Combined impacts of elevated CO_2 and anthropogenic noise on European sea bass (*Dicentrarchus labrax*)

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Ocean acidification (OA) and anthropogenic noise are both known to cause stress and induce physiological and behavioural changes in fish, with consequences for fitness. OA is also predicted to reduce the ocean's capacity to absorb low-frequency sounds produced by human activity. Consequently, anthropogenic noise could propagate further under an increasingly acidic ocean. For the first time, this study investigated the independent and combined impacts of elevated carbon dioxide (CO_2) and anthropogenic noise on the behaviour of a marine fish, the European sea bass (*Dicentrarchus labrax*). In a fully factorial experiment crossing two CO_2 levels (current day and elevated) with two noise conditions (ambient and pile driving), *D. labrax* were exposed to four CO_2/noise treatment combinations: 400 μatm/ambient, 1000 μatm/ambient, 400 μatm/pile-driving, and 1000 μatm/pile-driving. Pile-driving noise increased ventilation rate (indicating stress) compared with ambient noise conditions. Elevated CO_2 did not alter the ventilation rate response to noise. Furthermore, there was no interaction effect between elevated CO_2 and pile-driving noise, suggesting that OA is unlikely to influence startle or ventilatory responses of fish to anthropogenic noise. However, effective management of anthropogenic noise could reduce fish stress, which may improve resilience to future stressors.

Keywords: elevated CO_2, pile-driving, sensory behaviour, ventilation.

Introduction

Humans depend on the oceans for the goods and ecosystem services they provide, but with global population rise humans are increasingly altering the oceans, with impacts on marine organisms and ecosystems (Vitousek *et al.*, 1997; Halpern *et al.*, 2008). To date, most studies have focused on the effects of a single stressor, yet ecosystems are usually affected by multiple stressors (Breitburg *et al.*, 1998; Crain *et al.*, 2008; Halpern *et al.*, 2008; Ling *et al.*, 2009). This study investigated the combined impacts of two recently acknowledged environmental changes: ocean acidification (OA) and anthropogenic noise (IPCC, 2007; Slabbekoorn *et al.*, 2010; IGBP, IOC, SCOR, 2013).

OA results from atmospheric carbon dioxide (CO_2), primarily released during combustion of fossil fuels, dissolving into the ocean (Orr *et al.*, 2005; Denman *et al.*, 2011; IPCC, 2014). High CO_2 levels and reduced pH affect behaviour and physiology in a range of marine taxa including fish (Fabry *et al.*, 2008; Kroeker *et al.*, 2010; Hurst *et al.*, 2013). For example, clownfish larvae become highly attracted to predator odours (*Amphiprion percula*; Dixson *et al.*, 2010), while damselfish exhibit bolder behaviour (*Pomacentrus wardi*; Munday *et al.*, 2010) and lose typical lateralization behaviour (*Neopomacentrus azysron*; Domenici *et al.*, 2012). OA-induced disruptions to neurotransmitter function appear to underpin these behavioural changes (Nilsson *et al.*, 2012).

Anthropogenic noise in aquatic environments, including that produced by shipping, seismic surveys, and pile-driving, is also increasing (Bailey *et al.*, 2010; Slabbekoorn *et al.*, 2010; Normandeau Associates, Inc., 2012; Simpson *et al.*, 2015). Since this noise generally overlaps with the hearing ranges of marine taxa, a variety of effects have been documented (Slabbekoorn *et al.*, 2010; Simpson *et al.*, 2015). Increasing interest in offshore renewable energy has stimulated the rapid growth of offshore windfarms, with pile-driving noise being a dominant feature during construction (Gill, 2005; Bailey *et al.*, 2010; Normandeau Associates, Inc., 2012; Bruintjes

et al., 2014). Pile-driving sounds cause freezing reactions in cod (*Gadus morhua*), increases in swimming speed in sole (*Solea solea*; Mueller-Blenkle *et al.*, 2010) and greater oxygen consumption in European sea bass (*Dicentrarchus labrax*; Bruintjes *et al.*, 2014). However, research into its effects on fish remains limited and presents a current knowledge gap to policy-makers and marine managers (Mueller-Blenkle *et al.*, 2010).

OA is predicted to reduce the ocean's capacity to absorb low-frequency sounds produced from human activities (Ilyina *et al.*, 2010). Consequently, anthropogenic noise could propagate further under an increasingly acidic ocean. Additionally, fish otoliths, free-floating earbones in the back of the head that act as accelerometers and enable hearing, were considerably larger in white sea bass (*Atractoscion nobilis*) reared under elevated CO_2 conditions (Checkley *et al.*, 2009). Furthermore, clownfish (*Amphiprion percula*) that normally avoid daytime reef noise during the pelagic larval stage, did not respond to such noise if reared in elevated CO_2 conditions (Simpson *et al.*, 2011). An important question arising from these studies is whether OA influences fish responses to anthropogenic noise. No study has yet investigated the combined impact of OA and anthropogenic noise.

To address this knowledge gap, we investigated the independent and combined impacts of elevated CO_2 and pile-driving noise on the behaviour and physiology of the economically important European sea bass (*D. labrax*). In a crossed design, *D. labrax* were reared in control ($\sim$400 μatm) or elevated ($\sim$1000 μatm) CO_2 conditions then acutely exposed to ambient or pile-driving noise. Experimental CO_2 levels matched current ocean conditions and predictions made for the end of the century (IPCC, 2014). To establish whether impacts may carry direct fitness consequences, we assessed anti-predator responses using an established looming stimulus assay (Simpson *et al.*, 2015), and measured ventilation rate during acoustic exposure, providing an indication of stress (Simpson *et al.*, 2015).

Previous work that has found ventilation rate in *D. labrax* is considerably higher during playback of pile-driving noise (Bruintjes *et al.*, 2016), a result also seen in the European eel (*Anguilla anguilla*) during playback of ship noise (Simpson *et al.*, 2015), whereas chronic elevated CO_2 exposure previously had no effect on ventilation rate in eels (McKenzie *et al.*, 2003). Therefore, we hypothesized that pile-driving noise would increase ventilation rate, while elevated CO_2 would not. Exposure to high CO_2 levels can impair *D. labrax* sensory response to various smells (Porteus *et al.*, unpublished data), and pile-driving noise can impair *D. labrax* startle response to simulated predator strikes (Everley *et al.*, 2015). Therefore, we further hypothesized that elevated CO_2 or pile-driving noise in isolation, would decrease the number of fish that startle. Previous work investigating multi-stressor impacts have reported interacting effects (Rosa and Seibel, 2008; Munday *et al.*, 2009; Nowicki *et al.*, 2012). Thus, we also tested for an interaction between pile-driving noise and OA to examine whether responses differ when stressors are combined.

Material and methods
Ethics statement
All experimental procedures were approved by the University of Exeter Ethics Committee (2013/247), and were deemed by the Home Office as being below the level of severity that would require licensing, although appropriate licences were in place (PPL 30/2860).

Table 1. Water chemistry parameters for control and elevated CO_2 treatments.

Parameter	Control CO_2	Elevated CO_2
pH_{NBS}	8.12 ± 0.004	7.82 ± 0.003
Temperature (°C)	15.3 ± 0.02	15.3 ± 0.02
Salinity (ppt)	35.8 ± 0.04	35.9 ± 0.03
TA (μmol/kgSW)	2407.6 ± 9.1	2377.8 ± 10.2
pCO_2 (μatm)	471.7 ± 7.4	1032.0 ± 12.8

pH_{NBS}, temperature and salinity data are direct measurements. pCO_2 and TA were calculated from these parameters in CO2SYS. Data are presented as mean ± SE.

CO_2 treatments
Juvenile European sea bass (*D. labrax*; 4–8 g, 7 cm total length TL) were obtained from Ecloserie Marines de Gravelines, France and initially held within a 130 l plastic stock tank (76 × 56 × 30 cm) containing recirculating artificial seawater (15°C) in the University of Exeter Aquatic Resource Centre. Before testing, 48 fish were exposed to seawater at current day CO_2 ($\sim$400 μatm) and a further 48 to elevated CO_2 ($\sim$1000 μatm) levels for 20–21 d. Each CO_2 treatment consisted of recirculated artificial seawater held in 20 l containers (six fish per container) supplied with seawater at 15.3°C ± 0.02 (SE) at a flow of 1.7 l/min from one of two 100 l header tanks. Photoperiod was controlled (12:12 light:dark) and fish were fed once daily with 1% body mass of Perla MP pellets (Skretting, Shay Lane, Longridge, Preston, UK).

In a recirculating system (450 l), seawater diffused with ambient air scrubbed of CO_2 using Soda lime (Sigma; control), thus offsetting CO_2 produced by fish, or air enriched with CO_2 (elevated) was bubbled into containers. A pH Computer (Aqua Medic, Bissendorf, Germany), calibrated to a set pH and linked to a solenoid valve, controlled the delivery of CO_2 gas (BOC, England, UK) to the elevated CO_2 header tank to maintain a pH of 7.82 ± 0.02. Temperature, salinity, and pH_{NBS} were measured daily using a salinity, conductivity, and temperature system (YSI Model 30, YSI incorporated, Yellow Springs, Ohio, USA), pH meter (Model HI 8314, Hanna Instruments, Leighton Buzzard, UK) and pH probe (Model pHC2401, Radiometer Analytical, Lyon, France) calibrated with National Bureau of Standards (NBS) buffers. Ammonia was monitored weekly using a test kit (TetraTest Ammonia kit, Ark Pets, UK), with water changes every 7–10 d to avoid build up. Seawater was sampled from each container 2–3 times per week, poisoned with mercuric chloride ($HgCl_2$) according to standard methods and stored at 4°C (following Hurst *et al.*, 2013). Seawater samples were analysed for dissolved inorganic carbon (DIC) using a custom built system (described in Lewis *et al.*, 2013). This allowed a ± 3 μmol/kg precision for DIC measurements. Measured temperature, salinity, pH, and DIC values were used to calculate average seawater pCO_2 and total alkalinity (TA) using the CO2SYS software (http://cdiac.ornl.gov/oceans/co2rprt.html) using the GEOSECS constants (NBS scale; Takahashi *et al.*, 1982; Table 1). In preliminary experiments, seawater temperature, oxygen, and CO_2 levels in experimental tanks (see below) were measured over a 35-min period, ensuring no significant deviations in any of these parameters.

General experimental design
The experimental set up comprised two 3 l plastic experimental tanks (20 × 12 × 13 cm, >90% acoustically transparent) filled with water from the CO_2 exposure set up appropriate to each

test fish. Each 3 l tank floated within two larger 90 l glass tanks (54 × 44 × 38 cm) filled with standard seawater in a temperature controlled room (15.6 ± 1°C). Each large tank rested on 5 cm expanded polystyrene and six pieces of evenly distributed plastic pipe insulation to minimize transfer of vibrations from the building. An opaque divider was placed between the two tanks to avoid disruption of ongoing trials or visual interaction between fish during experiments. Recordings were initially made in both tanks to ensure there was no noise transfer between tanks (details of recording equipment below). Playback tracks were MP3 files played through a sound system (identical for both tanks) consisting of a battery (12 V 7.2 Ah sealed lead-acid), MP3 player (PCM-M10, Sony Corp., Japan), amplifier (M033N, 18W, frequency response 0.04–20 kHz; Kemo Electronic, Germany), and underwater speaker (University Sound UW-30; maximal output 156 dB re 1 μPa at 1 m, frequency response 0.1–10 kHz; Lubell Labs, USA). Speakers were positioned facing upwards to the floating experimental tanks and hidden under a false bottom of drilled Perspex.

A fully factorial experiment crossing two CO_2 levels (∼400 and ∼1000 μatm) with two acoustic conditions (playback of ambient and pile-driving recordings) was used (2 × 2 design). Twenty-four fish were tested in each of the four different CO_2/noise treatment combinations: 400 μatm/ambient (hereafter 400/amb), 1000 μatm/ambient (1000/amb), 400 μatm/pile-driving (400/pile), and 1000 μatm/pile-driving (1000/pile). Using an independent samples design to avoid carry over effects, each fish was tested once in each experiment. For each trial, an individual fish was transferred by scoop into an experimental tank with the relevant CO_2 treatment water. Similar to Simpson *et al.* (2015), each trial included 30 min ambient playback allowing fish to acclimatize, using one of three different ambient tracks, followed by a change to 5 min of another ambient or pile-driving track (see Playback tracks). The CO_2/noise treatment combinations were randomly allocated to each tank, controlling for any tank effect or bias due to the room layout. Treatment order was randomized and counterbalanced between days and in three blocks of four on each day, controlling for any daily variation and time of day effects. After each trial, the test fish was placed in a different holding tank and the water in experimental tanks was changed.

Playback tracks

Thirty-five minutes playback tracks were produced in Audacity 2.0.6 (http://audacity.sourceforge.net/) using sounds recorded at four UK ports: Gravesend (51°26′42″N, 0°22′37″E), Plymouth (50°21′33″N, 4°7′26″W), Portsmouth (50°47′21″N, 1°6′25″W), and Newcastle (55°7′46″N, 1°30′24″W) (further details in Wale *et al.*, 2013a, b; Voellmy *et al.*, 2014a, b; Simpson *et al.*, 2015). Field recordings of pile-driving noise, at ca. 120 m distance from the source (55°8′46″N, 1°25′15″W; Narec, Blyth, Newcastle), and ambient harbour noise were taken. Six different ambient noise files (3 × 30 min and 3 × 5 min) and three different pile-driving noise files (all 5 min) were made by looping the sound tracks together (following Voellmy *et al.*, 2014b). Recordings were taken in the centre of the 3 l experimental tanks, in the experimental set up described above, using a hydrophone (HiTech HTI-96-MIN with inbuilt preamplifier, sensitivity −165 dB re 1 V/μPa, frequency range 0.02–30 kHz; High Tech Inc., USA) and calibrated recorder (PCM-M10, 44.1 kHz sampling rate; Sony Corp., Japan).

Before the experiments, Avisoft SASLab Pro (v5.2.07; Avisoft Bioacoustics) was used to analyse the sound pressure level [SPL, root mean square (RMS), full spectra] and power spectra of

recordings when played back in experimental tanks (Figure 1a; following Simpson *et al.*, 2015). The SPL of the ambient tracks was averaged over 30 s recordings (similar to Bailey *et al.*, 2010) and SPL of pile-driving tracks calculated as the average peak pulse level for 21 piling strikes (pulse length = 100 ms, averaging window 10 ms) in a 30 s recording (similar to Everley *et al.*, 2015). The three ambient tracks were played at 127.2 ± 0.03 dB RMS re 1 μPa, while the peak pulse level of the three pile-driving tracks was 161.3 ± 0.05 dB RMS re 1 μPa in experimental tanks. A spherical model of sound propagation predicts that the pile-driving sound levels in the experimental tank were similar in the pressure domain to field conditions at 200 m from the source, while a cylindrical model predicts a distance of several kilometres, although local reflections and scattering would become a factor (Au and Hastings, 2008). These levels could affect a large number of fish in the wild, thus it is important to monitor impacts of noise levels corresponding to these larger spatial scales (Simpson *et al.*, 2015).

Ventilation rate

Fish were tested one at a time. Following Simpson *et al.* (2015), an observer hidden from the fish by a screen measured ventilation rate (opercular beat rate) through a small hole using a counter and stopwatch. The number of opercular beats in 1 min was counted at 27 min from the start of the trial (during ambient noise) then following the change to either another ambient or a pile-driving track. These measurements were used to calculate change in ventilation rate. If opercular beats were not visible, counting was stopped and resumed when they could be seen again, ensuring 1 min of beats was measured for each individual in each test period. Some ventilation measurements were double checked against the video recordings to make sure there was no observer bias and no discrepancies were found.

Startle response

A predator attack was simulated by a looming stimulus (following Simpson *et al.*, 2015). The stimulus was a black squash ball fixed to the end of a swinging transparent Perspex pendulum arm. Fishing line, attached to the pendulum arm, positioned the stimulus at a 45° angle beyond one end of each tank. Once released the squash ball ended up next to the glass but without making contact, due to a foam stopper positioned near to the hinge of the swinging arm on the ceiling. The stimulus was manually released by an observer (through releasing the taut fishing line hooked around a clamp) at 3 min after the track change, thereby standardizing trials. Behaviour was recorded from 27 min until the trial ended using a video camera (Sony HDR-XR155, 25 frames per second) supported by a tripod and pointing through a hole in the screen. A startle was only scored if it was elicited during the first advance of the ball towards the tank. The trial number was visible in the film allowing subsequent subject identification. Videos were analysed without sound (avoiding unintentional observer effects) fish were scored (one per trial) on whether they startled or not in response to the looming stimulus (startle response).

A few trials (1 or 2 per treatment) were omitted from statistical analysis due to problems during the trial, such as accidental stimulus release. Consequently, sample sizes differed slightly between treatment combinations and variables measured.

Data analysis

Statistical analyses were conducted in SigmaPlot (version 10.0) and SPSS (IBM SPSS Statistics, IBM Corporation, Version 22).
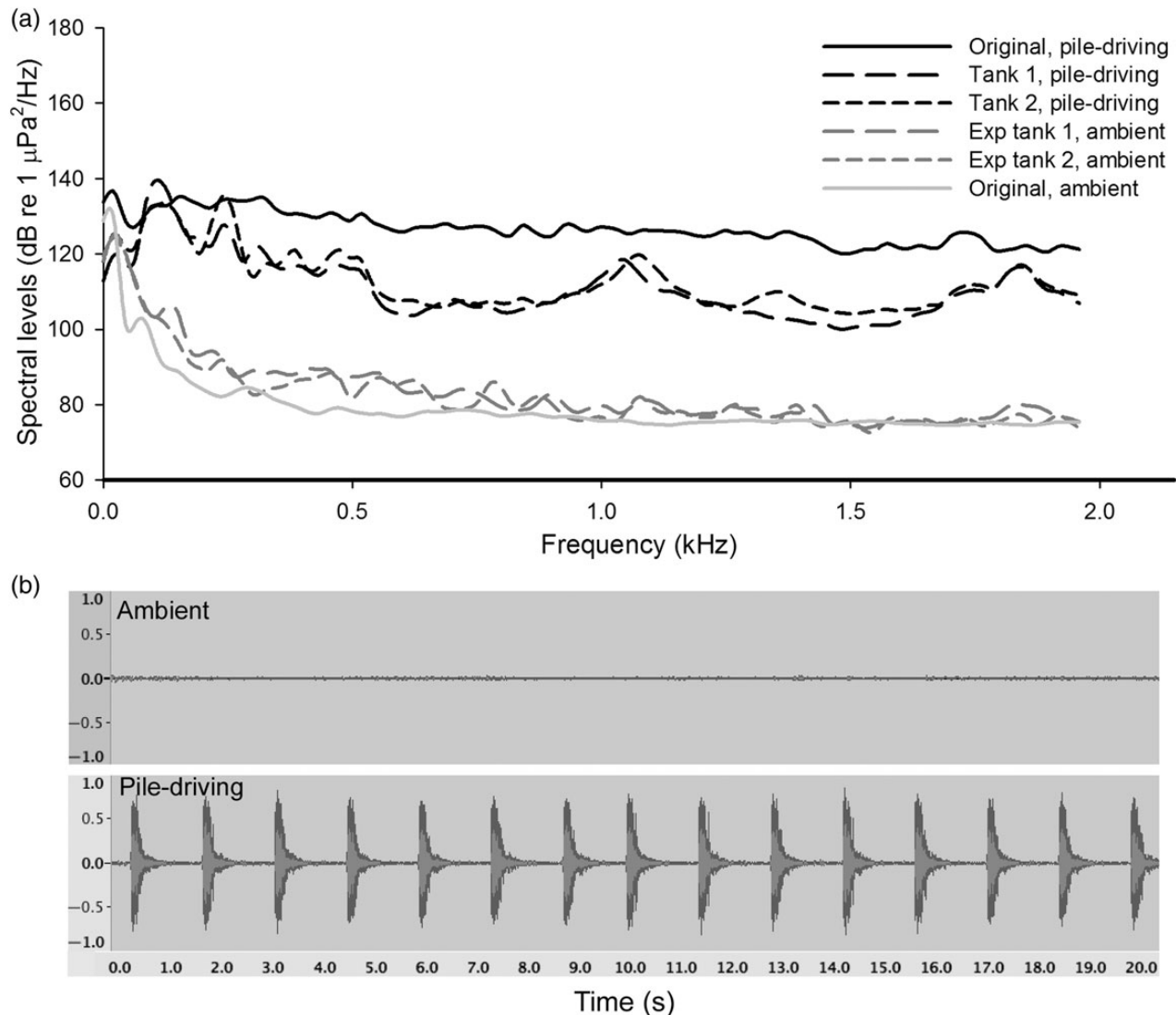
Figure 1. (a) Power spectra for recordings taken in the field and experimental tanks. Analysis conducted in Avisoft SASLab Pro version 5.2.08 (Avisoft Bioacoustics, Berlin, Germany): Hann evaluation window, Fast Fourier Transform size 2048, averaged from a 30 s section from each recording. One recording from each noise treatment is shown for comparison. (b) Example of waveforms of ambient and pile-driving playbacks, 20 s long.

A two-way ANOVA was used to test for differences between CO_2 and noise treatments on the change in ventilation rate before and after the track change. A χ^2 test was used to identify differences in the number of fish that startled across treatment combinations.

## Results

Ventilation rate

There was a significant difference in the change in ventilation rate after the track change between noise treatments (two-way repeated measures ANOVA, $F_{1,\ 88} = 54.854$, n_{amb}, $n = 46$, $p < 0.001$; Figure 2a) but not between CO_2 conditions (two-way repeated measures ANOVA, $F_{1,\ 88} = 1.342$, $n = 46$, $p = 0.250$; Figure 2a). Thus, there was no interaction. Specifically, ventilation rate increased by 9 beats min^{-1} (6%) when an ambient track was switched for a pile-driving track in control CO_2 conditions (Figure 2a).

Startle response

The number of fish producing a startle response was not significantly different between the CO_2/noise treatment combinations (χ^2 test, $\chi^2 = 3.892$, $n = 23$, $p = 0.273$), despite 67% more fish startling when exposed to 1000 μatm/pile-driving noise compared with 400 μatm/ambient noise (Figure 2b).

Discussion

In this study, fish ventilation rate significantly increased during playback of pile-driving noise when compared with ventilation rate during ambient noise, supporting the hypothesis that pile-driving noise increases stress. This is consistent with elevated ventilation rate in European eels (Simpson *et al.*, 2015) and greater cortisol levels in three fish species during exposure to playback of ship noise (Wysocki *et al.*, 2006). Furthermore, the ventilation rate of fish exposed to elevated CO_2 was not significantly different to that of control fish, indicating that elevated CO_2 does not have an
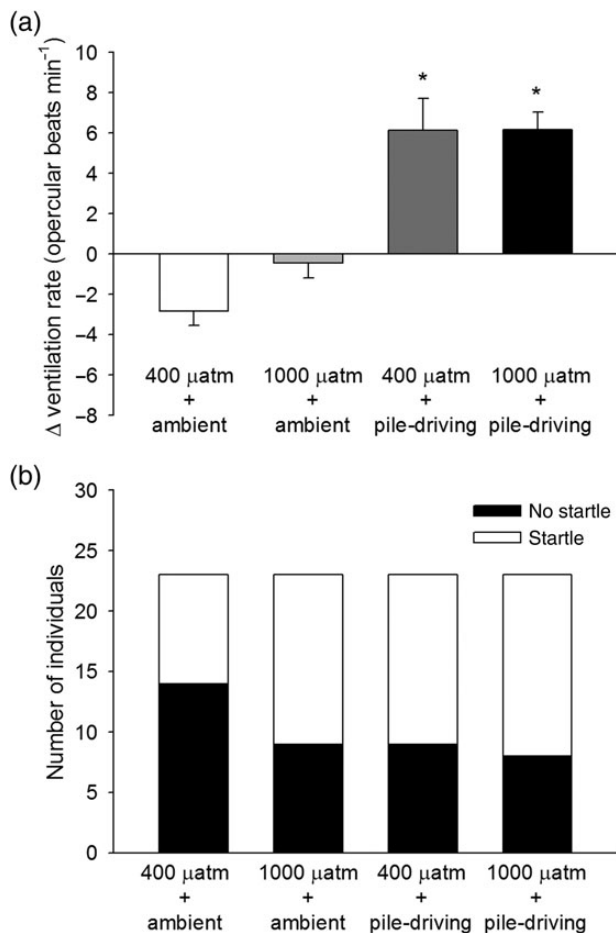
Figure 2. The effect of CO_2 and noise treatments on ventilation rate, and startle response of European sea bass (*D. labrax*). (a) Change in ventilation rate before and after the track change; $n = 23$ for each treatment combination. (b) Number of fish startling or not startling in response to a looming stimulus; $n = 23$ for each treatment combination. Data are presented as mean $\pm$ SE and asterisks above the bars denote significant differences from controls.

additional impact on the ventilatory response of sea bass. This study used acute noise exposures, but long-term stress can negatively impact reproduction and growth (Kight and Swaddle, 2011; Nedelec *et al.*, 2014). Furthermore, behavioural habituation of European sea bass can happen within an hour of exposure to pile-driving noise (Neo *et al.*, 2014). Further work should examine whether elevated CO_2 can affect the habituation of marine fish to noise and how repetitive or chronic exposure to anthropogenic noise affect fish behaviour.

Some fish respond to a predator by freezing; however, with our low-resolution video frame rates (25 fps), we were not able to quantify this response and thus a potential freeze response could not be analysed. Despite this, more fish tended to startle in elevated CO_2 than control CO_2 but this was also not statistically significant. Previous work reported that high CO_2 exposure caused changes to the functioning of gamma-aminobutyric acid type A neurotransmitter receptors, leading to altered behaviours (Nilsson *et al.*, 2012; Chivers *et al.*, 2014; Hamilton *et al.*, 2014; Spady *et al.*, 2014), such as the reversal of olfactory preferences in clownfish (*Amphiprion percula*) larvae (Nilsson *et al.*, 2012). Our initial findings, that warrant additional

investigation, suggest that elevated CO_2 has the potential to increase fish responsiveness to predators, potentially improving their chances of escape through reacting sooner to a threat stimulus. Guppies that reacted earlier to a predator model have a reduced predation risk (Krause and Godin, 1996; Voellmy *et al.*, 2014b). The results suggest that OA could have positive effects on the anti-predator behaviour of *D. labrax*. Future research could determine whether the observed effects are temporary or sustained in the long term (i.e. whole lifespan) and if other processes, such as growth, are compromised. These findings contrast with previous OA studies, demonstrating negative impacts on anti-predator behaviour in damselfish (*Pomacentrus chrysurus*; Ferrari *et al.*, 2011) and conch snails (*Gibberulus gibberulus gibbosus*; Watson *et al.*, 2014).

There was no interaction effect between elevated CO_2 and pile-driving noise on anti-predator behaviour or ventilation rate. This may be because fish that had been exposed to noise had potentially reached a ceiling effect, as there were no significant differences in anti-predator responses or ventilation rate between CO_2 treatments following the addition of pile-driving noise (Hamilton *et al.*, 2014). This is the first study to show that OA is unlikely to influence the anti-predator behaviour and physiological responses of fish to anthropogenic noise. Pile-driving noise increased fish stress, with possible fitness consequences, indicating that effective management of anthropogenic noise will likely reduce stress, especially as oceans further acidify (Denman *et al.*, 2011; Brown *et al.*, 2013; IPCC, 2014; Simpson *et al.*, 2015). However, the value of reducing a present-day stressor to improve the resilience of animals to future stressors should be explored (Pandolfi *et al.*, 2005; Barber *et al.*, 2010; Brown *et al.*, 2013; Simpson *et al.*, 2015). As a recommendation, this question should be considered in policy and management decisions moving further into the 21st century.

Acknowledgements

References

Au, W. W. L., and Hastings, M. C. 2008. Principles of Marine Bioacoustics. Springer, New York, USA. 696 pp.

Bailey, H., Senior, B., Simmons, D., Rusin, J., Picken, G., and 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: 888–897.

Barber, J. R., Crooks, K. R., and Fristrup, K. M. 2010. The costs of chronic noise exposure for terrestrial organisms. Trends in Ecology and Evolution, 25: 180–189.

Breitburg, D. L., Baxter, J. W., Hatfield, C. A., Howarth, R. W., Jones, C. G., Lovett, G. M., and Wigand, C. 1998. Understanding effects of multiple stressors: ideas and challenges. In Successes, Limitations, and Frontiers in Ecosystem Science, pp. 416–431. Ed. by M. L. Pace, and P. M. Groffman. Springer, New York, USA.

Brown, C. J., Saunders, M. I., Possingham, H. P., and Richardson, A. J. 2013. Managing for Interactions between local and global stressors of ecosystems. PLoS ONE, 8: 1–10.

Bruintjes, R., Armstrong-Smith, E., Botterell, Z., Renshaw, E., Tozer, B., Benson, T., and Rossington, K., *et al.* 2014. A tool to predict the

impact of anthropogenic noise on fish. *In* Proceedings of the 2nd International Conference on Environmental Interactions of Marine Renewable Energy Technologies (EIMR2014), [Online]. pp. 1–2. http://tethys.pnnl.gov/sites/default/files/attachments/EIMR2014_586_Bruintjes.pdf (last accessed 1 March 2015).

Bruintjes, R., Purser, J., Everley, K. A., Mangan, S., Simpson, S. D., and Radford, A. N. 2016. Rapid recovery following short-term acoustic disturbance in two fish species. Royal Society Open Science, doi: 10.1098/rsos.150686.

Checkley, D. M., Dickson, A. G., Takahashi, M., Radich, J. A., Eisenkolb, N., and Asch, R. 2009. Elevated CO_2 enhances otolith growth in young fish. Science, 324: 1683.

Chivers, D. P., McCormick, M. I., Nilsson, G. E., Munday, P. L., Watson, S-E., Meekan, M. G., Mitchell, M. D., *et al.* 2014. Impaired learning of predators and lower prey survival under elevated CO_2: a consequence of neurotransmitter interference. Global Change Biology, 20: 515–522.

Crain, C. M., Kroeker, K., and Halpern, B. S. 2008. Interactive and cumulative effects of multiple human stressors in marine systems. Ecology Letters, 11: 1304–1315.

Denman, K., Christian, J. R., Steiner, N., Portner, H-O., and Nojiri, Y. 2011. Potential impacts of future ocean acidification on marine ecoystems and fisheries: current knowledge and recommendations for future research. ICES Journal of Marine Science, 68: 1019–1029.

Dixson, D. L., Munday, P. L., and Jones, G. P. 2010. Ocean acidification disrupts the innate ability of fish to detect predator olfactory cues. Ecology Letters, 13: 68–75.

Domenici, P., Allan, B., McCormick, M. I., and Munday, P. L. 2012. Elevated carbon dioxide affects behavioural lateralization in a coral reef fish. Biology Letters, 8: 78–81.

Everley, K. A., Radford, A. N., and Simpson, S. D. 2015. Pile-driving noise impairs anti-predator behavior of the European sea bass *Dicentrarchus labrax*. In The Effects of Noise on Aquatic Life II. Advances in Experimental Medicine and Biology, 875, pp. 273–279. Ed. by A. N. Popper, and A. D. Hawkins. Springer Science and Business Media, New York.

Fabry, V. J., Seibel, B. A., Feely, R. A., and Orr, J. C. 2008. Impacts of ocean acidification on marine fauna and ecosystem processes. ICES Journal of Marine Science, 65: 414–432.

Ferrari, M. C. O., Dixson, D. L., Munday, P. L., McCormick, M. I., Meekan, M. G., Sih, A., and Chivers, D. P. 2011. Intrageneric variation in antipredator responses of coral reef fishes affected by ocean acidification: implications for climate change projections on marine communities. Global Change Biology, 17: 2980–2986.

Gill, A. B. 2005. Offshore renewable energy − ecological implications of generating electricity in the coastal zone. Journal of Applied Ecology, 42(4): 605–615.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., *et al.* 2008. A global map of human impact on marine ecosystems. Science, 319: 948–952.

Hamilton, T. J., Holcombe, A., and Tresguerres, M. 2014. CO_2-induced ocean acidification increases anxiety in Rockfish via alteration of $GABA_A$ receptor functioning. Proceedings of the Royal Society B: Biological Sciences, 281: 1–7.

Hurst, T. P., Fernandez, E. R., and Mathis, J. T. 2013. Effects of ocean acidification on hatch size and larval growth of walleye pollock (*Theragra chalcogramma*). ICES Journal of Marine Science, 70: 812–822.

IGBP, IOC, SCOR. 2013. Ocean Acidification Summary for Policymakers. *In* Third Symposium on the Ocean in a High-CO2 World. International Geosphere-Biosphere Programme, Stockholm, Sweden. 22 pp.

Ilyina, T., Zeebe, R. E., and Brewer, P. G. 2010. Future ocean increasingly transparent to low-frequency sound owing to carbon dioxide emissions. Nature Geoscience, 3: 18–22.

IPCC. 2007. Executive summary. *In* Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II

to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Ed. by M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson. Cambridge University Press, Cambridge, UK. 976 pp.

IPCC. 2014. Climate Change 2014: Synthesis Report. *In* Summary for Policymakers. Ed. by R. K. Pachauri, and L. Meyer, *et al.* 35 pp. http://www.ipcc.ch/pdf/assessment-report/ar5/syr/SYR_AR5_SPMcorr2.pdf (last accessed 2 March 2015).

Kight, C. R., and Swaddle, J. P. 2011. How and why environmental noise impacts animals: an integrative, mechanistic review. Ecology Letters, 14: 1052–1061.

Krause, J., and Godin, J-G. J. 1996. Influence of prey foraging posture on flight behavior and predation risk: predators take advantage of unwary prey. Behavioural Ecology, 7(3): 264–271.

Kroeker, K. J., Kordas, R. L., Crim, R. N., and Singh, G. G. 2010. Meta-analysis reveals negative yet variable effects of ocean acidification on marine organisms. Ecology Letters, 13: 1419–1434.

Lewis, C., Clemow, K., and Holt, W. V. 2013. Metal contamination increases the sensitivity of larvae but not gametes to ocean acidification in the polychaete *Pomatoceros lamarckii* (Quatrefages). Marine Biology, 160: 2089–2101.

Ling, S. D., Johnson, C. R., Frusher, S. D., and Ridgway, K. R. 2009. Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift. Proceedings of the National Academy of Sciences of the United States of America, 106: 22341–22345.

McKenzie, D. J., Piccolella, M., Dalla Valle, A. Z., Taylor, E. W., Bolis, C. L., and Steffensen, J. F. 2003. Tolerance of chronic hypercapnia by the European eel *Anguilla anguilla*. The Journal of Experimental Biology, 206: 1717–1726.

Mueller-Blenkle, C., McGregor, P. K., Gill, A. B., Andersson, M. H., Metcalfe, J., Bendall, V., Sigray, P., *et al.* 2010. Effects of pile-driving noise on the behaviour of marine fish. COWRIE Ref: Fish 06-08, Technical Report 31st March 2010, [Online]. pp. 1–57. http://www.thecrownestate.co.uk/media/5967/2010-03%20Effects%20of%20pile%20driving%20noise%20on%20fish.pdf last accessed 1 March 2015).

Munday, P. L., Crawley, N. E., and Nilsson, G. E. 2009. Interacting effects of elevated temperature and ocean acidification on the aerobic performance of coral reef fishes. Marine Ecology Progress Series, 388: 235–242.

Munday, P. L., Dixson, D. L., McCormick, M. I., Meekan, M., Ferrari, M. C. O., and Chivers, D. P. 2010. Replenishment of fish populations is threatened by ocean acidification. Proceedings of the National Academy of Sciences of the United States of America, 107: 12930–12934.

Nedelec, S. L., Radford, A. N., Simpson, S. D., Nedelec, B., Lecchini, D., and Mills, S. C. 2014. Anthropogenic noise playback impairs embryonic development and increases mortality in a marine invertebrate. Scientific Reports, 4: 1–4.

Neo, Y. Y., Seitz, J., Kastelein, R. A., Winter, H. V., ten Cate, C., and Slabbekoorn, H. 2014. Temporal structure of sound affects behavioural recovery from noise impact in European seabass. Biological Conservation, 178: 65–73.

Nilsson, G. E., Dixson, D. L., Domenici, P., McCormick, M. I., Sorensen, C., Watson, S-A., and Munday, P. L. 2012. Near-future carbon dioxide levels alter fish behaviour by interfering with neurotransmitter function. Nature Climate Change, 2: 201–204.

Normandeau Associates, Inc. 2012. Effects of noise on fish, fisheries, and invertebrates in the U.S. Atlantic and Arctic from energy industry sound-generating activities. A Literature Synthesis for the U.S. Dept. of the Interior, Bureau of Ocean Energy Management. Contract # M11PC00031, 153 pp. http://www.data.boem.gov/homepg/data_center/other/espis/espismaster.asp?appid=1 (last accessed 7 July 2015).

Nowicki, J. P., Miller, G. M., and Munday, P. L. 2012. Interactive effects of elevated temperature and CO_2 on foraging behavior of juvenile coral reef fish. Journal of Experimental Marine Biology and Ecology, 412: 46–51.

Orr, J. C., Fabry, V. J., Aumount, O., Bopp, L., Doney, S. C., Feely, R. A., Gnanadesikan, A., *et al.* 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature, 437: 681–686.

Pandolfi, J. M., Jackson, J. B. C., Baron, N., Bradbury, R. H., Guzman, H. M., Hughes, T. P., Kappel, C. V., *et al.* 2005. Are U.S. coral reefs on the slippery slope to slime? Science, 307: 1725–1726.

Rosa, R., and Seibel, B. A. 2008. Synergistic effects of climate-related variables suggest future physiological impairment in a top oceanic predator. Proceedings of the National Academy of Sciences of the United States of America, 105: 20776–20780.

Simpson, S. D., Munday, P. L., Wittenrich, M. L., Manassa, R., Dixson, D. L., Gagliano, M., and Yan, H. Y. 2011. Ocean acidification erodes crucial auditory behaviour in a marine fish. Biology Letters, 7: 917–920.

Simpson, S. D., Purser, J., and Radford, A. N. 2015. Anthropogenic noise compromises antipredator behaviour in European eels. Global Change Biology, 21: 586–593.

Slabbekoorn, H., Bouton, N., Opzeeland, I. V., Coers, A., Cate, C. T., and Popper, A. N. 2010. A noisy spring: the impact of globally rising underwater sound levels on fish. Trends in Ecology and Evolution, 25: 419–427.

Spady, B. L., Watson, S-A., Chase, T. J., and Munday, P. L. 2014. Projected near-future CO₂ levels increase activity and alter defensive behaviours in the tropical squid *Idiosepius pygmaeus*. Biology Open, 3: 1063–1070.

Takahashi, T., Williams, R. T., and Bos, D. L. 1982. Carbonate chemistry. *In* GEOSECS Pacific Expedition, Vol. 3, Hydrographic Data, pp. 5–10, 77–82. Ed. by W. S. Broecker, D. W. Spencer, and H. Craig. National Science Foundation, Washington, DC.

Vitousek, P. M., Mooney, H. A., Lubchenco, J., and Melillo, J. M. 1997. Human domination of earth's ecosystems. Science, 277: 494–499.

Voellmy, I. K., Purser, J., Flynn, D., Kennedy, P., Simpson, S. D., and Radford, A. N. 2014a. Acoustic noise reduces foraging success in two sympatric fish species via different mechanisms. Animal Behaviour, 89: 191–198.

Voellmy, I. K., Purser, J., Simpson, S. D., and Radford, A. N. 2014b. Increased noise levels have different impacts on the anti-predator behaviour of two sympatric fish species. PLoS ONE, 9: 1–8.

Wale, M. A., Simpson, S. D., and Radford, A. N. 2013a. Size-dependent physiological responses of shore crabs to single and repeated playback of ship noise. Biology Letters, 9: 1–4.

Wale, M. A., Simpson, S. D., and Radford, A. N. 2013b. Noise negatively affects foraging and antipredator behaviour in shore crabs. Animal Behaviour, 86: 111–118.

Watson, S-A., Lefevre, S., McCormick, M. I., Domenici, P., Nilsson, G. E., and Munday, P. L. 2014. Marine mollusc predator-escape behaviour altered by near-future carbon dioxide levels. Proceedings of the Royal Society B: Biological Sciences, 281: 1–9.

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

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