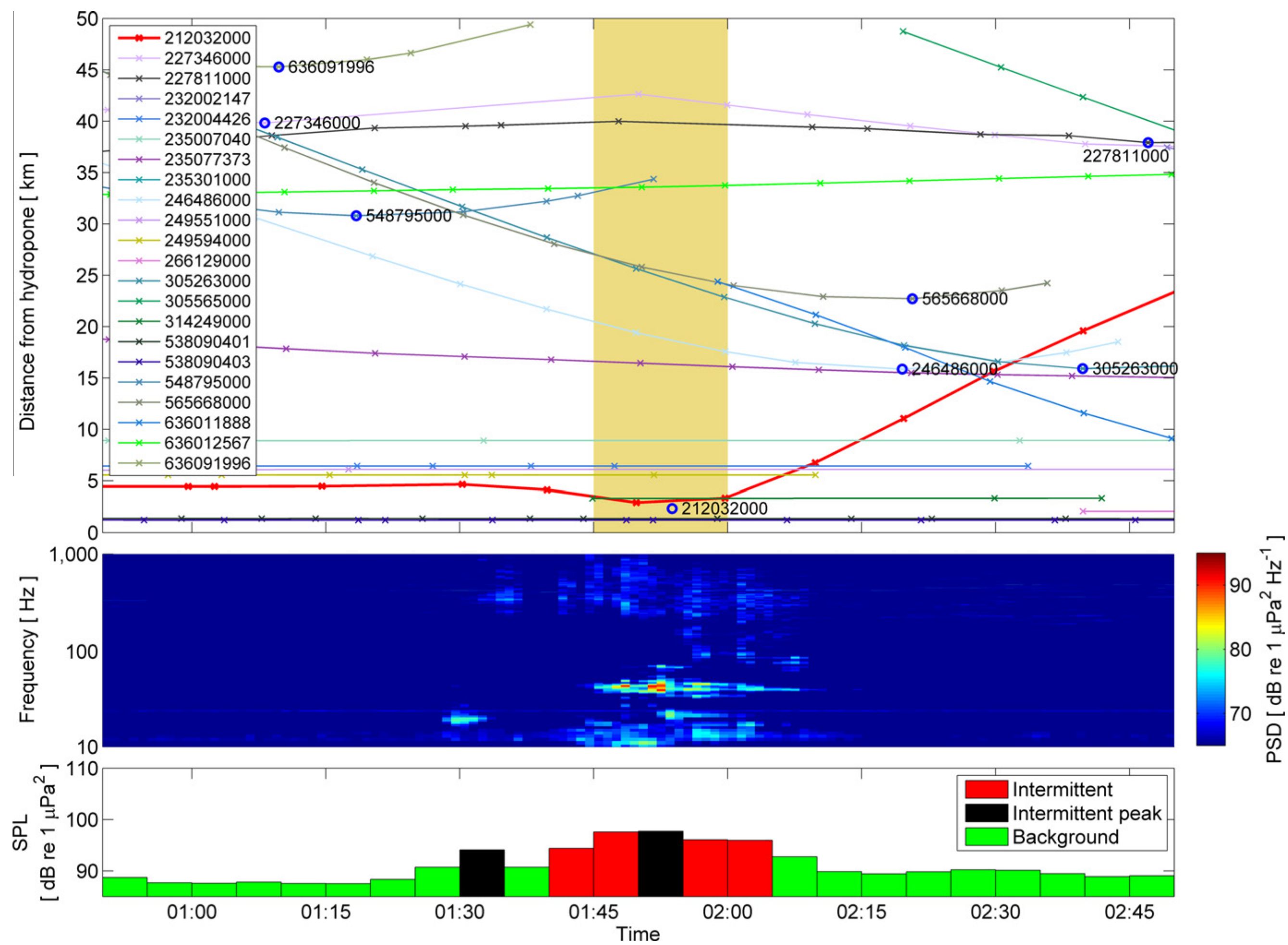

Fig. 6. Example of ship identification using AIS data. *Top*: Range from hydrophone vs. time. Crosses denote individual AIS transmissions; lines connect transmissions from the same vessel; circles indicate closest points of approach, labelled with the MMSI number. Shaded area denotes 15-min time window around SPL peak at 01:50; heavy line indicates track of vessel identified as source of peak. Note that the horizontal lines indicate AIS transmissions from stationary vessels. *Middle*: Power spectral density of concurrent acoustic data. *Bottom*: Broadband (0.01–1 kHz) SPL, showing 'background', 'intermittent', and 'intermittent' peaks.

few small vessels were identified, these may constitute only a small proportion of the overall small vessel fleet operating in the Bay, since AIS transceivers are only mandatory for vessels over 299 GT. The absence of these vessels from the AIS data may partially account for the 36% of acoustic peaks which remained unidentified.

Several factors limited the identification of ship sources of noise in the study area. Firstly, the density of shipping within a 50 km radius was high: multiple potential ship sources were identified for 40% of peaks, and manual oversight was necessary to detect ambiguous identifications, preventing automation of the technique. Secondly, it was clear that many vessels were mooring in Falmouth Bay, possibly for bunkering services. This meant it was often not possible to determine the CPA, and that ship noise not associated to CPAs such as manoeuvring, bunkering activity and idling was detected but could not be uniquely attributed to vessels by this method. Consequently, it is suggested that this method may be more successful in locations where most shipping traffic is transiting the deployment site, or where the density of shipping is lower.

One application of this approach could be for site-specific assessment of shipping noise in designated regions such as Marine Protected Areas (MPAs). There is evidence that current exclusion zones in MPAs deemed acoustically sensitive may be insufficient (Agardy et al., 2007; Haren, 2007; Hatch and Fristrup, 2009), and several authors have recommended the use of buffer zones in addition to exclusion zones (Hatch et al., 2008; Codarin et al., 2009; Wright et al., 2011). Since many MPAs are located in coastal waters (Toropova et al., 2010), where land-based receivers can track AIS transmissions of vessels, this assessment technique could be used to measure the spatial distribution of significant ship noise sources. This would help to prioritise shipping noise mitigation strategies, such as ship-quieting, speed restrictions and rerouting of shipping lanes, leading to more informed environmental management of shipping noise pollution.

The shipping noise recorded in Falmouth Bay was predominantly within the nominal frequency range of shipping (0.01–1 kHz; Figs. 3 and 5c), and the inclusion of higher frequencies (up to 8 kHz) resulted in total SELs only ~1 dB higher (Table 1). However, the peak frequency of sound exposure from intermittent ship noise (315 Hz) was considerably higher than that of reported source spectra for large commercial vessels, which are typically around 100 Hz or below (Arveson and Vendittis, 2000; Wales and Heitmeyer, 2002; McKenna et al., 2012). Propagation of sound in shallow water is subject to high attenuation at both high and low frequencies (Jensen et al., 2011), and favourable propagation at mid-frequencies may partly explain the spectral composition

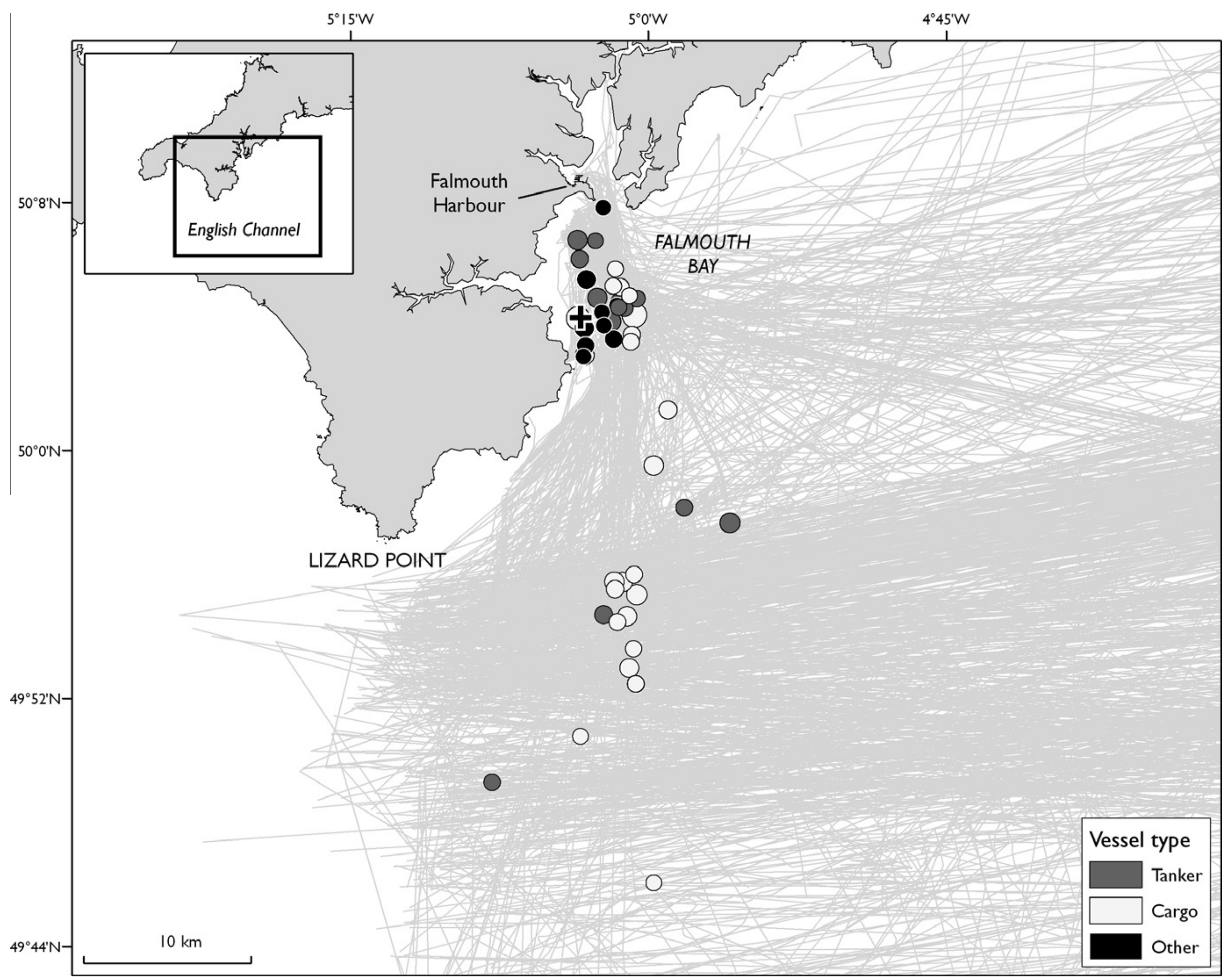

Fig. 7. Positions of uniquely identified CPAs categorised by vessel type. The size of each circle corresponds to the magnitude of the associated SPL peak, ranging from 92.8 to 148.6 dB re 1 μPa^2. Cross denotes the location of the deployment. Lines indicate paths of AIS transmissions during the deployment period.

of noise observed. A more significant factor is likely to be the composition of the shipping fleet contributing to underwater noise, which may have included more small vessels than were indicated by the AIS data.

Received SPLs of transiting vessels were comparable to previous studies. Peak SPLs of uniquely identified CPAs were between 92.8 and 148.6 dB re 1 μPa^2 for CPAs ranging from 0.18 to 34.1 km. McKenna et al. (2012) reported received levels of noise from 29 commercial vessels of 106.0 to 117.9 dB re 1 μPa^2 for CPAs at distances of 2.6–3.5 km over a similar frequency range (0.02–1 kHz). Hatch et al. (2008) reported received levels ranging from 113 to 131 dB re 1 μPa^2 for CPAs between 0.4 and 3.4 km over a narrower frequency range (71–141 Hz). In both studies, the narrower range of received levels reflects the narrower range of CPAs.

The equivalence of the unweighted 24-h SEL and the M-weighted level for low-frequency cetaceans (Table 1) highlights the degree of overlap between likely baleen whale hearing ranges and the dominant frequencies of radiated ship noise. The received SPLs of vessels observed in Falmouth Bay (92.8–148.6 dB re 1 μPa^2) are within ranges at which baleen whales have been observed to exhibit behavioural responses, which are particularly acute above received SPLs of around 120 dB re 1 μPa^2 (Southall et al., 2007). Recent evidence points towards increased stress levels in right whales associated to shipping noise (Rolland et al., 2012), though the long-term consequences for baleen whales and other marine mammals of sustained exposure to shipping noise remain largely unknown.

The dominance of ship noise in the range 0.1–1 kHz also coincides with the frequencies of greatest hearing sensitivity for many fish species (Popper and Hastings, 2009a). The SPLs of ship passages observed in this frequency range (Figs. 3,4) are at levels which may cause masking of communication in vocal fish species, as has been observed in several impact studies (e.g. Vasconcelos et al., 2007; Codarin et al., 2009). Exposure to ship noise may also have longer term effects associated to physiological stress responses (Wysocki et al., 2006) and reduced foraging efficiency (Purser and Radford, 2011).

In this study, the PAM device was mounted on the seafloor, which has potential drawbacks related to acoustic propagation. In shallow water, propagation is strongly affected by interactions with the seabed and varies with depth (Kuperman and Lynch, 2004). Consequently, the noise levels recorded by bottom-mounted PAM devices may differ from levels recorded elsewhere in the water column. The potential for these effects could be reduced by positioning the hydrophone in the water column suspended on a buoy.

There is increasing awareness of the potential for chronic exposure to shipping noise to have harmful impacts on marine ecosystems. Developing techniques to measure long-term sound exposure in coastal habitats is a necessary step towards understanding how these dynamic acoustic environments affect marine fauna. Our results suggest that by using continuous acoustic monitoring to determine the 24-h sound exposure level, the contribution of intermittent shipping to underwater noise levels can be assessed with greater clarity. Further work is needed to establish the efficacy of this approach in other coastal environments. The method we present of analysing the spatial distribution of ship contributions to noise exposure using AIS data could be used to inform the prioritisation of mitigation strategies in acoustically sensitive areas.

Acknowledgements

NDM is funded by an EPSRC Doctoral Training Award (#EP/P505399/1). BJG and GHS are funded by NERC, PRIMaRE and the South West Regional Development Agency. We gratefully acknowledge I. McConnell for providing the AIS log, L. Johanning, D. Parish and D. Raymond for assisting with the deployment and for access to the deployment site, J. Seal for helpful comments on the manuscript, and Falmouth Harbour Commissioners for their continued support.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.marpolbul.2012.05.004.

References

Agardy, T., Aguilar, N., Cañadas, A., Engel, M., Frantzis, A., Hatch, L., Hoyt, E., Kaschner, K., LaBrecque, E., Martin, V., Notarbartolo di Sciara, G., Pavan, G., Servidio, A., Smith, B., Wang, J., Weilgart, L., Wintle, B., Wright, A., 2007. A Global Scientific Workshop on Spatio-Temporal Management of Noise. Report of the Scientific Workshop. 44pp.

Ainslie, M.A., 2010. Principles of Sonar Performance Modelling. Springer-Praxis, UK.

André, M., Solé, M., Lenoir, M., Durfort, M., Quero, C., Mas, A., Lombarte, A., van der Schaar, M., López-Bejar, M., Morell, M., Zaugg, S., Houégnigan, L., 2011. Low-frequency sounds induce acoustic trauma in cephalopods. Frontiers in Ecology and the Environment 9 (9), 489–493.

Andrew, R.K., Howe, B.M., Mercer, J.A., Dzieciuch, M.A., 2002. Ocean ambient sound: comparing the 1960s with the 1990s for a receiver off the California coast. Acoustics Research Letters Online 3 (2), 65–70.

Arveson, P.T., Vendittis, D.J., 2000. Radiated noise characteristics of a modern cargo ship. Journal of the Acoustical Society of America 107 (1), 118–129.

Au, W.W.L., Banks, K., 1998. The acoustics of the snapping shrimp Synalpheus parneomeris in Kaneohe Bay. Journal of the Acoustical Society of America 103 (1), 41–47.

Au, W.W.L., Green, M., 2000. Acoustic interaction of humpback whales and whale-watching boats. Marine Environmental Research 49 (5), 469–481.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G., Thompson, P.M., 2010. Assessing underwater noise levels during pile-driving at an offshore windfarm and its potential effects on marine mammals. Marine Pollution Bulletin 60 (6), 888–897.

Chan, A., Giraldo-Perez, P., Smith, S., Blumstein, D.T., 2010. Anthropogenic noise affects risk assessment and attention: the distracted prey hypothesis. Biology Letters 6 (4), 458–461.

Chapman, N.R., Price, A., 2011. Low frequency deep ocean ambient noise trend in the Northeast Pacific Ocean. Journal of the Acoustical Society of America 129 (5), EL161–EL165.

Clark, C.W., Ellison, W.T., Southall, B.L., Hatch, L., Van Parijs, S.M., Frankel, A., Ponirakis, D., 2009. Acoustic masking in marine ecosystems: intuitions, analysis, and implication. Marine Ecology Progress Series 395, 201–222.

Codarin, A., Wysocki, L.E., Ladich, F., Picciulin, M., 2009. Effects of ambient and boat noise on hearing and communication in three fish species living in a marine protected area (Miramare, Italy). Marine Pollution Bulletin 58 (12), 1880–1887.

Department for Transport, 2010. Transport Statistics Report: Maritime Statistics 2009. DfT, London.

Dinwoodie, J., Tuck, S., Knowles, H., Benhin, J., Sansom, M., 2012. Sustainable development of maritime operations in ports. Business Strategy and the Environment 21 (2), 111–126.

Ellison, W.T., Southall, B.L., Clark, C.W., Frankel, A.S., 2012. A new context-based approach to assess marine mammal behavioral responses to anthropogenic sounds. Conservation Biology 26 (1), 21–28.

European Commission, 2005. Directive 2005/33/EC of the European Parliament and of the Council of 6 July 2005 amending Directive 1999/32/EC as regards the sulphur content of marine fuels. Official Journal of the European Union L191, 59–69.

Evans, D.L., England, G.R.E., 2001. Joint interim report Bahamas marine mammal stranding event of 15–16 March 2000. Tech. Rep., US Department of Commerce and Secretary of the Navy. 59pp.

Haren, A.M., 2007. Reducing noise pollution from commercial shipping in the Channel Islands National Marine Sanctuary: a case study in marine protected area management of underwater noise. Journal of International Wildlife Law and Policy 10 (2), 153–173.

Hatch, L., Clark, C., Merrick, R., Van Parijs, S., Ponirakis, D., Schwehr, K., Thompson, M., Wiley, D., 2008. Characterizing the relative contributions of large vessels to total ocean noise fields: a case study using the Gerry E. Studds Stellwagen Bank National Marine Sanctuary. Environmental Management 42 (5), 735–752.

Hatch, L., Fristrup, K., 2009. No barrier at the boundaries: implementing regional frameworks for noise management in protected natural areas. Marine Ecology Progress Series 395, 223–244.

Hinz, H., Capasso, E., Lilley, M., Frost, M., Jenkins, S.R., 2011. Temporal differences across a bio-geographical boundary reveal slow response of sub-littoral benthos to climate change. Marine Ecology Progress Series 423, 69–82.

Holme, N.A., 1966. The bottom fauna of the English Channel Part II. Journal of the Marine Biological Association of the United Kingdom 46 (02), 401–493.

IMO, 1974. International convention for the Safety of Life at Sea (SOLAS), Chapter V Safety of Navigation, Regulation 19 (amended Dec 2000).

Jensen, F.B., Kuperman, W.A., Porter, M.B., Schmidt, H., 2011. Computational Ocean Acoustics. Springer, New York.

Kuperman, W.A., Lynch, J.F., 2004. Shallow-water acoustics. Physics Today 57 (10), 55–61.

Lucke, K., Siebert, U., Lepper, P.A., Blanchet, M.A., 2009. Temporary shift in masked hearing thresholds in a harbor porpoise (Phocoena phocoena) after exposure to seismic airgun stimuli. Journal of the Acoustical Society of America 125 (6), 4060–4070.

McDonald, M.A., Hildebrand, J.A., Wiggins, S.M., 2006. Increases in deep ocean ambient noise in the Northeast Pacific west of San Nicolas Island, California. Journal of the Acoustical Society of America 120 (2), 711–718.

McKenna, M.F., Ross, D., Wiggins, S.M., Hildebrand, J.A., 2012. Underwater radiated noise from modern commercial ships. Journal of the Acoustical Society of America 131 (1), 92–103.

McQuinn, I.H., Lesage, V., Carrier, D., Larrivée, G., Samson, Y., Chartrand, S., Michaud, R., Theriault, J., 2011. A threatened beluga (*Delphinapterus leucas*) population in the traffic lane: vessel-generated noise characteristics of the Saguenay – St. Lawrence Marine Park, Canada. Journal of the Acoustical Society of America 130 (6), 3661–3673.

Merchant, N.D., Witt, M.J., Blondel, P., Godley, B.J., Smith, G.H., 2011. Ambient noise in the Western English Channel: temporal variability due to shipping and biological sources. Proceedings of the Institute of Acoustics 33 (5), 27–29.

Nowacek, D.P., Thorne, L.H., Johnston, D.W., Tyack, P.L., 2007. Responses of cetaceans to anthropogenic noise. Mammal Review 37 (2), 81–115.

Nystuen, J.A., 2001. Listening to raindrops from underwater: An acoustic disdrometer. Journal of Atmospheric and Oceanic Technology 18 (10), 1640–1657.

Picciulin, M., Sebastianutto, L., Codarin, A., Farina, A., Ferrero, E.A., 2010. In situ behavioural responses to boat noise exposure of Gobius cruentatus (Gmelin, 1789; fam.Gobiidae) and Chromis chromis (Linnaeus, 1758; fam. Pomacentridae) living in a Marine Protected Area. Journal of Experimental Marine Biology and Ecology 386 (1-2), 125–132.

Popper, A.N., Hastings, M.C., 2009a. The effects of anthropogenic sources of sound on fishes. Journal of Fish Biology 75 (3), 455–489.

Popper, A.N., Hastings, M.C., 2009b. The effects of human-generated sound on fish. Integrative Zoology 4 (1), 43–52.

Purser, J., Radford, A.N., 2011. Acoustic noise induces attention shifts and reduces foraging performance in three-spined sticklebacks (*Gasterosteus aculeatus*). Plos One 6 (2), e17478.

Radford, C., Jeffs, A., Tindle, C., Montgomery, J., 2008. Temporal patterns in ambient noise of biological origin from a shallow water temperate reef. Oecologia 156 (4), 921–929.

Richardson, W.J., Greene, C.R., Malme, C.I., Thompson, D.H., 1995. Marine Mammals and Noise. Academic Press, San Diego.

Richardson, W.J., Würsig, B., 1997. Influences of man-made noise and other human actions on cetacean behaviour. Marine and Freshwater Behaviour and Physiology 29 (1), 183–209.

Rolland, R.M., Parks, S.E., Hunt, K.E., Castellote, M., Corkeron, P.J., Nowacek, D.P., Wasser, S.K., Kraus, S.D., 2012. Evidence that ship noise increases stress in right whales. Proceedings of the Royal Society B: Biological Sciences 279 (1737), 2363–2368.

Slabbekoorn, H., Bouton, N., van Opzeeland, I., Coers, A., ten Cate, C., Popper, A.N., 2010. A noisy spring: the impact of globally rising underwater sound levels on fish. Trends in Ecology & Evolution 25 (7), 419–427.

Southall, B.L., Bowles, A.E., Ellison, W.T., Finneran, J.J., Gentry, R.L., Greene, Charles, R.J., Kastak, D., Ketten, D.R., Miller, J.H., Nachtigall, P.E., Richardson, W.J., Thomas, J.A., Tyack, P.L., 2007. Marine mammal noise exposure criteria: initial scientific recommendations. Aquatic Mammals 33 (4), 411–521.

Tasker, M., Amundin, M., André, M., Hawkins, A., Lang, W., Merck, T., Scholik-Schlomer, A., Teilmann, J., Thomsen, F., Werner, S., Zakharia, M., 2010. Marine Strategy Framework Directive – Task Group 11 Report Underwater noise and other forms of energy. EUR 24341 EN – Joint Research Centre, Luxembourg: Office for Official Publications of the European Communities, 55pp.

TNO, 2011. Standard for measurement and monitoring of underwater noise, Part I: physical quantities and their units. TNO Report TNO-DV 2011 C235. Tech. Rep., TNO, The Hague. 52pp.

Toropova, C., Meliane, I., Laffoley, D., Matthews, E., Spalding, M. (Eds.), 2010. Global Ocean Protection: Present Status and Future Possibilities. Springer, Gland Switzerland.

Trevorrow, M.V., Vasiliev, B., Vagle, S., 2008. Directionality and maneuvering effects on a surface ship underwater acoustic signature. Journal of the Acoustical Society of America 124 (2), 767–778.

Tyack, P.L., 2008. Implications for marine mammals of large-scale changes in the marine acoustic environment. Journal of Mammalogy 89 (3), 549–558.

Urick, R.J., 1983. Principles of Underwater Sound, 3rd ed. McGraw-Hill, New York.

Vasconcelos, R.O., Amorim, M.C.P., Ladich, F., 2007. Effects of ship noise on the detectability of communication signals in the Lusitanian toadfish. Journal of Experimental Biology 210 (12), 2104–2112.

Wales, S.C., Heitmeyer, R.M., 2002. An ensemble source spectra model for merchant ship-radiated noise. Journal of the Acoustical Society of America 111 (3), 1211–1231.

Wright, A.J., Deak, T., Parsons, E.C.M., 2011. Size matters: management of stress responses and chronic stress in beaked whales and other marine mammals may require larger exclusion zones. Marine Pollution Bulletin 63 (1-4), 5–9.

Wright, A.J., Soto, N.A., Baldwin, A.L., Bateson, M., Beale, C.M., Clark, C., Deak, T., Edwards, E.F., Fernández, A., Godinho, A., Hatch, L.T., Kakuschke, A., Lusseau, D., Martineau, D., Romero, M.L., Weilgart, L.S., Wintle, B.A., Notarbartolo-di Sciara, G., Martin, V., 2007. Do marine mammals experience stress related to anthropogenic noise? International Journal of Comparative Psychology 20 (2), 274–316.

Wysocki, L.E., Dittami, J.P., Ladich, F., 2006. Ship noise and cortisol secretion in European freshwater fishes. Biological Conservation 128 (4), 501–508.