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# First indications for reduced mortality of non-target invertebrate benthic megafauna after pulse beam trawling

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Two alternative stimulation techniques to reduce mortality in benthic megafauna were tested relative to standard tickler chain beam trawling: longitudinal electrodes (pulse trawl) and longitudinal chains. Longitudinal chains caused higher mortality than pulse trawling in 3 species. Standard trawling gave higher mortality in Echinocardium cordatum than pulse trawling. Between longitudinal chain and standard trawling were no significant differences. This trend in decreasing mortality from longitudinal, to standard and then pulse trawling was confirmed by a similar decline in: i) numbers of significant species mortalities per trawl type, ii) average mortalities, i.e. longitudinal chain caused 41% more mortality than standard trawling and pulse trawling 43% less, iii) pre- and post-trawling community dissimilarities. A significant majority of species showed higher mortalities after longitudinal than after standard trawling and, conversely, lower mortalities after pulse trawling. Trawls with longitudinal chains instead of cross tickler chains increase megafaunal impact. On the contrary, pulse trawling can reduce the impact, although average mortality remains substantial (25%) even in impoverished benthic test assemblages in the southern North Sea. Power, generally was low and reference areas, free of (pulse) trawling, and inhabited by more vulnerable taxa will facilitate higher powered studies on the impact of standard and alternative trawling techniques.

**Keywords:** alternative beam trawls, benthic megafauna, longitudinal chain beam trawl, pulse beam trawl, trawling mortality, Triple-D benthos dredge

## Introduction

In the southern North Sea, bottom trawls have been used for centuries until the tickler chain beam trawl became the dominant flatfish gear in the 1960s. Graham (1955) reported damaged non-target invertebrate specimens in trawl nets, and Groenewold and Fonds (2000) quantified this fraction as up to 60% of caught phyla. Similar percentages of mortality were measured in some infauna in the trawl path (Rees and Eleftheriou, 1989; Bergman and Hup, 1992), while Bergman and van Santbrink (2000) reported 7–39% annual mortality among longer-lived benthic populations in the Dutch sector. Latter authors stated that most of the casualties did not die while trapped in the net but in the trawl path due to physical damage or successive predation. Long-term trawling consequences were reduced species richness, evenness, and abundances of burrowing shrimps and bivalves when compared to a fishery exclusion zone (Duineveld *et al.*, 2007). Also, bivalve length was negatively correlated with trawling frequency (van Kooten *et al.*, 2015). Declines in suspension feeders (de Juan *et al.*, 2007), epifauna (Buhl-Mortensen *et al.*, 2016), annual faunal production (Hermsen *et al.*, 2003), biomass, species richness, species diversity, and habitat complexity (Collie *et al.*, 1997; Thrush and Dayton, 2002; Reiss *et al.*, 2009) have likewise been attributed to trawling.

Since 2000, various approaches have been implemented to reduce human impact on the North Sea ecosystem. NATURA 2000 and the Marine Strategy Framework Directive (MSFD; EC, 2008) require North Sea countries to improve marine biodiversity and to safeguard Good Environmental Status (GES) in 2020 (EU, 2010). National and EU rules seek to preserve specific benthic habitats from, e.g. trawling impact by the designation of marine protected areas (Anon, 2012). The Common Fisheries Policy (CFP, 2014) aims to make fishing fleets more selective to protect fragile ecosystem components. This context fits the quest for beam trawls with less destructive stimulation techniques than the traditional tickler chains.

The REDUCE project (1998–2001) categorized alternative stimulation techniques to replace the cross (perpendicular to towing direction) tickler chains, which could be less destructive for invertebrate bottom fauna (Linnane *et al.*, 2000) while warranting catch efficiencies for fish (Anon, 2002). The following two stimulation systems, fitted between beam and ground rope, were selected for further examination: (i) longitudinal electrical drag wires triggering fish to cramp and be scooped up, the "pulse trawl", and (ii) longitudinal parallel chains to chase fish off the seabed. Both alternatives would supposedly reduce seabed disturbance. In field experiments in 2000, the short-term mortality in benthic megafauna (>1 cm) induced by these two alternative trawls was compared to standard tickler chain beam trawling. Although some results were reported earlier (van Marlen *et al.*, 2001, 2005; Anon, 2002), this unique data set has not been fully analysed. Since more recent field comparisons between pulse and standard trawling have remained inconclusive (e.g. Teal *et al.*, 2014), we decided to analyse the REDUCE data in more detail.

The data set is still highly relevant as between 2009 and 2014 the majority of the Dutch flatfish trawl fleet (74 cutters) became EU certified to use pulse trawls up to 2020 (Rijnsdorp *et al.*, 2016). This certification was for up to 5% of any Member State's fleet (EC, 1998), and the surplus was granted only to the Dutch government as experimental licences. The pulse trawl originated already from the 1960s (de Groot and Boonstra, 1970), was refined during the energy crisis in the seventies and vetoed by the German and Dutch governments in 1987/1988 (Soetaert *et al.*, 2015). Pulse trawling appeared to have several advantages: 16–24% less discarded fish, up to 62% less discarded benthos, higher quality marketable fish, and reductions of 25–46% in fuel consumption and $CO_2$ emission per kilogram of fish (Taal and Klok, 2014; Quirijns *et al.*, 2015; ICES, 2018). In early 2019, the EU parliament decided to forbid any pulse trawling after July 2021. Up to then the experimental licences in the Dutch pulse fleet will be gradually phased out, and only a few pulse trawlers will be involved in well-defined scientific studies.

In the present study, we document our re-analysed results of the REDUCE field test. Benthic megafauna was chosen as test category since it is a relevant benthic fraction and described as contributing significantly to "sea-floor integrity", which is denoted as Descriptor 6 in a series of qualitative descriptors defining GES of marine waters in the MSFD (EU, 2010). Megafauna has also proven to be a category vulnerable to trawling (Duineveld *et al.*, 2007; Bergman *et al.*, 2015). Quantitative abundance estimates were made by means of a pneumatic version of the original Triple-D benthos dredge (Bergman and van Santbrink, 1994; NIOZ, 2012). Differences in trawling mortality in megafauna exerted by alternative and standard trawls are discussed in the context of North Sea fisheries.

## Material and methods

### Test area

The test site with the experimental strips was situated in the Oyster Ground (southern North Sea, Figure 1; 54°17′N and 004°58′E) and selected for its relatively high abundance of megafaunal species (e.g. bivalves, echinoderms, crabs). Pre-trawling side-scan sonar recordings in the area displayed numerous trawl tracks over a seabed with the north-bound small sand ripples of

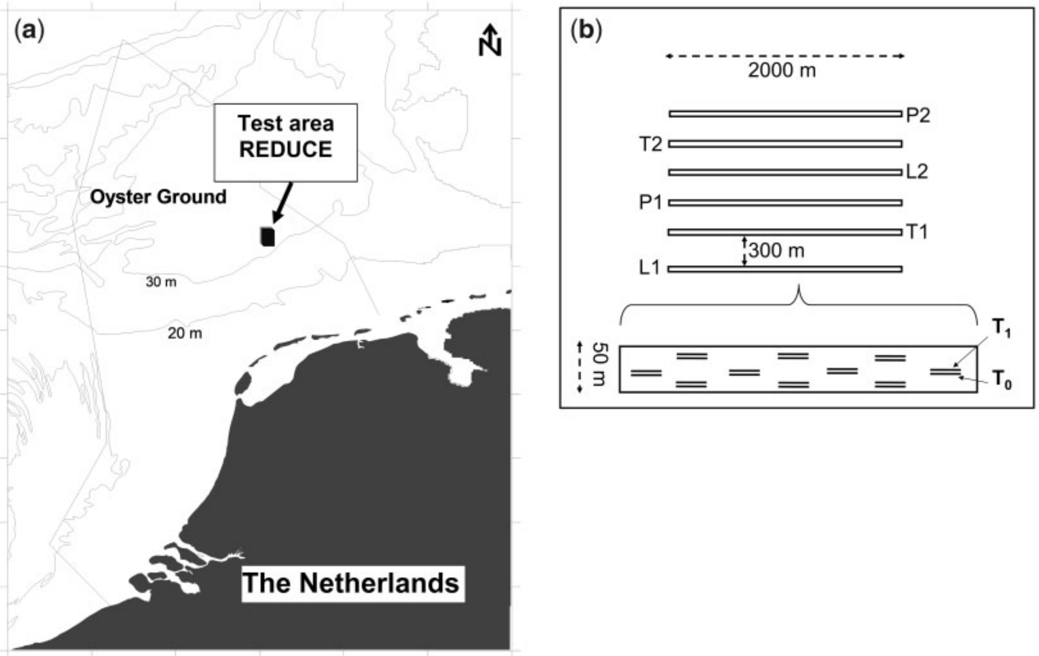

**Figure 1.** Field experiment in the Oyster Ground (May–June 2000): (a) test area and (b) experimental design showing three pairs of replicate strips trawled with longitudinal chain beam trawl (L), standard tickler chain beam trawl (S), and pulse beam trawl (P). In detail, ten Triple-D transects in a strip before ($T_0$) and after trawling ($T_1$).

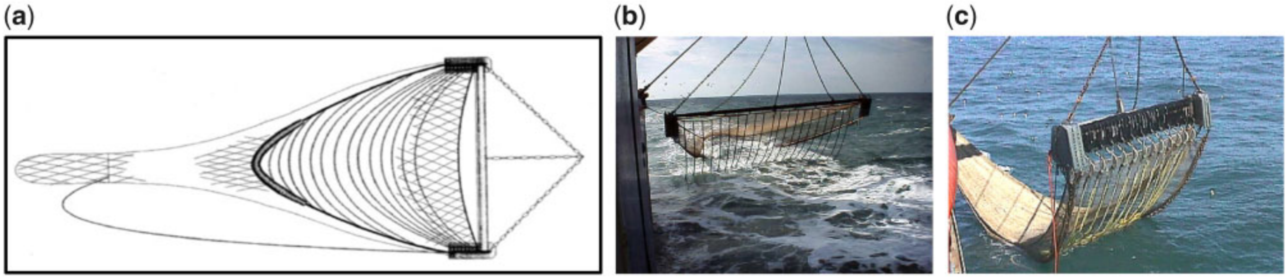

**Figure 2.** (a) An outline of the standard 7-m tickler chain beam trawl. Note that the experimental trawl had nine tickler chains instead of seven. (b) An example of a 12-m beam trawl with 21 longitudinal chains (van Marlen *et al.*, 2005); the tested trawl was 7-m wide with longitudinal 13 chains. (c) The tested 7-m pulse beam trawl (van Marlen *et al.*, 2001).

<15-cm height (wave length 20–25 m). The muddy sediment (on average 20.6% <63 µm, *SD* = 5.2) had an average median grain size of 135.9 µm (*SD* = 7.7; *n* = 5). Water depth ranged from 43.5 to 45.0 m.

## Standard and alternative trawls

All trawls used in the field test were supplied by the former Netherlands Institute for Fisheries Research (RIVO), nowadays Wageningen Marine Research. The trawls had a width of 7 m, while most commercial flatfish trawls are 4 or 10–12-m wide. The standard tickler chain beam trawl was rigged with nine cross running tickler chains and seven net ticklers in front of the net opening. The longitudinal chain trawl was rigged with 13 parallel longitudinal chains between beam and ground rope (Figure 2a and b). Trawls and nets were designed as similar as possible (Table 1). Due to the design of the stimulation systems, the final weights of the experimental trawls differed. Alike commercial trawls, the pulse trawl had less weight than the standard trawl. The longitudinal chain trawl had an intermediate weight. To reflect commercial operating procedures, the towing speed was 4 knots for the pulse trawl and 5 knots for the other trawls. Nets (meshes 8 cm stretched) are described in van Marlen *et al.* (2001).

Verburg Holland Ltd used the opportunity given by the European Union (EC, 1998) to test an innovative prototype 7-m pulse trawl in the late nineties (van Marlen *et al.*, 2001). This prototype was also used in the REDUCE field experiment, and for a comparison, all three types of beam trawls had to be 7-m wide. The pulse trawl had 13 longitudinal arrays of wired electrodes between beam and ground rope (Figure 2c). How the electrical design exactly relates to the current commercial pulse trawls is not fully clear, since its specifications were kept secret by Verburg (D. de Haan, pers. comm.). We assume, however, that it would fit within the present technical limitations documented in national and EU directives as described in Rijnsdorp *et al.* (2016). In short, the wired electrodes in pulse trawls consist of copper conductors alternated with isolators and conduct electric current generally during 2.5% of the time (50 V, 38–80 Hz, pulse width 100–270 µs). The field strength decreases exponentially with distance (de Haan *et al.*, 2016), and the shape of the electric field relates not only to type, number and distance between electrodes and isolators but also to pulse shape (Quirijns *et al.*, 2015). Since temperature and salinity affect conductivity and thus fish responses (de Haan *et al.*, 2011), electrical parameters can be adjusted on board, also during the haul. Technical innovations and changes in electrode configurations, however, have led to continuously changing pulse parameter settings used in the field (ICES, 2018). Recently, Soetaert *et al.* (2019) presented a guideline to the terminology of electrical characteristics to facilitate the discussion of electrofishing studies.

**Table 1.** Particulars of the standard and alternative trawl types in the field test (van Marlen *et al.*, 2001; Anon 2002).

| Specifics | Longitudinal | Standard | Pulse |
|---|---|---|---|
| Width (m) | 7 | 7 | 7 |
| Weight in air without net (kg) | 3 000 | 3 700 | 2 600 |
| Towing speed (knots) | 5 | 5 | 4 |
| Ticklers chains cross (*n*) | | 9 | |
| Diameter (mm) | | 18 | |
| Weight incl. net ticklers (kg) | | ~1 200 | |
| Net tickler chains cross (*n*) | | 7 | |
| Diameter (mm) | | 2 × 16; 5 × 14 | |
| Parallel chains longitudinal (*n*) | 13 | | |
| Diameter (mm) | 18 | | |
| Weight (kg) | 525 | | |
| Spacing (m) | 0.4 | | |
| Parallel arrays wired electrodes (*n*) | | | 13 |
| Spacing (m) | | | 0.5 |
| Ground chain length (m) | 24 | 24 | 19 |
| Diameter (mm) | 18 | 18 | 18 |
| Rollers length (m) | 4 | 4 | |
| Net mesh size stretched (mm) | 80 | 80 | 80 |
| Net opening shape | V | V | U |
| Net opening depth (m) | 10 | 10 | 5.2 |

## Experimental design

The field experiments were carried out using a stratified test design in a regular order with alternating trawl treatments (Figure 1b). The difference between pre-trawling ($T_0$) and post-trawling ($T_1$) megafaunal densities was assumed to be due to the passage of the trawl (i.e. short-term trawling mortality). For consistent results, only species matching the following criteria were included in the test: (i) sedentary or low motility species, so unable to flee from or to colonize the trawled strips between $T_0$ and $T_1$, (ii) large and robust enough to be reliably sampled by the Triple-D, and (iii) caught at least in two of the ten $T_0$ Triple-D transects in each strip. All crabs were excluded, except the burrowing female and juveniles (<1 cm carapax width) of *Corystes cassavilaunus*. Because of their motility *C. cassavilaunus* males were excluded. The 13 test species consisted of five bivalves

**Table 2.** Experimental trawling.

| Strip | Beam width (m) | Hauls (n) | Strip width trawled (m)[a] | Trawling frequency (mean number of hauls $m^{-2}$) | Interval trawling to $T_1$ (days) |
|---|---|---|---|---|---|
| L1 | 7 | 12 | 35 | 2.4 | 5 |
| L2 | 7 | 12 | 35 | 2.4 | 2–5[b] |
| S1 | 7 | 12 | 45 | 1.9 | 5 |
| S2 | 7 | 12 | 40 | 2.1 | 6 |
| P1 | 7 | 12 | 35 | 2.4 | 5 |
| P2 | 7 | 12 | 37.5 | 2.2 | 5 |

Three pairs of replicate strips were trawled with the longitudinal chain beam trawl (L), the standard tickler chain beam trawl (S), and the pulse beam trawl (P).

[a] Width of the trawled strip was estimated from side-scan recordings.

[b] Interval between trawling and the $T_1$ sampling of the first six Triple-D transects was 2 days, of the last four transects 5 days.

**Table 3.** Average catch efficiency of standard and alternative trawls for the test species expressed as the percentage of the $T_0$ Triple-D density (Table 5) that was caught in the trawl and brought aboard.

| | Catch efficiency trawls | | | | | |
|---|---|---|---|---|---|---|
| | Longitudinal | | Standard | | Pulse | |
| Species | % | SD | % | SD | % | SD |
| *A. echinata* | 0.6 | 0.7 | 2.0 | 2.3 | 1.0 | 1.2 |
| *A. aculeata* | 3.4 | 0.7 | 9.9 | 3.5 | 3.7 | 0.4 |
| *A. islandica* | 1.3 | 1.2 | 14.9 | 1.0 | 0.1 | 0.1 |
| *A. irregularis* | 2.2 | 0.9 | 7.4 | 4.7 | 1.9 | 0.2 |
| *C. gallina* | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| *C. cassivelaunus* | 1.0 | 0.5 | 2.8[a] | – | 2.9 | 0.7 |
| *C. elongata* | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| *D. lupinus* | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| *E. cordatum* | 0.1 | 0.2 | 0.0[a] | – | 0.0 | 0.0 |
| *G. vulgaris* | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| *M. truncata* | 0.1 | 0.1 | 0.0 | 0.0 | 0.0 | 0.0 |
| *N. norvegicus* | 12.9 | 11.9 | 4.2 | 3.5 | 17.8 | 21.1 |
| *T. communis* | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |

[a] Based on only one strip.

(*Acanthocardia echinata*, *Arctica islandica*, *Chamelea gallina*, *Dosinia lupines*, and *Mya truncata*), three echinoderms (*Astropecten irregularis*, *Cucumaria elongate*, and *Echinocardium cordatum*), two crustaceans (*Corystes cassivelaunus* and *Nephrops norvegicus*), a gastropod (*Turritella communis*), a polychaete (*Aphrodita aculeata*), and a sipuncoloid (*Golfingia vulgaris*).

Studies on anthropogenic disturbance are often advised to use (beyond-) BACI designs to minimize the influence of location-specific temporal differences (Underwood, 1992; Løkkeborg, 2007). Field experiments are, however, often limited by budgetary and logistical constrains. Because the period between $T_0$ (before trawling) and $T_1$ (after trawling) sampling was not >6 days (Table 2), and target species consisted only of long-lived merely sessile or low motility species, we assumed that no natural shifts in species densities unrelated to trawling would occur within this period. Therefore, we only sampled once before and after trawling. Spatial variation in species distributions between strips in $T_0$ sampling was compensated for by pairing strips by treatment and by alternating strips spatially, such that replicate strips were 1000 m apart (Figure 1b). Due to ship manoeuvring limitations, strips needed to be parallel and sampling transects had to be regularly ordered within strips.

At the test site, six parallel experimental strips (2000 by 50 m each) separated by 300 m were delineated and trawled alternatingly with the three trawl types (two replicate strips per trawl type; Figure 1b). Before ($T_0$) and after trawling ($T_1$), larger-sized, often more sparsely distributed, and longer-lived in- and epifauna was quantitatively sampled by the Triple-D dredge, which excised a 20-cm-wide, 14-cm-deep, and 100-m-long strip off the seabed (Bergman and van Santbrink, 1994). The length was controlled by an odometer triggering a pneumatic opening/closing cutting mechanism. The sediment washed through the 7 mm × 7 mm meshes of its 6-m net. Catches representing the fauna in 20 $m^2$ were sorted and identified on board (NIOZ, 2012; Witbaard *et al.*, 2013).

In the $T_0$ Triple-D sampling, megafaunal densities were assessed in ten regularly spaced transects per strip to reduce potential bias by faunal patchiness (Figure 1b). Sampling was followed shortly after by trawling with one of the three trawl types in a series of 12 hauls of 2000-m length to assure complete coverage of the strip. To estimate trawl-specific catch efficiencies, the first three trawl hauls on each strip were non-overlapping and sorted separately. Species-specific catch efficiencies were calculated by relating average species strip densities based on these three hauls (back-transformed square root transformed data) to the average strip density based on the ten Triple-D $T_0$ transects. Catch efficiencies in replicate strips were averaged (Table 3). The nine further hauls in each strip were made slightly overlapping to ensure complete coverage. Returns of trawl hauls were discarded far outside the test area.

After trawling, a side-scan sonar survey revealed that the width of the created strips varied between 35 and 45 m. The total width trawled ($n = 12$ hauls, trawl width 7 m) divided by this strip width made clear that trawling frequency of the strips, i.e. the average number each $m^2$ in a strip was trawled, varied (Table 2). The mean trawling frequency was 2.2 ($SD = 0.2$) and appeared at lowest in standard strips (1.9 and 2.1) but higher in strips trawled with pulses (2.4 and 2.2) and longitudinal chains (2.4 and 2.4). The relative low trawling frequencies in the standard strips might have led to an overestimated $T_1$ density, resulting in a potentially underestimated mortality. However, trawling causes a mosaic of random patches trawled at different frequencies within each strip in which ten transects were sampled with the Triple-D, assumedly minimizing the effect of varying mean trawl frequencies.

A similar Triple-D sampling was planned 5–6 days after trawling at $T_1$ (Table 2), long enough for fish and epibenthos to scavenge the organisms damaged or dislodged by the trawling (Groenewold and Fonds, 2000) and short enough to induce no natural changes in community composition between the strips, because the test species are long-lived, sessile, and of low mobility. For logistic reasons, scavengers only had 2 days instead of 5 days to remove sub-lethally damaged specimens in six of the ten transects in one of the six strips ($L_2$) after trawling (Table 2), which might have resulted in overestimated $T_1$ densities in this strip and thus an underestimated mortality. The $T_1$ transects were planned parallel to and within 3 m of the $T_0$ transects. Side-scan sonar surveys confirmed that all Triple-D $T_1$ transects were located inside the trawled strip.

## Data analyses

### *Pre-trawling strip differences in faunal distribution*

The potential existence of pre-trawling differences in faunal composition at the strip level was assessed using multivariate analysis (PERMANOVA) by comparing a model where strips were nested within trawl type to a non-nested design.

### *Trawling mortality*

Differences between standard and alternative trawling were examined by comparing both shifts in species composition using multivariate statistics (community level) and mortality rates in single species. At the community level, a non-metric multi-dimensional scaling (nMDS) plot was used to display the $T_0$ and $T_1$ species composition by plotting six centroids each representing a "gravity" centre (community mean) of the ten transects in a strip (Figure 3). The density data of the 13 selected species were square root transformed, and dissimilarity matrices were calculated using the Bray–Curtis similarity. Differences between trawl types were tested by PERMANOVA tests. The first factor "trawl" divided the data set into three trawl types, the second factor "timing" divided the data set into two sampling moments, the third factor nested the replicate strips under trawl type, and the interaction term "trawl × timing" tested whether the effects of trawl types differed over the interval $T_0$–$T_1$. Pairwise comparisons used Monte Carlo tests to correct for the limited number of possible permutations. A SIMPER analysis was used to investigate per trawl type the individual species contributions to the dissimilarity in species composition between $T_1$ and $T_0$ (Table 4). PERMANOVA and SIMPER analyses used PRIMER (v7.0.13; Clarke and Gorley, 2006) together with PERMANOVA+ add on (Anderson *et al.*, 2008).

Within single species, the effect of the trawl types was analysed by comparing $T_0$ and $T_1$ densities (Table 5). Initial analyses using mixed effect models with strip as a random effect showed that the strip effect was not significant. Therefore, we used generalized linear models (GLMs) without a random effect with a Poisson error distribution and a logarithmic link function to compare $T_0$ and $T_1$ densities. The error distribution was corrected for overdispersion (so-called "quasipoisson" in R). Statistical significance of main effects and interaction was tested using likelihood ratio tests. Each single species model contained trawl type (three levels), timing ($T_0$, $T_1$), and their interaction as main effects. A significant interaction indicates a difference between trawl types from $T_0$ to $T_1$. To elucidate the reason for a significant interaction, 95% confidence limits were calculated. Means and 95% confidence limits were determined using simulation and afterwards back-transformed ($n = 1000$; Gelman and Hill, 2007). The simulations were also used to calculate the average percentage of trawling mortalities and 95% confidence limits (Figures 4 and 5). All univariate analyses were performed in R v. 3.2.3 (R Core Team, 2019). Mixed effects models used package nlme (Pinheiro

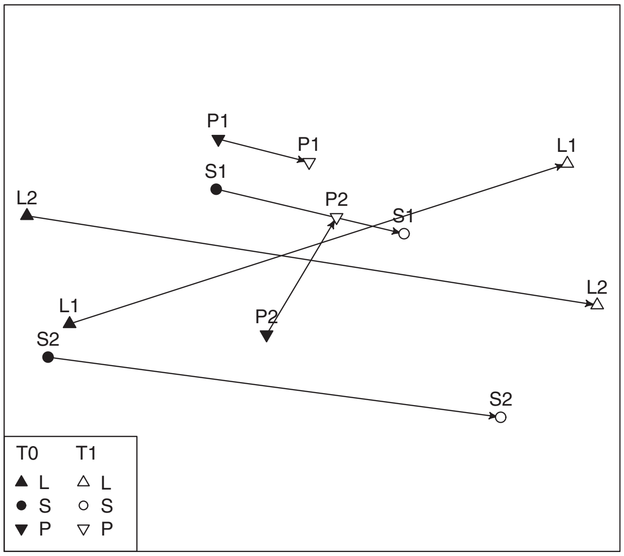

**Figure 3.** Average species composition (nMDS ordination) plot of the strips prior ($T_0$) and after ($T_1$) trawling. L1 and L2, longitudinal chain beam trawl; S1 and S2, standard tickler chain beam trawl; P1 and P2, pulse beam trawl, all pre-trawling and post-trawling, respectively. Each point represents the centroid of the species densities of ten Triple-D tracks.

**Table 4.** Species contributions to the total dissimilarity between $T_0$ and $T_1$ for different trawl types (up to 75%, SIMPER analysis).

| **Longitudinal** | | **Standard** | | **Pulse** | |
|---|---|---|---|---|---|
| **Species** | **Contrib. %** | **Species** | **Contrib. %** | **Species** | **Contrib. %** |
| *E. cordatum* | 15.1 | *E. cordatum* | 17.7 | *T. communis* | 13.8 |
| *A. irregularis* | 12.2 | *T. communis* | 13.8 | *E. cordatum* | 12.4 |
| *T. communis* | 11.5 | *A. irregularis* | 8.4 | *A. irregularis* | 8.9 |
| *C. cassivelaunus* | 10.2 | *D. lupinus* | 8.3 | *C. cassivelaunus* | 8.7 |
| *A. aculeata* | 7.5 | *C. cassivelaunus* | 7.7 | *D. lupinus* | 8.1 |
| *D. lupinus* | 6.9 | *C. gallina* | 6.8 | *A. aculeata* | 7.7 |
| *C. elongata* | 6.4 | *A. aculeata* | 6.4 | *C. gallina* | 7.3 |
| *C. gallina* | 6.3 | *C. elongata* | 5.7 | *G. vulgaris* | 6.7 |
| Av. diss. | 33.6 | Av. diss. | 30.6 | Av. diss. | 27.2 |

Last row (Av. diss.) gives the average dissimilarity between all pairs of $T_0$–$T_1$ samples. For densities see Table 5.

**Table 5.** Mean species densities ($n = 100\,\mathrm{m}^{-2}$) based on back-transformed logarithmically transformed values with 95% confidence interval for Tripe-D transects in replicate strips before ($T_0$) and after ($T_1$) trawling.

| | Longitudinal | | Standard | | Pulse | | |
|---|---|---|---|---|---|---|---|
| **Species** | **$T_0$ density ($n = 100\,\mathrm{m}^{-2}$)** | **$T_1$ density ($n = 100\,\mathrm{m}^{-2}$)** | **$T_0$ density ($n = 100\,\mathrm{m}^{-2}$)** | **$T_1$ density ($n = 100\,\mathrm{m}^{-2}$)** | **$T_0$ density ($n = 100\,\mathrm{m}^{-2}$)** | **$T_1$ density ($n = 100\,\mathrm{m}^{-2}$)** | ***P*-value trawl type × time** |
| *A. echinata* | **5.5** (3.5–8.3) | **0.8** (0.3–2.4) | 4 (2.4–6.7) | 1.2 (0.5–3) | 2.6 (1.4–4.7) | 1.7 (0.8–3.7) | 0.11 |
| *A. aculeata* | **25.9** (20.4–34) | **10.7** (7.3–16.1) | 21.4 (16.4–28.1) | 12.2 (8.7–17.7) | 16.9 (12.5–23) | 18.9 (14.3–25.4) | **0.01** |
| *A. islandica* | 7.5 (4.9–11.8) | 3.6 (2–6.4) | 3.2 (1.7–6.4) | 2.5 (1.3–5.3) | 5 (3–8.4) | 3.8 (2.1–7.1) | 0.62 |
| *A. irregularis* | **53.7** (42.6–66.3) | **15.3**(10–22.8) | **42.4** (32.7–54.4) | **22** (15.6–31.5) | 39.4 (30–51.1) | 30.9 (23.6–41.6) | **0.01** |
| *C. gallina* | 20.6 (15.6–26.8) | 12.3 (8.6–18) | 16.7 (12.2–22.6) | 10.6 (7.3–15.9) | 14 (10.3–19.9) | 7.9 (5.1–12.2) | 0.95 |
| *C. cassivelaunus* | **33** (26.5–40.9) | **9.8**(6.6–14.6) | 24.2 (18.7–31.6) | 15.5 (11.2–21.6) | 29.8 (23.1–37.7) | 19.6 (14.8–26.9) | **0.02** |
| *C. elongata* | **15.5** (11.9–20.2) | **6.2** (4.1–9.4) | 11.5 (8.5–15.4) | 10 (7.2–13.8) | 11.3 (8.3–15.4) | 10.3 (7.5–14.4) | **0.03** |
| *D. lupinus* | **44.9** (36.6–54.8) | **20.5** (15.2–27.4) | 35 (28.2–44) | 22 (16.6–29.6) | 32 (25.6–40.7) | 30 (23.6–38) | **0.01** |
| *E. cordatum* | **154.3** (129.5–183.7) | **54.4**(39.8–74.8) | **161.5** (135.6–194.1) | **63.2**[a] (47.4–84.9) | 113.5 (94–138.5) | 105.5[a] (86–130.7) | **0.00** |
| *G. vulgaris* | 4.5 (2.4–8.6) | 2.5 (1.1–5.5) | 5.5 (3.2–9.9) | 3.3 (1.6–6.9) | 6 (3.6–10.4) | 1.5 (0.5–4.6) | 0.48 |
| *M. truncata* | 6 (3.7–9.2) | 1.8 (0.8–3.9) | 5.2 (3.3–8.3) | 3.5 (2–6.3) | 4 (2.2–7.3) | 4.5 (2.7–7.4) | 0.08 |
| *N. norvegicus* | **3.2** (1.9–5.5) | **0.5** (0.1–1.9) | **4.9** (3–7.7) | **0.3** (0–2.2) | 2.3 (1.2–4.4) | 0.5 (0.1–2) | 0.50 |
| *T. communis* | **137.9** (110.9–173.4) | **74.1** (53.7–98.7) | 123.3 (96.7–156.5) | 78.4 (58.4–105.7) | 91.3 (69.4–122.2) | 69.4 (50.5–96.5) | 0.47 |

Within trawl type, bold number indicates significant density decreases per trawl type (based on 95% confidence limits). Between trawl types, underlined values denote significant differences between longitudinal chain and pulse $T_1$ values. In the last column, bold *P*-values indicate a significant interaction between trawl type and timing ($T_0$–$T_1$; $P < 0.05$; GLM with Poisson error distribution and logarithmic link function).

[a] Significant difference between standard and pulse $T_1$ values.

et al., 2019), and simulations were carried out using package arm (Gelman and Su, 2018).

### *Power analysis*

To assess the statistical power to find a significant difference between species densities, we performed a power analysis (Johnson *et al.*, 2015) using the average Triple-D $T_0$-density of the 13 species and their variation between strips ($n = 6$) and between transects ($n = 10$ per strip). Two datasets were generated for each species. The first set was based on the mean pre-trawling densities ($T_0$) and the variances between transects and strips. The second dataset (treatment) simulated post-trawling densities ($T_1$) with an effect size of 10–90% in 10% increments using $T_0$ variances. We used two generalized linear mixed models with a Poisson distribution and strips as a random effect. The null model included one variable, namely trawling effect. The nested model was an intercept-only model with the same random effect. Overdispersion was modelled via an extra random effect for each Triple-D transect (Bates *et al.*, 2015). A likelihood ratio test between null and nested models gave the significance of the trawling effect. This was repeated 400 times for each effect size. The power was calculated as 1 − the relative number of times the test failed to detect a significant difference between $T_0$ and $T_1$ (trawl mortality) expressed as a percentage of the total number of simulations (i.e. 1 − type-II error; Table 6). Power analysis was repeated for several experimental designs to examine the increase in power due to variation in the numbers of strips and transects. We calculated power for 2 strips with 10 and 20 transects, for 4 strips with 5, 10, and 20 transects, and for 8 strips with 5 and 10 transects (Figure 6). Per species, this resulted in a total of 28000 simulations.

## Results

### Pre-trawling faunal distribution

Before trawling, no statistical difference was found in the species composition of the three pairs of replicate strips (PERMANOVA, $P = 0.7$), even though some differences existed between individual strips (PERMANOVA, $P = 0.014$). Thus, existing variation between individual strips was successfully compensated for by pairing two strips within each trawl type.

### Catch efficiency

The catch efficiency of the trawls was relatively low for most test species (Table 3). The highest efficiencies (7–15%) were found for large-sized, epibenthic or shallow-burrowing species, such as *A. irregularis*, *A. aculeate*, and *A. islandica*, as well as for the commercially targeted *N. norvegicus* (18%). Less than 3% of the shallow-burrowing *C. cassivelaunus* (females, juveniles) and *A. echinata* were caught. Catch efficiency for the large-sized deeper-burrowing *M. truncata* and *E. cordatum* was <0.1%, and that for the other five smaller-sized species was nihil.

### Trawling mortality

#### *Multiple species*

After trawling, all megafaunal communities changed in a similar direction, regardless the type of trawl as indicated by a similar shift of the centroids to the right in Figure 3. The distance of displacement is indicative of the degree of change. Compared to standard trawling, trawling with longitudinal chains gave a larger displacement and, contrastingly, pulse trawling gave a smaller displacement. Shifts between trawl types over the interval $T_0$–$T_1$ were significantly different ($P = 0.032$ for the interaction term trawl × timing). Pairwise PERMANOVA tests between $T_0$ and $T_1$ communities within trawl types may well indicate a ranking in the effects of trawl types: the shift from $T_0$ to $T_1$ was strongly significant for longitudinal chain trawling ($P = 0.005$), almost significant for standard trawling ($P = 0.10$), and insignificant for pulse trawling ($P = 0.42$). The ranking in *P*-values calculated for pairwise comparisons between trawl types at $T_1$ seems in line with this: a significant difference in species composition between longitudinal and pulse trawling ($P = 0.04$), whereas the *P*-value was 0.63 for longitudinal chain trawling vs. standard trawling and

**Table 6.** Power characteristics.

| Species | $T_0$ density ($n = 20\ m^{-2}$) | Variance | | Overdispersion | | Effect size (% change) for 80% of power | |
|---|---|---|---|---|---|---|---|
| | | Transects | Strips | $\chi^2$ | *P*-value | 2 strips; 10 transects each | 8 strips; 20 transects each |
| *A. echinata* | 0.67 | 0.37 | 0 | 2.05 | 0.15 | **85** | 38 |
| *A. aculeata* | 3.89 | 0.19 | 0 | 12.04 | <0.001 | **63** | 33 |
| *A. islandica* | 0.81 | 0.37 | 0.14 | 3.50 | 0.06 | **87** | 54 |
| *A. irregularis* | 7.79 | 0.27 | 0.03 | 51.22 | 0 | **64** | 38 |
| *C. gallina* | 3.05 | 0.21 | 0.02 | 11.09 | <0.001 | **57** | 29 |
| *C. cassivelaunus* | 5.18 | 0.22 | 0.03 | 19.78 | 0 | **68** | 39 |
| *C. elongata* | 2.47 | 0.06 | 0 | 0.63 | 0.43 | **55** | 22 |
| *D. lupinus* | 6.97 | 0.13 | 0.01 | 16.38 | <0.001 | **45** | 23 |
| *E. cordatum* | 25.56 | 0.23 | 0.02 | 187.51 | 0 | **51** | 25 |
| *G. vulgaris* | 0.72 | 0.79 | 0 | 16.57 | 0 | **80** | 36 |
| *M. truncata* | 0.97 | 0.08 | 0.02 | 0.11 | 0.74 | **73** | 31 |
| *N. norvegicus* | 0.56 | 0.40 | 0 | 1.90 | 0.17 | **88** | 40 |
| *T. communis* | 18.32 | 0.51 | 0.08 | 336.20 | 0 | **62** | 39 |

Average $T_0$ density of the Triple-D transects per species ($n = 20\ m^{-2}$), transect and strip variance, results of overdispersion test, and the effect size that would give a statistically significant ($\alpha = 0.05$) difference between two means with 80% of power for (i) current sampling design: two strips with each ten transects (bold) and as an example (ii) eight strips with each 20 transects.

0.19 for pulse trawling vs. standard trawling. In addition, the average dissimilarity, indicative for the change in community composition between $T_0$ and $T_1$ communities, decreased from longitudinal trawling via standard trawling to pulse trawling (SIMPER dissimilarity values 33.6, 30.6, and 27.2; Table 4).

Contributions of the individual species to the shifts in community centroids in Figure 3 (i.e. the average dissimilarity between $T_0$ and $T_1$ for each trawl type) indicated that 9 of the 13 species contributed 75% to the total dissimilarity (Table 4). The seven species common to all three trawl types contributed >66% to the total dissimilarity between $T_0$ and $T_1$. The highest ranked three species are the same for each trawl type, namely *E. cordatum*, *T. communis*, and *A. irregularis*.

### Single species

Almost all species showed reduced mean densities after trawling irrespective of trawl type (Table 5). Only in pulse trawling, two species did increase on average slightly (though not significantly), namely *A. aculeata* and *M. truncata*. Significant decreases in density, however, existed after longitudinal chain trawling in nine species (bold in columns 5 and 6) and after standard trawling in three species (bold values in columns 1 and 2 in Table 5). None of the species showed a significant decline after pulse trawling (columns 3 and 4 in Table 5). Before trawling, species densities were the same, since none of the $T_0$ densities were significantly different.

In 6 of the 13 species, density changes from $T_0$ to $T_1$ differed significantly between trawl types (GLM interaction between trawl type and timing, last column of Table 5), namely *A. aculeata*, *A. irregularis*, *C. cassivelaunus*, *Cucumaria elongata*, *Dosinia lupinus*, and *E. cordatum*. In three of these species, significance appears to be due to a strong decrease only after longitudinal trawling and almost no decrease after pulse trawling (*A. aculeata*, *C. elongata*, and *D. lupinus*), while in the other three species (*A. irregularis*, *C. cassivelaunus*, and *E. cordatum*), $T_1$ densities decreased just stronger after longitudinal trawling than after pulse trawling. Only in *E. cordatum*, there is also a significant difference between $T_1$ densities after standard and pulse trawling. Between $T_1$ densities in standard and longitudinal trawling, no significant differences were found.

Figure 4 condenses the results of Table 5, showing the relative decrease (i.e., mortality) after trawling for each species and three trawl types. In 11 species (4 significant) longitudinal trawling gives a higher mortality than pulse trawling and, similarly, in 12 species (0 significant) a higher mortality than standard trawling. In 10 of the 13 species, however, pulse trawling results in less mortality than standard trawling, although only significant for *E. cordatum*. The probability to find 10, 11, or 12 of the 13 species with lower or higher mortalities by chance is 0.046, 0.011, and 0.0017, respectively (one-sided binomial test assuming less mortality for alternative trawls).

Averaged over all species (ignoring within species variation) trawling mortality decreased from longitudinal chain trawling (62%) through standard (44%) to pulse trawling (25%), although mortality was only significantly different between pulse and longitudinal chain trawling (Figure 5). Based on these averages, longitudinal trawling may cause 41% more mortality when compared to standard trawling, while pulse trawling may give 43% less mortality.

### Statistical power

The minimum species-specific effect size required to detect a statistically ($\alpha = 0.05$) significant difference between two means with an 80% certainty is given for two test designs (Table 6, last two columns). In the used test design with two strips (ten transects each), the minimum change between, e.g. $T_0$ and $T_1$ densities has to be on average 68%, varying from 45% for *D. lupinus* to 88% for *N. norvegicus*. Lower changes in density as found for many test species (Figure 4 and Table 5) lead to less power and thus lower probability to detect an existing difference. Power can be improved if sampling error is reduced, e.g. by including more strips and/or transects. For example, a test design with 160 samples spread over 8 strips (20 transects each) requires a minimum mean trawling mortality of only 34% to achieve 80% of power (Table 6).

Figure 6 gives two typical power plots, each depicting the power curves for seven different test designs regarding number of

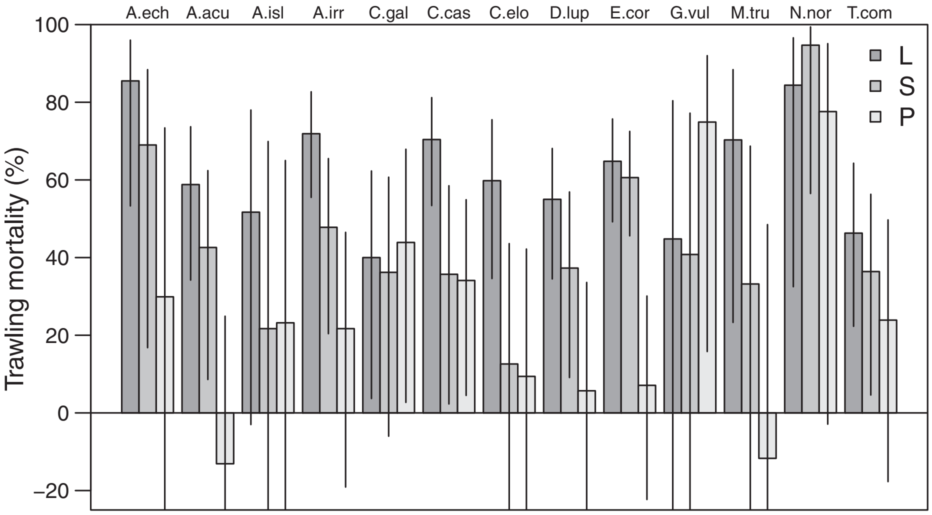

**Figure 4.** Average trawling mortality per species due to different trawl types expressed as percentage change relative to the initial $T_0$ species density (with 95% confidence limits based on simulations, see "Material and methods" section). L, longitudinal chain beam trawl; S, standard tickler chain beam trawl; P, pulse beam trawl. Note that confidence limits extending below −25% are not shown.

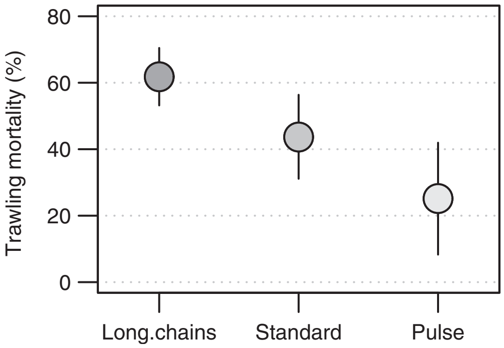

**Figure 5.** Mean trawling mortality (with 95% confidence limits) per trawl type averaged over all species.

strips (S) and transects (T) for a particular species. The power plots of the other 11 species are available in the Supplementary material. *Astropecten irregularis* (Figure 6a) represents the best example of a group of species where an increase in strip numbers leads to a lower effect size to achieve 80% of power. In a design of 8 strips with 5 transects each ($n = 40$), a smaller difference (lower effect size) can be detected with 80% of power than in 4 strips with 20 transects each ($n = 80$); thus, half the effort still has more power if spread over double the number of strips. Similar responses are found for *T. communis*, *C. cassivelaunus*, *E. cordatum*, *A. aculeata*, *D. lupinus* and *A. islandica* (Supplementary Figures S1–S6), all species with clustered distributions, as indicated by relatively high mean strip variances and a significant overdispersion (Table 6). Contrastingly, in the second group of species represented by *C. elongata* (Figure 6b), more strips do not lead to lower effect sizes at 80% of power. Lower effect sizes are simply achieved by including more transects. Analogous responses are found in *N. norvegicus*, *G. vulgaris*, *M. truncata*, and *A. echinata*, all species with homogeneous distributions, smaller mean strip variances, and no overdispersion (Supplementary Figures S7–S10). *Chamelea gallina* (Supplementary Figure S11) shows an intermediate response.

## Discussion

### Differences in trawling mortality

In this study, we tested the impact of two alternative trawl types on 13 megafaunal species. Many fragile, larger-sized, and longer-lived in- and epibenthic species are vulnerable to trawling and have become rare in the southern North Sea during the last decennia (de Vooys, 2004; Capasso *et al.*, 2010; Witbaard *et al.*, 2013; van Denderen *et al.*, 2014). Regular trawling also affects long-living sponge and bivalve beds including their biogenically structured three-dimensional habitats (Sheppard, 2006) and leads to less habitat complexity (Gray *et al.*, 2007). The loss of vulnerable species and habitats hampers the detection of trawling impact in field studies (Teal *et al.*, 2014; this study), while un-trawled test areas are virtually non-existing. The still present megafaunal species, such as our 13 test species, are most likely characterized by life history strategies that make them less vulnerable to trawling. In addition, many benthic species including 8 of the 13 test species show clustered distributions (Table 6; Morrisey *et al.*, 1992).

Less species, less vulnerable to trawling, and lower and clustered abundances lead to the loss of statistical power to detect trawling impact. Species-specific differences in density (Table 5) were often too small (Table 6) to result in significant comparisons, both between $T_1$ densities due to different trawl types and between $T_0$ and $T_1$ densities within trawl types. Comparisons between trawl types were only significant between trawl types generating the most contrasting $T_1$ densities, namely longitudinal and pulse trawling (three species; Table 5). The difference between $T_1$ densities after standard trawling and pulse trawling was only for *E. cordatum* large enough to become significant, for other species, the $T_1$ densities differed not enough for significance, and comparisons most likely had too little power (Table 6). For the same reason, $T_1$ differences between longitudinal chain trawling and standard trawling were not statistically significant. These results indicate that standard trawling has an impact that lies between those of longitudinal chain and pulse trawling.

Other results, although not always based on statistically significant differences, consistently support the above conclusion of a

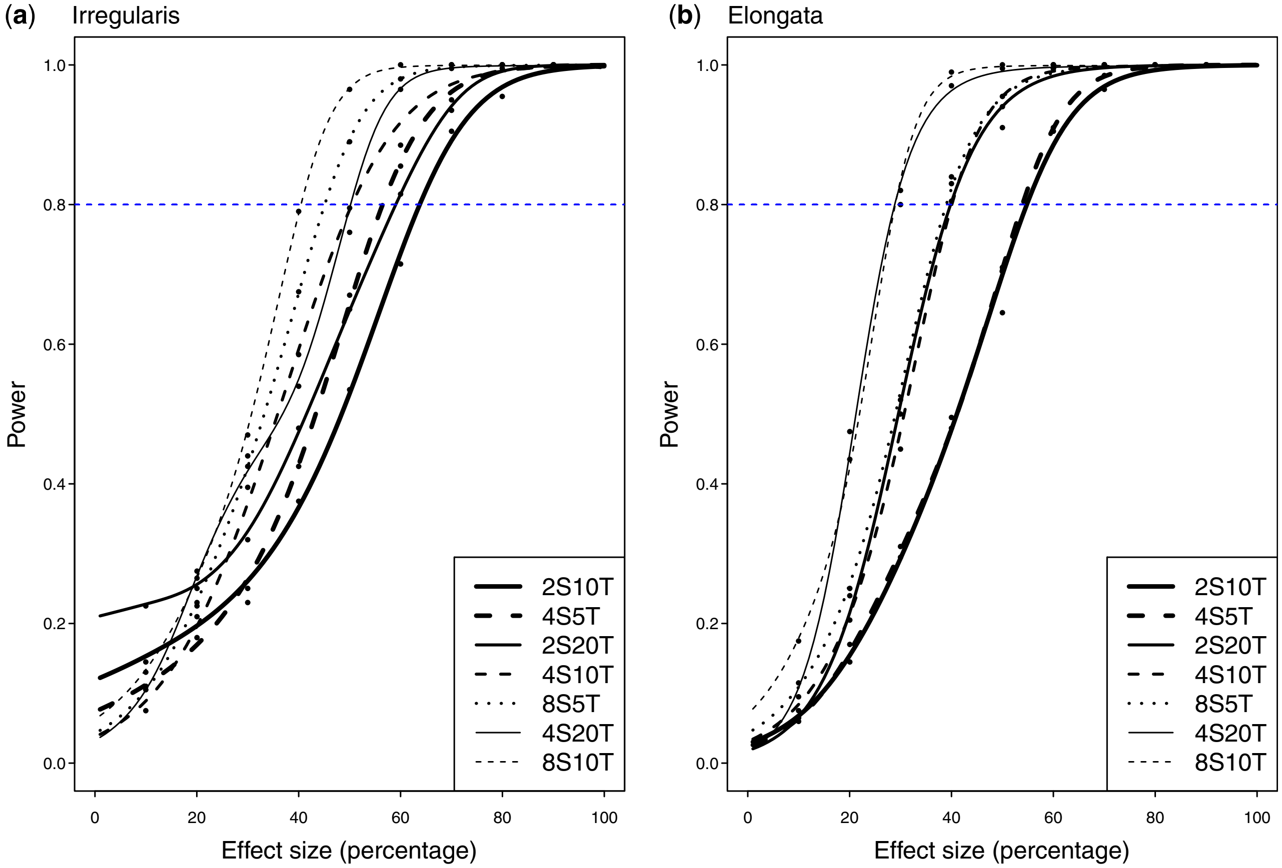

**Figure 6.** Power plots of (a) *A. irregularis* and (b) *C. elongata* for detecting a significant difference between two means of 7 different scenarios combining different numbers of strips (S) and sampling transects (T). Horizontal line depicts 80% power.

decreasing impact from longitudinal chain trawling, via standard to pulse trawling:

(i) The number of single species with significant trawling mortality show a ranking: longitudinal trawling (9), standard trawling (3), and pulse trawling (0); (Table 5).

(ii) Mortality averaged over all species (Figure 5) decreased from longitudinal chain trawling (62%) to standard trawling (44%) to pulse trawling (25%), although differed significantly only between longitudinal chain trawling and pulse trawling. These averages suggest that compared to standard trawling, longitudinal chain trawling may give 41% more mortality and pulse trawling may give 43% less mortality.

(iii) The change in species composition was largest and significant after longitudinal trawling, less after standard trawling, and least after pulse trawling (Figure 3; $P = 0.005$, 0.10, and 0.42, respectively), corresponding with a decreasing dissimilarity between $T_0$ and $T_1$ sampling from 33.6, to 30.6, to 27.2, respectively (Table 4). Pairwise comparisons between trawl types at $T_1$ also confirm this trend: the *P*-value is 0.04 for longitudinal trawling vs. pulse trawling, 0.63 for longitudinal chain trawling vs. standard trawling, and 0.19 for standard trawling vs. pulse trawling.

(iv) The probability that 12 of the 13 species show higher mortality after longitudinal trawling than after standard trawling (Figure 4) and 10 of the 13 species ($P$=0.046) show lower mortality was solely due to chance appeared 0.0017 and 0.046, respectively.

This study was aimed at comparing direct mortality rates due to the three trawl types. The absolute mortality rates found are most likely related to the type of test species and the soft sediment test area. That might be the reason that our estimate for standard mortality (25%) is on the higher end of the range of a large number of experimental studies reported in Hiddink *et al.* (2017) who give a median depletion of 14% with 25% being the 95% percentile. Estimates for standard trawling mortality in other studies seem to suggest that mortality rates are higher in softer sediments than in sandier sediments (Bergman and van Santbrink, 2000; van Denderen *et al.*, 2014). A higher mortality can also be anticipated in un-trawled areas when inhabited by vulnerable taxa, such as ascidians, soft corals, bivalve banks, and reef-building polychaetes.

## Physical differences between trawl types

Differences in trawling impact can be related to properties of the trawls (i.e. stimulation type, weight, and towing speed) generating a disturbance of the sediment over a certain depth (Table 1). The median disturbance depth for a wide variety of beam trawls is 2.7 cm (Hiddink *et al.*, 2017) but can range up to 8 cm for conventional tickler chain trawls (Paschen *et al.*, 2000). The stronger impact found for longitudinal chain trawling relative to standard trawling cannot be related to weight or towing speed as weight was less than the standard trawl and speed similar (Table 1). Most likely the longitudinal chains wiggled and thrashed through the seabed, contrary to the initial expectations that the chains would stay parallel while being towed, leaving the seabed in between intact. If so, fitting extra longitudinal chains or cables in pulse trawls, often common practice to improve their commercial efficiency, most probably will increase direct mortality substantially.

Pulse gears are lacking cross running tickler chains and are consequently lighter than standard trawls. Depestele *et al.* (2016) reported a mean disturbance depth of 1–2 cm for both standard and pulse beam trawls and related the diminished alteration of the seabed bathymetry after pulse trawling to reduced disturbance. Likewise, differences in sediment depth disturbance were found for electro-fitted SumWing (PulseWing) trawls, which penetrated 55% less deep than tickler chain SumWing trawls (1.8 and 4.0 cm, respectively; Depestele *et al.*, 2019). The lower towing speed of pulse trawls, both in commercial practice and in field studies, may contribute to reduced damage and mortality by the pass of the trawl.

## Bias in results

Variations in mean trawling frequency of the strips caused by more or less accidental overlap in hauls may introduce bias. Both standard strips appeared trawled at the lowest frequencies (Table 2) potentially leading to overestimated $T_1$ densities (i.e. underestimated mortality). As a consequence, differences in mortality between standard trawling and pulse trawling might have been underestimated and differences in mortality between standard and longitudinal trawling might have been overestimated. On the other hand, average mortality in the longitudinal strips may have been underestimated, since the interval between trawling and $T_1$ sampling in $L_2$ was shorter than planned, 2 days instead of 5 days. Due to both sources of bias, differences between pulse and other trawl types may have been larger if the strips were trawled with equal frequencies and intervals.

Another potentially biasing factor is the survival of specimens discarded after trawling, causing an underestimated $T_1$ density and consequently overestimated mean mortality. In *A. islandica* with its relative high catchability in standard trawls (15%; Table 3) but low survival (12% in survival tanks; Lindeboom and de Groot, 1998), this overestimate could have been at most 1.8%, which will not change conclusions (Figure 4). For *A. irregularis* and *A. aculeata* pairing relative high catch efficiencies with high mean survival percentages (90% in tanks; Kaiser and Spencer, 1995; Depestele *et al.*, 2014), overestimates in mortality could have reached up to 7 and 9%, respectively, also not changing the conclusions. For the majority of test species, however, this bias due to survival is negligible since their species-specific catch efficiencies in the trawls were small in comparison to trawling mortality (also reported in Bergman and van Santbrink, 2000).

## Statistical power

In the current experimental design, statistical power to detect a significant ($\alpha = 0.05$) difference between two means was <80% for many species (c.f. Tables 5 and 6). Power can be improved by attaining a smaller sampling error, e.g. by increasing the numbers of samples. The power plots (Figure 6) indicate that statistical power can be higher after increasing the number of strips or transects. For species such as *A. irregularis* (Figure 6a and Supplementary Figures S1–S6) with more clustered distributions at strip scale (Table 6), power improves if the same number of transects (or even less) is spread over more strips. On the other hand, species such as *C. elongata* (Figure 6b and Supplementary Figures S7–S10) representing more homogeneous distributed species (Table 6), require simply more transects irrespective the number of strips. Since in our experiment seven test species required more strips and five test species required more transects to attain sufficient power, an optimal design would have included a combination of more strips with more transects. Budgetary limitations often will impede such project extensions, and another option is to perform field studies in un-trawled reference areas with higher abundances of vulnerable species.

## Supplementary data

Supplementary material is available at the *ICESJMS* online version of the manuscript.

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