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Atrina fragilis and echinoderms on circalittoral mud

MarLIN – Marine Life Information Network
Marine Evidence–based Sensitivity Assessment (MarESA) Review

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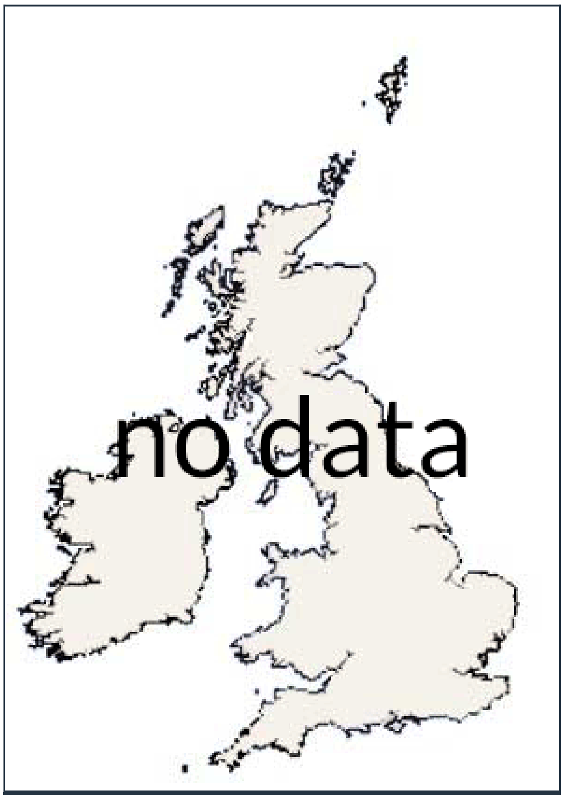

Biotope distribution data provided by EMODnet Seabed Habitats (www.emodnet-seabedhabitats.eu)

Researched by Dr Harvey Tyler-Walters **Refereed by** This information is not refereed

Summary

UK and Ireland classification

EUNIS 2008	A5.363TMP	None (TBC)
JNCC 2022	SS.SMu.CFiMu.AtrEch	*Atrina fragilis* and echinoderms on circalittoral mud
JNCC 2015	None	None
JNCC 2004	None	None
1997 Biotope	None	None

Description

Circalittoral mud and mixed sediments characterised by the epifaunal assemblage of fan mussel *Atrina fragilis* and brittle star *Amphiura*. The description of this biotope is based on epifauna recorded from Small Isles (northwest Scotland) but could be found in other areas with similar environmental conditions. The fauna was diverse with burrowing crustaceans, echinoderms, sea anemones, ascidians and bryozoans. The characteristic fauna includes anemone *Cylista lacerata*, *Alcyonidium digitatum*, Hydrozoa, serpulid worm *Salmacina dysteri* and *Munida rugosa*. The description of this biotope is based on video data so the characterizing fauna include only those species with an epifaunal expression. This epifaunal biotope may be associated with infaunal biotopes described for similar environmental conditions. (Information from JNCC, 2022).

Depth range

100-150 m

Additional information

The biotope description is based on surveys of the Sound of Canna, the Inner Hebrides, Scotland (Howson *et al*., 2012). Stirling *et al.* (2016) predicted that suitable habitat for *Atrina fragilis* was probably present in other Scottish MPAs, in particular the Small Isles, Loch Sunart and Wester Ross MPAs, based on species distribution models using existing records in western Scotland. Therefore, this biotope may be under-recorded at present.

Listed By

- none -

Further information sources

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Sensitivity review

Sensitivity characteristics of the habitat and relevant characteristic species

This is an epifaunal biotope (SS.SMu.CFiMu.AtrEch) dominated by *Atrina fragilis* and *Amphiura* spp. and other echinoderms. *Atrina fragilis* is the largest bivalve in UK waters and sits in the sediment with one-third to one-half of its shell length above the sediment surface, which may be 10-15 cm above the sediment surface in large specimens. The presence of *Atrina* may also alter small-scale flow dynamics, as is the case with *Atrina zealandica* where water flow in its natural habitat (from 0.1 - 0.2 m/s) created an internal boundary layer over the surface of the bed (Nikora *et al.*, 2002). The size of the boundary layer depended on *Atrina* density. At 340 individuals/4m^2 the boundary layer was ca 12 cm above the seabed, ca 1.3 to 3 times the height (4-9 cm) of the shells above the seabed but only ca 6 cm above the seabed (0.7-1.5 times the height of the shells) at 50 individuals/4m^2. Nikora *et al.* (2002) reported that the internal boundary layer may provide shelter for some organisms but also increase vertical exchange at the surface of the boundary layer and affect nutrient flux, the settlement of larvae, suspended sediment, and the suspension and deposition of biodeposits, within the bed. However, *Atrina zealandica* occurs in far greater densities than UK examples of *Atrina fragilis,* which were recorded at 2-4 individuals per square metre in the densest patches in the Sound of Canna (Howson *et al.*, 2012).

Large suspension-feeding bivalves, such as *Atrina*, have been shown to affect bentho-pelagic coupling, increase sediment flux to the seabed, deplete phytoplankton, and change ammonia and oxygen concentrations (Hewitt *et al.*, 2006). Gibbs *et al.* (2005) reported that *Atrina* beds increased nutrient supply to the water column in turbid waters in light and may contribute ca 80% of the nutrient supply for pelagic primary production in Marhurangi Harbour, New Zealand. Hewitt *et al.* (2002) found that the distance between individuals of *Atrina zealandica* affected the associated benthic macrofauna. Hewitt *et al.* (2006) noted that proximity to *Atrina* increased the abundance of small macrofauna that used the sediment-water interface (e.g. tube or surface dwelling surface deposit or suspension feeders), although the strength of the effect varied between sites. The settlement of biodeposits (faeces and pseudofaeces) from *Atrina zealandica* is likely to have localised effects on benthic community structure (Miller *et al.*, 2002). Pseudofaeces from pen shells resulted in biodeposits that reduced the variability of nematode meiofauna in *Atrina zealandica* beds (Warwick *et al.*, 1997). In addition, the *Atrina* beds increase surface roughness and alter water flow across the sediment (Nikora *et al.*, 2002; Norkko *et al.*, 2006). Norkko *et al.* (2006) reported increased macrofaunal abundance and species richness in *Atrina zealandica* beds in low suspended sediment but that increased suspended sediment loads decreased or reversed the effect. However, Cummings *et al.* (1998, 2001) noted that the relationship between *Atrina* beds and macrofaunal abundance was more complicated and also dependent on sediment characteristics and hydrography rather than the presence of *Atrina* and biodeposits alone. The same effects may be true of *Atrina fragilis* beds, although these effects are likely to be reduced due to their much lower densities.

Mollusc shells are often important settlement substrata for sessile organisms, or shelters/nest sites for mobile benthic animals in marine soft sediments (Kuhlmann, 1998). For example, *Pinna bicolor* hosts a species-rich epifauna composed of sponges, bryozoans, tunicates, Cnidaria, annelids and molluscs (Kay & Keough, 1981; Ward & Young, 1983). Similarly, Howson *et al.* (2012) reported that fan mussels in the Sound of Canna supported tasselly sponges, *Alcyonium digitatu*m and hydroids on their shells.

The brittlestars *Amphiura* filiformis and *Amphiura chaeji* are surface suspension-feeders and/or deposit feeders that may benefit from the biodeposits and shelter provided by *Atrina fragilis* (see Hewitt *et al*., 2006). *Ophiocomina nigra* is an obligate predator, scavenger and suspension feeder, living at the sediment surface and *Ophiura ophiura* is also a suspension feeder, predator and detritivore living on the sediment surface. Both may also benefit from their proximity to *Atrina fragilis* beds. The majority of other species recorded in this biotope are either typical of the surrounding sediments, e.g. burrowing anemones or mobile scavengers that roam the sediment surface e.g. starfish, crabs, squat lobster and whelks.

Overall, *Atrina* beds and aggregations have been shown to modify the water flow over the substratum, affect benthopelagic coupling and nutrient flux, create organic-rich biodeposits on the sediment and alter the macrofaunal communities within their vicinity, depending on their density. Fan mussels can also provide additional hard substratum for diverse epifauna on their shells. The density of *Atrina* reported in this biotope is substantially lower than that reported in commercial beds in New Zealand or elsewhere, so their effect on the surrounding community is uncertain but undoubtedly lower. However, *Atrina* is the unique, structural species in this biotope and its loss from the habitat would result in the loss of the biotope. The remaining characteristic species are either dependent on *Atrina* for hard substratum (i.e. epifauna), occur in a range of other sedimentary habitats or are mobile and ubiquitous. Therefore, *Atrina fragilis* is the focus of the assessment of the sensitivity of this biotope. The sensitivity of other characteristic species is discussed where relevant.

Resilience and recovery rates of habitat

Fan mussels (pen shells) (fan mussels) are vulnerable to over-exploitation due to their long life, slow growth, limited reproductive output and sporadic recruitment (Butler *et al*., 1993). Fan mussels (pen shells) are a commercially important food source but have been over-exploited worldwide, with the exception of Australia and New Zealand where catches are regulated. Over-exploitation combined with habitat loss, disease, trawling and anchoring, and suspended sediments from coastal activities has resulted in a drastic decline in wild populations of *Atrina* and *Pinna* in past decades (Chavez-Villabla *et al*., 2022).

Rapid shell repair in pinnids suggests high metabolic demand, that may result in reduced gamete production (Anon, 1999c; Butler *et al*., 1993), consistent with a long-lived species. Stirling *et al.* (2018) suggested that the larval phase had a long pelagic duration of up to four months in west Scotland but noted that the duration may be shorter in more southern or warmer regions. Nevertheless, Stirling *et al.* (2018) noted that while the long larval duration may increase the species' potential for dispersal, it may also increase mortality rates. Fertilization efficiency in patchy populations of low density may also be low as individuals may be too far apart to reproduce (Anon, 1999c; Hiscock *et al*., 2011). Poor recruitment may result in population subdivision due to a lack of gene flow over distance. Butler *et al.* (1993) mentioned evidence for genetic population subdivision in *Pinna bicolor* within the Gulf of St Vincent in South Australia, which suggested that effective dispersal was lower than expected. However, molecular studies have revealed a mixture of cryptic species, with widespread and/or diverse mtDNA lineages that suggest that population subdivision and dispersal are dependent on local conditions and vary between regions and species (Yu *et al*., 2004; Liu *et al*., 2011; Lemer *et al*., 2014; Hashimoto *et al*., 2021). Therefore, with the exception of embayments and inlets where larvae may be trapped, effective recruitment of *Atrina fragilis* may be poor and variable in comparison with other bivalve species (Anon, 1999c). However, surviving adults increase the possibility of fertilization and local recruitment. Anon (1999c) suggested that changes in factors that shorten the adult life of this species cannot be compensated for by an immediate reproductive response and recruitment.

In the Adriatic, Šimunović *et al.* (2001) concluded that the resident population of *Atrina fragilis* were self-sustaining in spite of trawl fishing and occasional hypoxic events, based on experimental bottom trawls in 17 cruises of the PIPETA Expedition between 1982 and 1994. These cruises recorded *Atrina fragilis* from 20% of 780 hauls in that period and averaged between ca 1 to ca 5000 individuals per km^2. In addition, numerous hauls included both juvenile and adult *Atrina fragilis* (Šimunović *et al.*, 2001). Fryganiotis *et al.* (2013) and Papoutsi & Galinou-Mitsoudi (2010) also reported a range of size classes, including juveniles, from the Thermaikos Gulf, Adriatic, which indicated that the populations studied were recruiting. However, Fryganiotis *et al.* (2013) recorded a density of 0.03 to 6.27 individuals/km in the Thermaikos Gulf, in the Adriatic. Furthermore, Fryganiotis *et al.* (2013; Fig 2) reported that the density of fan mussels in trawled areas (ca 0.03 individuals /km) was sparse compared to the areas in which bottom trawling was prohibited for 25 years (ca 5.5 individuals /km). Fryganiotis et al. (2013) suggested that 25 years was probably a time period that allows population recovery in this species. The largest known area of fan mussels in the UK, in the Sound of Canna, covered an area of at least 170 ha. The densest patches were estimated to be 2 -4 /m^2 (but ranged to 1-2 m^2) where the fan mussels occurred in clumps or scattered individuals (Howson *et al.*, 2012).

Stirling *et al.* (2016) developed species distribution models for *Atrina fragilis* in the waters of west Scotland. They reported that depth and habitat complexity (ruggedness) were important factors in the suitability of the habitat for *Atrina fragilis*, together with current speed and substratum type (the percentage of mud and gravel). In particular, they suggested that habitat complexity, either natural or artificial, protected the substratum and, hence, adults from the effects of fishing activities. Stirling *et al.* (2016) noted that *Atrina fragilis* was most abundant in the Sound of Canna, which had the highest habitat complexity (ruggedness) in their study. The Sound of Canna is a deep-sided channel deepened by glaciation with a complex benthic profile including glacial moraines and deep water dredge disposal site (Stirling *et al.*, 2016). They also predicted several other suitable sites for *Atrina fragilis* in the waters of Western Scotland (Stirling *et al.*, 2016).

It is possible that areas of habitat complexity provide refugia for *Atrina fragilis* populations from the effects of fishing activities and, together with the depth at which it can occur, it may be under-recorded at present. Nevertheless, the species has declined in abundance in the last 100 years, especially in inshore waters (Solandt, 2003). It was once regularly caught in trawls in the Celtic Sea with anecdotal records of large individuals and 'decks covered with fragments of their shells' (Solandt, 2003). The decline in the Mediterranean (Richardson *et al.*, 1999) and its loss from inlets in south-west England (Anon 1999c) suggest that any recovery from disturbance would be slow.

Amphiura filiformis is a small brittlestar, disc up to 10 mm in diameter, with very long arms (10x disc diameter) that lives buried in muddy sand. Muus (1981) showed the mortality of new settling *Amphiura filiformis* to be extremely high with less than 5% contributing to the adult population in any given year. Sköld *et al.* (1994) also commented on the high mortality and low rates of recruitment in this species. In Galway Bay populations (O'Connor *et al.*, 1983), small individuals make up ca 5% of the population in any given month, which also suggests the actual level of input into the adult population is extremely low. Muus (1981) estimated the lifespan of *Amphiura filiformis* to be 25 years based on oral width (which does not change with gonadal growth) with recruitment taking place at the 0.3 mm disc size. In very long-term studies of *Amphiura filiformis* populations in Galway Bay, a lifespan of some 20 years is possible (O'Connor *et al.*, 1983). *Amphiura filiformis* reaches sexual maturity after 2 years, breeds annually and, in the UK, one period of recruitment occurs in the autumn (Pedrotti, 1993). The species is thought to have a long pelagic life. Sköld *et al.* (1994) estimated the time lag between full gonads and settlement to be 88 days. This duration is comparable to the time period when pelagic larvae have been recorded in the plankton from July to November in one prior study, and August to December in another prior study (Fosshagen, 1965; Thorson, 1946, respectively, cited in Sköld *et al.*, 1994). A long planktonic life stage means this species is predicted to disperse over considerable distances. *Ophiocomina nigra* grows slowly and lives for up to 14 years (Hughes, 1998b). Juvenile *Ophiocomina* appears not to settle among adults. The Firth of Clyde populations studied by Gorzula (1977) were each dominated by a single size-class of animals, suggesting that each *Ophiocomina* bed is formed by a single settlement of juveniles, which thereafter receives little or no recruitment.

Uthicke *et al.* (2009) reviewed 'boom-bust' dynamics in populations of echinoderms. The reported significant increases in population density of *Amphiura filiformis* in the southern North Sea, the Skagerrak and Kattegat area of the North Sea between the 1930s and 1980s and *Amphiura chaeji* in Loch Eil and Loch Linnhe, Scotland from the 1960s to the 1970s onwards, due to eutrophication and/or overfishing of their predators. They suggested that the high survival of settled recruits, adults or larvae prompted an increase in the population that resulted in increased fertilization efficiency, higher larval production, and a sustained, stable high population density.

Resilience assessment. The brittlestar present in the biotope are typical of the surrounding sediment but may benefit from the localised changes in water flow and organic content of biodeposits in the vicinity of the *Atrina fragilis*, especially areas of highest *Atrina* density. *Ophiura* and *Ophiocomina* spp. are mobile and may recruit from the surrounding area but if removed may take several years to return due to the sporadic recruitment typical of echinoderms. *Amphiura* spp. demonstrates stable populations but may alosa take 2-10 years ('Medium' resilience) to recover if removed. However, *Atrina fragilis* is the most unique characteristic and structuring species in this biotope.

The decline of *Atrina fragilis* in UK inshore waters and the Mediterranean over the last hundred years suggests that recovery is slow (Richardson *et al.*, 1999; Solandt, 2003). Their long life, slow growth, limited reproductive output, low fertilization efficiency in sparse populations, and sporadic recruitment (Butler *et al.*,1993; Anon, 1999c) are also likely to hamper their ability to recover from disturbance and population mortality. The increased numbers of records from deep waters could suggest that *Atrina fragilis* is under-recorded in offshore areas, which themselves could provide a reservoir for recruitment to the inshore areas but there is no evidence to support this idea. Nevertheless, recruitment to and recovery of populations is likely to be prolonged. The large area of fan mussels in the Sound of Canna may have resulted from a single successful recruitment event. The presence of viable larvae in the Sound of Canna and the single late-stage larva in the Isles of Scilly (Stirling *et al.*, 2018) suggests that reproductively active populations exist in these areas and have the potential to recruit internally and to their surrounding areas. But larval/juvenile mortality is probably high, and juveniles and adults require areas protected from physical disturbance to survive. Therefore, recovery from any loss of the population of *Atrina fragilis* (i.e. a reduction in the extent or abundance, resistance is 'Medium' or 'Low') may take up to 25 years where the populations are sparsely distributed (e.g. in the UK). Hence, resilience is assessed as **Low'**. However, where the population is severely reduced in abundance or extent (i.e. resistance is 'None') resilience is assessed as **'Very low'**.

Hydrological Pressures

Resistance	Resilience	Sensitivity

Temperature increase (local)	Medium Q: Low A: NR C: NR	Low Q: Medium A: Medium C: Medium	Medium Q: Low A: Low C: Low

No information on temperature tolerance in *Atrina fragilis* was found, although it has been suggested that changes in seawater temperature are likely to affect larval recruitment patterns (Anon., 1999c). *Atrina* spp. are annual episodic or protracted spawners with one or two periods of spawning often associated with temperature and food supply (Chávez-Villalba *et al.*, 2022). Stirling *et al.* (2018) concluded that *Atrina fragilis* in west Scotland exhibited peaks of spawning in summer and winter with low levels of spawning throughout the year. Wang *et al.* (2017) reported that *Atrina pectinata* in the Bohai Sea, China had a single spawning event; gametogenesis began in October, completed between June and July, and spawning occurred in August when temperatures and food availability were at their highest.

The tropical pen shell *Atrina maura* was found to reach maturity more quickly at higher temperatures, taking only one month (normal maturation at lower temperatures of 20°C takes two months). However, with higher temperatures, oocytes are of poor quality than at cooler temperatures (Rodriguez-Jaramillo, 2001). Similarly, Leyva-Valencia *et al.* (2001) reported that activity, ingestion rates, scope for growth, and growth rates were highest at 29°C in juvenile *Atrina maura* and suggested an optimal temperature of 29°C or higher. No mortality occurred between 19 and 30°C, 10% mortality at 32°C and the authors reported a 96-hour LD_{50} of 33.2°C. In flume experiments, Arrieche *et al.* (2010) noted that specimens of the tropical *Atrina maura* survived daily fluctuations of 8°C between 13°C and 33°C. OBIS (July 2022) included records of *Atrina fragilis* from sites where sea surface temperature ranged from 5 to 20°C although most records were recorded from 10-15°C.

Kröncke *et al.* (2011) reported an increase in abundance and regional changes in the distribution of various species with a southern distribution in the North Sea in 2000 and suggested the changes were largely associated with an increase in sea surface temperature, primary production and, thus, food supply. The authors suggested that the increase in annual average temperature was about 1.1°C. *Amphiura filiformis* was observed to have decreased in abundance. In Galway Bay, long-term recordings of water temperature at a site of high-density aggregations of *Amphiura filiformis* showed the species is subject to annual variations in temperature of about 10°C (O'Connor *et al.*, 1983). Increases in temperature may affect growth and fecundity. Muus (1981) showed that juvenile *Amphiura filiformis* are capable of much higher growth rates in experiments with temperatures between 12 and 17°C.

Sensitivity assessment. Subtidal species such as *Atrina fragilis* are likely to exhibit lower temperature tolerance than intertidal species and are not likely to be resistant to rapid temperature change indicated in this benchmark. However, they occur from the Mediterranean to the Shetland Isles and are probably resistant to the range of temperatures that occur within that range. Therefore, *Atrina fragilis* is probably resistant to a long-term change in temperature of 2°C for a year (see benchmark). But shallow subtidal and sublittoral fringe populations may be adversely affected by short-term changes in temperature of 5°C for a month (see benchmark). Therefore, a resistance of **'Medium'** is recorded to represent the loss of the upper shore or shallow populations, albeit with 'Low' confidence. Resilience is probably **'Low'** and sensitivity is assessed as **'Medium',** albeit with 'Low' confidence.

Temperature decrease (local)	Medium Q: Low A: NR C: NR	Low Q: Medium A: Medium C: Medium	Medium Q: Low A: Low C: Low

No information on temperature tolerance in *Atrina fragilis* was found, although it has been suggested that changes in seawater temperature are likely to affect larval recruitment patterns (Anon., 1999c). No information on temperature tolerance in *Atrina fragilis* was found, although it has been suggested that changes in seawater temperature are likely to affect larval recruitment patterns (Anon., 1999c). *Atrina* spp. are annual episodic or protracted spawners with one or two periods of spawning often associated with temperature and food supply (Chávez-Villalba *et al.*, 2022). Stirling *et al.* (2018) concluded that *Atrina fragilis* in west Scotland exhibited peaks of spawning in summer and winter with low levels of spawning throughout the year. Wang *et al.* (2017) reported that *Atrina pectinata* in the Bohai Sea, China had a single spawning event; gametogenesis began in October, completed between June and July, and spawning occurred in August when temperatures and food availability were at their highest.

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Holme (1967) reported the absence of *Amphiura filiformis* from samples taken from Weymouth Bay and Poole Bay, England, after severe winter temperatures (4 and 5°C, respectively, below the mean for about a month). In Galway Bay, long-term recordings of water temperature at a site of high-density aggregations of *Amphiura filiformis* showed the species is subject to annual variations in temperature of about 10°C (O'Connor *et al.*, 1983). However, echinoderms, including *Amphiura filiformis*, in the North Sea, seem periodically affected by winter cold. A population at 27 m depth off the Danish coast was killed by the winter of 1962-63 (Muus, 1981) and a population at 35-50 m depth in the inner German Bight was killed in the winter of 1969-1970 and a new population was not re-established until 1974 (Gerdes, 1977). Ursin (1960, cited in Gerdes, 1977) suggests that *Amphiura filiformis* does not occur in areas with winter temperatures below 4°C although in Helgoland waters it can tolerate temperatures as low as 3.5°C.

Sensitivity assessment. Subtidal species such as *Atrina fragilis* are likely to exhibit lower temperature tolerance than intertidal species and are not likely to be resistant to rapid temperature change indicated in this benchmark. However, they occur from the Mediterranean to the Shetland Isles and are probably resistant to the range of temperatures that occur within that range. *Amphiura* spp. also have a wide distribution but may not be able to survive extreme decreases in temperature, although they may only be vulnerable in shallow waters. Therefore, *Atrina fragilis* is probably resistant to a long-term change in temperature of 2°C for a year (see benchmark). But shallow subtidal and sublittoral fringe populations may be adversely affected by short-term changes in temperature of 5°C for a month (see benchmark). Therefore, a resistance of **'Medium'** is recorded to represent the loss of the upper shore or shallow populations, albeit with 'Low' confidence. Resilience is probably **'Low'** and sensitivity is assessed as **'Medium',** albeit with 'Low' confidence.

Salinity increase (local)	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Atrina fragilis occurs subtidally at full salinity but shallow subtidal populations may experience variable salinity. Dan Minchin (pers. comm.) suggested that *Atrina fragilis* may be exposed to reduced or variable salinities for brief periods. A tropical pen shell *Atrina maura*, was found to be halotolerant between 16-50 (Leyva-Valencia *et al.*, 2001). OBIS (July 2022) included records of *Atrina fragilis* from sites where sea surface salinity ranged from 30-35. *Atrina fragilis* is probably not resistant to hypersaline (>40) conditions for a year.

Echinoderms, such as *Amphiura filiformis* and *Ophiocomina spp.*, are stenohaline owing to the lack of an excretory organ and a poor ability to osmo- and ion-regulate (Stickle & Diehl, 1987; Russell, 2013). A review by Russell (2013) confirmed that none of the echinoderm species relevant in this assessment occurs in hypersaline conditions. Pagett (1981) suggested that localised physiological adaption to reduced or variable salinities may occur in nearshore areas subject to freshwater runoffs. However, individuals in these biotopes are unlikely to experience variable salinities, and resident species unlikely to be adapted to variation in salinity, as suggested by the results given by Pagett (1981).

Sensitivity assessment. The evidence suggests that exposure to hypersaline conditions may exclude the resident echinoderms. However, no evidence of the effect of hypersaline conditions on *Atrina* spp. was found. Roberts *et al.* (2010b) noted that the effects of hypersaline (brine) effluents were limited to within 10s of metres of the outfall. This biotope occurs at depth (100-150 m) (JNCC, 2022) in wave-exposed conditions in weak to moderately strong currents (Stirling *et al.*, 2016) so mixing is probably good. In the unlikely event that this biotope is exposed to hypersaline conditions (>40) then *Atrina* may be adversely affected. However, in the absence of direct evidence, no assessment is made.

Salinity decrease (local)	Low Q: High A: Medium C: Medium	Low Q: Medium A: Medium C: Medium	High Q: Medium A: Medium C: Medium

Atrina fragilis occurs subtidally at full salinity but shallow subtidal populations may experience variable salinity. Dan Minchin (pers. comm.) suggested that *Atrina fragilis* may be exposed to reduced or variable salinities for brief periods. A tropical pen shell *Atrina maura* was found to have a wide range of halotolerance, from 16-50 (Leyva-Valencia *et al.*, 2001). OBIS (July 2022) included records of *Atrina fragilis* from sites where sea surface salinity ranged from 30 to 35.

Kurihara *et al.* (2018) exposed juvenile *Atrina pectinata* (3.2 to 3.6 cm in length) to both rapid and gradual changes in a range of salinities at 12°C and 24°C in the laboratory. They reported that all juveniles survived 83 hours of gradual exposure to >=21.1 ppt at 24°C but all died after exposure <=18.6 ppt for <20 hours (with one exception). At 12°C, all juveniles survived 83 hours of gradual exposure to >=17.2 ppt but all died after <33.6 hours at <=15.1 ppt. Rapid exposure to reduced salinity, by direct transfer of specimens into the required salinities, decreased the survival time. At 24°C, all juveniles (except one) survived rapid exposure to >=17.7 ppt for 140 hours but all died rapid exposure to <=15.9 ppt for 13 to 20 hours. At 12°C, all specimens survived 140 hours at >=23.1 ppt and all died at <=21.3 ppt. Kurihara *et al.* (2018) concluded that salinity tolerance was lower at 24°C than 12°C but that the difference was reduced during rapid changes in salinity. Kurihara *et al.* (2018) also suggested that juveniles were more sensitive to rapid changes in salinity than adults and report a period study that showed that adult *Atrina pectinata* survived rapid exposure to >15 ppt but died after exposure to <15 ppt for 96 hours at 28°C.

Echinoderms, such as *Amphiura filiformis*, are stenohaline owing to the lack of an excretory organ and a poor ability to osmo- and ion-regulate (Stickle & Diehl, 1987; Russell, 2013). However, *Amphiura filiformis* was recorded in hyposaline conditions in the Sado estuary in Portugal (Monteiro-Marques, 1982 cited in Russell, 2013) where the salinity was 25.5‰, and in the Black Sea where it tolerated 8.9‰ (Russell, 2013). Russell (2013) reported that *Ophiocomina nigra* and *Ophiura albida* can tolerate 27.6‰ and 20‰ in experiments, respectively. Pagett (1981) suggested that localised physiological adaption to reduced or variable salinities may occur in nearshore areas subject to freshwater runoffs. However, individuals in this circalittoral biotope are unlikely to experience variable salinities, and resident species are unlikely to be adapted to variation in salinity, as suggested by the results given by Pagett (1981).

Sensitivity assessment. *Atrina fragilis* may be able to tolerate short periods of exposure to decreases in salinity but is probably not resistant to changes in salinity (e.g. from 'full' to 'reduced') for a year (see benchmark) based on the evidence from *Atrina pectinata*. Therefore, resistance is assessed as **'Low'**, resilience as **'Low'** and sensitivity as **'High'**.

Water flow (tidal current) changes (local)	High Q: Low A: NR C: NR	High Q: High A: High C: High	Not sensitive Q: Low A: Low C: Low

Atrina fragilis is known from weak (<0.5 m/s) to moderately strong currents (0.5-1.5 m/s). For example in Knightstown, Valentia Island, the population is exposed to >2 knots (ca >1 m/s) on spring tides (Dan Minchin pers. comm.). Stirling *et al.* (2016) reported that Scottish records occurred in areas with peak currents, during a mean spring tide, of 0.65 m/s on average and a maximum of 1.24 m/s. Stirling *et al.* (2016) suggested that extremely high currents (i.e. >2-3 m/s) would probably prevent the settlement of spat and the resultant resuspension of sediments would impact adults. Changes in current patterns may affect larval recruitment (Anon., 1999c). Arrieche *et al.* (2010) found that *Atrina maura* grew significantly larger above a flow rate of 7.3 cm/s in flume experiments. Arrieche *et al.* (2011) reported that ingestion rate, absorption rate, and scope for growth were highest at 1.6 cm/s (0.16 m/s) in *Atrina maura* under laboratory conditions. Scope for growth, ingestion and absorption rates were correlated with food supply at 0.8 and 1.6 cm/s and respiratory rate was positively correlated at 2.5 cm/s (0.25 m/s).

Amphiura filiformis respond rapidly to currents by extending their arms into the water column to feed. Under laboratory conditions, they were shown to maintain this vertical position at currents of 0.3 m/s (Buchanan, 1964). *Amphiura filiformis* feed on suspended material in flowing water but change to deposit feeding in stagnant water or areas of very low water flow (Ockelmann & Muus, 1978). Food requirements probably set a lower limit on the current regime of areas able to support brittlestars. *Amphiura filiformis* has also been reported on the Northumberland coast, the UK where tidal currents ranged from surface speeds of 0.65 m/s at springs to 0.4 m/s at neaps, on a flood tide. Bottom residual currents were much weaker than near-surface, reaching a maximum of 0.7 m/s (Jones, 1979, cited in Birchenough & Frid, 2009).

Sensitivity assessment. However, an increase of 0.1-0.2 m/s for one year (the benchmark) may not be significant. A decrease in flow to 'very weak' or 'negligible' may be detrimental as water flow is important to provide a food supply for suspension feeders, as well as oxygenate the water column, especially in isolated waters. A reduction in food supply may well decrease growth and reproduction in this species, although as the species is long-lived, a change for one year (see benchmark) may not result in mortality. Therefore, a resistance of **'High'** is recorded, albeit with

'Low' confidence. Hence, resilience is **'High'** and the species is assessed as **'Not sensitive' at the benchmark level.**

Emergence regime changes	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

This biotope was recorded in the circalittoral from 100-150 m in depth. Therefore, it is unlikely to be exposed to the air at low tide.

Wave exposure changes (local)	High Q: A: C:	High Q: High A: High C: High	Not sensitive Q: A: C:

Atrina fragilis occurs in sheltered or very sheltered waters (Anon 1999c; Butler *et al.* 1993) and can burrow into the substratum if partly uncovered by wave action or storms (Yonge, 1953). However, a prolonged increase in wave action could remove some individuals from the substratum, which would not then be able to survive to re-establish themselves. Juveniles may be removed from sediment more easily than adults.

García-March *et al.* (2007) reported that a bed of *Pinna nobilis* at 13 m Moraira Bay, in the Spanish Mediterranean suffered 13.6% mortality after a major storm in 2003 (greater than any recorded in the prior nine years), in which adults were broken or dislodged and died due to injury or exhaustion. The survivors were notably smaller (on average) than the dead specimens. García-March *et al.* (2007) examined the drag force on *Pinna nobilis* shells from this site and a shallow (6 m) site. They reported that the threshold for dislodgment in the deep population was ca 45 Newtons (N), which could be experienced at a water species of 1.25 m/s in large individuals with a relatively large surface area. The shallow area was populated by significantly smaller individuals, orientated with their dorsal-ventral surface in line with water flow, which reduced their mean drag force. García-March *et al.* (2007) suggested that the shallow site regularly experienced wave-mediated flow higher than the deep site, which selected the population for a smaller size (on average) while depth allowed individuals to grow larger. In the extreme storm, the deep site experienced more damage because it exposed the deeper water to higher than normal water speeds while the largest wave experienced during the storm broke before reaching the shallow site.

Amphiura filiformis is found in sheltered habitats characterized by fine muddy sandy sediments and low wave exposure. The species is unlikely to be resistant to increases in wave exposure because strong wave action can resuspend the sediment and break up and scatter *Amphiura filiformis*.

Sensitivity assessment. This biotope SS.SMu.CFiMu.AtrEch is recorded from very wave exposed to moderately exposed conditions but at a depth (100-150 m) where the resultant wave flow is probably attenuated by depth. For example, Stirling *et al.* (2016) reported that Scottish records occurred in areas with peak currents, during a mean spring tide, of 0.65 m/s on average and a maximum of 1.24 m/s. Similarly, the effects of extreme storms may also be reduced at this depth. Nevertheless, a change in significant wave height of 3-5% (the benchmark) is unlikely to be significant, especially as depth. Therefore, resistance is assessed as **'High'**, resilience as **'High'** and this species is probably **'Not sensitive' at the benchmark level.**

Chemical Pressures

	Resistance	Resilience	Sensitivity
Transition elements & organo-metal contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Anon (1999c) suggested that *Atrina fragilis* may be affected by pollutants such as TBT (tri-butyl tin). Reid & Brand (1989) describe kidney gigantism and nephroliths (calcium or iron granules) in *Pinna bicolor*. Their role in removing excess calcium or heavy metals and potential detoxification is unclear. Ward & Young (1983) examined changes in epifauna of *Pinna bicolor* due to heavy metal contamination in Spence Gulf, South Australia. They stated that *Pinna bicolor* was tolerant of high concentrations of heavy metals in sediments near a lead smelter and contained high body loads of heavy metals. The occurrence of populations of this species in heavy metal contaminated sediment suggests that it is 'Not sensitive'. However, the body burden of *Pinna bicolor* was not given and no citation was provided for the information. The studied population may represent a localised adaptation.

Gongora-Gomez *et al.* (2018) reported that *Atrina maura* from an aquaculture farm in the Gulf of California accumulated heavy metals in their body tissues but no information on any adverse effects was reported. They reported mean body burdens of ca 0.064 µg/g dwt Hg, ca 485.66 µg/g dwt Zn, 18.15 µg/g dwt Cd and 2.31 µg/g wwt Pb in soft tissues, and ca 0.058 µg/g dwt Hg, 64.83 µg/g dwt Zn, and1.82 µg/g dwt Cd in muscle.

Anon (1999c) suggested that *Atrina fragilis* may be affected by pollutants such as TBT (Tri-butyl Tin). Inoue *et al.* (2006) identified a range of TBT contamination of 0.009 to 0.095 µg/g wwt in *Atrina pectinata japonica* collected from northern Kyushu, Japan. Inoue *et al.* (2007) reported that exposure of *Atrina pectinata japonica* to 1 µg/l TBT for 72 hours in the laboratory reduced energy metabolism but did not cause mortality during the experiment.

Bryan (1984) reported that early work showed that echinoderm larvae were intolerant of heavy metals, e.g. the intolerance of larvae of sea urchin *Paracentrotus lividus* to copper (Cu) was used to develop a water quality assessment. Adult echinoderms are known to be efficient concentrators of heavy metals including those that are biologically active and toxic (Hutchins *et al.*, 1996) but there is no information available regarding the effects of this bioaccumulation. Studies by Deheyn & Latz (2006) at the Bay of San Diego found that heavy metal accumulation in brittlestars occurs both through dissolved metals as well as through diet, to the arms and disc, respectively. Similarly, Sbaihat *et al.* (2013) measured concentrations of heavy metals (Cu, Ni, Cd, Co, Cr and Pb) in the body of *Ophiocoma scolopendrina* collected from the Gulf of Aqaba, and found that most concentration was found in the central disc rather than arms and no simple correlations could be found between contaminant and body length.

Overall, information on the effects of heavy metal contamination and TBT exposure in *Atrina* is limited to sublethal effects and no information on the native *Atrina fragilis* was found. Similarly, while brittlestars may accumulate heavy metals no effects of exposure were found. Therefore, the evidence is probably not sufficient for an assessment.

Hydrocarbon & PAH contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Echinoderms were not resistant to the toxic effects of oil, likely because of the large area of the epidermis (Suchanek, 1993), and tend to be very sensitive to various types of marine pollution (Newton & McKenzie, 1995). In a study of the effects of oil exploration and production on benthic communities, Olsgard & Gray (1995) found *Amphiura filiformis* to be very intolerant of oil pollution. During monitoring of sediments in the Ekofisk oilfield, Addy *et al.* (1978) suggested that reduced abundance of *Amphiura filiformis* within 2-3 km of the site was related to discharges of oil from the platforms and to physical disturbance of the sediment. Brittlestars host symbiotic sub-cuticular bacteria (Kelly & McKenzie, 1995). After exposure to hydrocarbons, loadings of such bacteria were reduced indicating possible sub-lethal stress to the host (Newton & McKenzie, 1995).

However, **no evidence** of the effects of hydrocarbons and PAHs on *Atrina* spp. was found.

Synthetic compound contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

In laboratory experiments, Smith (1968) found the concentration of BP1002 (the detergent used in the *Torrey Canyon* oil spill clean-up) needed to kill the majority of brittlestar *Ophiocomina nigra* was 5 ppm. Dahllöf *et al.* (1999) studied the long-term effects of tri-n-butyl-tin (TBT) on the function of a marine sediment system. TBT spiked sediment was added to sediment that already had a TBT background level of approximately 27 ng/g (83 pmol TBT per g) and contained *Amphiura* spp. and several species of polychaete. Within two days of treatment with a TBT concentration above 13.7 µmol/m² all species except the polychaetes had crept up to the surface and after six weeks these fauna had started to decay. Thus, contamination from TBT is likely to result in the death of some non-resistant species such as brittlestars. However, Walsh *et al.* (1986) observed inhibition of arm regeneration in another brittlestar, *Ophioderma brevispina*, following exposure to TBT at levels between 10 ng/l and 100 ng/l. Loizeau & Menesguen (1993), found that 8-15% of the PCB burden in dab, *Limanda limanda*, from the Bay of Seine could be explained by ophiuroid consumption. Thus, *Amphiura* communities may play an important role in the accumulation, remobilization and transfer of PCBs and other sediment contamination to higher trophic levels.

However, **no evidence** of the effects of 'synthetic compounds' on *Atrina* spp. was found.

Radionuclide contamination	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Adult echinoderms are known to be efficient concentrators of radionuclides (Hutchins *et al.*, 1996). However, no information concerning the effects of such bioaccumulation was found. **No evidence** of the effects of radionuclides on *Atrina* spp. was found.

Introduction of other substances	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

No evidence of the effects of 'other substances' on *Atrina* spp. was found.

De-oxygenation	Low Q: Medium A: Medium C: Medium	Low Q: High A: High C: High	High Q: Medium A: Medium C: Medium

Bivalves are generally resistant to hypoxia and can respire anaerobically. Butler *et al.* (1993) state that *Pinna bicolor* and *Pinna nobilis* remain active at low oxygen concentrations (no value was given) and open their valves widely. Neither species stop pumping or respire anaerobically. Jaklin & Zahtila (1990, cited by Šimunović *et al*., 2010) reported a mass mortality in the northern Adriatic due to an anoxic event in November 1989. Subsequent diver surveys in January 1990 found many empty shells of *Atrina fragilis* in an area previously populated by 1-2 fan mussels per m^2 but suggested that 10% of the population survived. Nagasoe *et al.* (2020) examined the effects of hypoxia in juveniles, 1-year-old (mean of 9.4 cm in length) and 2 years old (mean of 14.6 cm in length) *Atrina japonica* for 96 hours (4 days) under laboratory conditions. As the oxygen concentration decreased, most individuals had open exhalent and inhalent siphons at or below 2 mg/l dissolved oxygen (DO) after 12 hours and under continued low oxygen the specimens emerged from the sediment exposing more than half their shell length. Nagasoe *et al.* (2020) reported 96-hour LD_{50}s of 0.84 mg/l DO for one-year-olds and 1.80 mg/l DO for two years olds. they concluded that two years old were more susceptible to hypoxic conditions than one-year-olds probably because of their productive condition since 96.7% of two years olds were ripe in contrast to only 7.6% of one-year-olds. Masato *et al.* (2017) exposed *Atrina lischkeana* from Kyushu, Japan, with an average length of 13.6 cm to <3.0 mg/l DO for 6 hours a day for 30 days. The animals moved upwards in the substratum under low oxygen levels and then downwards in aerobic conditions for 19 days but the response stopped after 20 days. No mortality was observed but the glycogen content of specimens decreased and Masato *et al.* (2017) suggested the specimen were exhausted by their movement relative to the sediment. Masato *et al.* (2017) also noted that prior studies had reported significant mortality of *Atrina lischkeana* exposed to 0.36 mg/l DO for 6 hours/day for 31 days and 50% mortality after exposure to 14.9% oxygen (ca 4.8 mg/l DO) continuously for 72 hours.

Stachowitsch (1984) observed the mass mortality of benthic organisms in the Gulf of Trieste, northern Adriatic Sea, caused by the onset of severe hypoxia in the near-bottom water. A wide variety of organisms were affected, including burrowing invertebrates, sponges, and the brittlestar *Ophiothrix quinquemaculata*. However, *Amphiura filiformis* was reported as a species resistant to moderate hypoxia (Diaz & Rosenberg, 1995). In experiments exposing benthic invertebrates to decreasing oxygen levels, *Amphiura filiformis* only left its protected position in the sediment when oxygen levels fell below 0.85 mg/l, and was able to survive for several weeks (Rosenberg *et al*., 1991). This escape response increases predation risk. Mass mortality of *Amphiura filiformis* was observed during severely low oxygen events (<0.7 mg/l) (Nilsson, 1999).

Sensitivity assessment. While *Amphiura* spp. may be resistant of hypoxia at the benchmark level the limited evidence suggests that *Atrina fragilis* is likely to suffer significant mortality in hypoxic conditions (e.g. below 2 mg/l for one week) and severe mortality in anoxic conditions. Therefore, resistance is assessed as **'Low'**. Hence, resilience is probably **'Low'** and sensitivity is assessed as **'High'.**

Nutrient enrichment	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Pinnids are mainly found in sheltered oligotrophic (low nutrient) waters (Butler *et al*., 1993), and they filter continuously, presumably an adaptation to low food availability. A small population of *Atrina fragilis* was recorded near a sewage discharge in Dingle Harbour (Dan Minchin pers comm.). An increase in nutrients is likely to increase phytoplankton production in the short term, which may benefit larvae and juvenile *Atrina*. But excessive nutrient enrichment may lead to the development of algal blooms, and hypoxic conditions in the benthos (see deoxygenation above).

Significant increases in *Amphiura* spp. populations have been associated with eutrophication (Uthicke *et al.*, 2009). However, **no evidence** of the direct or indirect effects of changes in nutrients (e.g. nitrogen or phosphates) on *Atrina* spp. was found.

Organic enrichment	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Pinnids are mainly found in sheltered oligotrophic (low nutrient) waters (Butler *et al.*, 1993), and they filter continuously, presumably an adaptation to low food availability. A small population of *Atrina fragilis* was recorded near a sewage discharge in Dingle Harbour (Dan Minchin pers comm.). Organic enrichment is likely to result in hypoxic sediment and an increase in opportunistic infauna, together with an increase in suspended sediments and siltation, which may be detrimental. However, *Atrina fragilis* is recorded from detritic bottoms with terrigenous ooze in the Adriatic (Fryganiotis *et al.*, 2013). Significant increases in *Amphiura* spp. populations have been associated with eutrophication (Uthicke *et al.*, 2009). However, the evidence of the effect of organic enrichment on *Atrina* was not adequate and no sensitivity assessment was made.

Physical Pressures

	Resistance	Resilience	Sensitivity
Physical loss (to land or freshwater habitat)	None Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

All marine habitats and benthic species are considered to have a resistance of **None** to this pressure and to be unable to recover from a permanent loss of habitat (Resilience is **Very Low**). Sensitivity within the direct spatial footprint of this pressure is therefore **High**. Although no specific evidence is described, confidence in this assessment is ‘High’, due to the incontrovertible nature of this pressure.

Physical change (to another seabed type)	None Q: High A: High C: High	Very Low Q: High A: High C: High	High Q: High A: High C: High

If the sediment that characterizes the biotopes was replaced with rock substrata, this would represent a fundamental change to the physical character of the biotope. The characterizing species would no longer be supported and the biotopes would be lost and/or reclassified. Therefore, resistance to the pressure is considered '**None'**, and resilience '**Very low'**, given the permanent nature of this pressure. Sensitivity has been assessed as '**High'**. Although no specific evidence is described, confidence in this assessment is ‘High’, due to the incontrovertible nature of this pressure.

Physical change (to another sediment type)	High Q: Low A: NR C: NR	High Q: High A: High C: High	Not sensitive Q: Low A: Low C: Low

The Pinnidae live embedded in soft substrata but with enough underlying gravel to provide attachment for their byssus threads (Yonge, 1953). *Atrina fragilis* has been recorded from a variety of sediment types e.g. muddy sands to clean sands, often mixed with gravels and shell (see habitat preferences above). For example, Fryganiotis *et al.* (2013) stated that this species was characteristic of 'detritic' bottoms, sandy bottoms and terrigenous ooze sediments, while Šimunović *et al.* (2001) reported that it was most abundant on sand-silt-clay sediments and clayey 'relict' sand. Howson *et al.* (2012) reported that the fan mussel bed in the sound of Canna occurred on mixed muddy sand with cobble, gravel, shell debris and occasional boulders but that one station with dense *Atrina fragilis* occurred on rippled sand with burrows. Furthermore, Stirling *et al.* (2016) reported that *Atrina fragilis* was recorded from 0.6% to 74% mud (with a mean of 32%) and 0.2%-62% gravel (with a mean of 12%) in the waters of west Scotland. Their model predicted the highest abundances in the range of 20-60% mud but approx. <20% gravel.

Sensitivity assessment. This biotope is recorded from mud and mixed sediments (JNCC, 2022) and *Atrina fragilis* from muddy sand and clean sands mixed with gravel and shell. *Amphiura filiformis* has been recorded in silty mud to mixed sediment (with stones and shells) (Tillin & Tyler-Walters, 2014b). Therefore, this *Atrina fragilis* dominated biotope is probably resistant to a change in one Folk class (see benchmark), and resistance is assessed as **'High'**. Hence, resilience is **'High'** and the species is assessed as **'Not sensitive'** at the benchmark level.

Habitat structure changes - removal of substratum (extraction)	None Q: Medium A: Medium C: Medium	Very Low Q: Medium A: Medium C: Medium	High Q: Medium A: Medium C: Medium

Sedimentary communities are likely to be highly intolerant of substratum removal, which will lead to partial or complete defaunation, expose underlying sediment which may be anoxic and/or of a different character and lead to changes in the topography of the area (Dernie *et al.*, 2003). Any remaining species, given their new position at the sediment/water interface, may be exposed to unsuitable conditions. Newell *et al.* (1998) stated that the removal of 0.5 m depth of sediment was likely to eliminate benthos from the affected area. Some epifaunal and swimming species may be able to avoid this pressure. However, the removal of sediment to a depth of 30 cm is likely to remove the entire population of fan mussels in the affected area, together with other species that occur at the surface and within the upper layers of sediment. Therefore, a resistance of **'None'** is recorded. Resilience is probably **'Very low'** so sensitivity is assessed as **'High'**.

Abrasion/disturbance of the surface of the substratum or seabed	Low Q: High A: Medium C: Medium	Low Q: Medium A: Medium C: Medium	High Q: Medium A: Medium C: Medium

Atrina fragilis has a fragile shell, damaged easily by anchor impact, or trampling by bathers or fish predation. It is probably adapted to such damage as the mantle and ctendia can be withdrawn into the remainder of the shell, and the damaged edge of the shell can repair quickly, e.g. at ca 1 cm /day in *Pinna carnea* (Yonge, 1953; Solandt, 2003). *Atrina* can burrow vertically but cannot 'right' itself if removed from the sediment and laid on its side (Yonge, 1953). Specimens removed from the sediment by a passing trawl, mooring chain etc. are unlikely to be able to reburrow.

Scallop dredging and demersal trawling have been implicated in the decline in populations of this species (Anon, 1999c; Hall-spencer *et al.*, 1999; Solandt, 2003; Šimunović *et al.*, 2001; Fryganiotis *et al.*, 2013). Solandt (2003) noted anecdotal records where 'considerable fragments of *Atrina* shells were collected by scallop trawlers', and large individuals caught in the Celtic Sea in the 1970s with 'decks covered with the broken fragments of this species'. Solandt (2003) also reported anecdotal records from the diving community of considerable numbers of *Atrina fragilis* found in areas where scallop trawlers and dredgers cannot set gear. Rapido trawling for scallops (a form of beam trawl) in the Gulf of Venice resulted in the removal of organisms from the top 2 cm of sediment and an 87% reduction in *Atrina fragilis* abundance in the trawl tracks. Some specimens were speared on the trawl teeth and pulled from the sediment (Hall-Spencer *et al.*, 1999). Anon (1999c) suggested that the destruction of a population of *Atrina fragilis* off Glengad Head, Ireland after 1975 was caused by scallop dredging. In the Adriatic queen scallop (*Aequipecten opercularis*) trawl fishery, *Atrina fragilis* incurred more damage as a result of the fishing and sorting process than any other species of bycatch (Pranovi *et al.*, 2001). In the Adriatic, Fryganiotis *et al.* (2013; Fig 2) reported that the density of fan mussels in trawled areas (ca 0.03 individuals /km) was sparse compared to the areas in which bottom trawling was prohibited for 25 years (ca 5.5 individuals /km). Furthermore, Stirling *et al.* (2016) used species distribution models, based on existing records, to examine the habitat preferences of *Atrina fragilis* and predict suitable habitat in the waters of the west coast of Scotland. Stirling *et al.* (2016) identified depth and habitat complexity (bathymetric ruggedness) as the most important determinants of distribution followed by current speed, and substratum type (percentage of mud and gravel), while aspect had less importance. In particular, they suggested that habitat complexity, either natural or artificial, protected the substratum and, hence, adults from the effects of fishing activities. Stirling *et al.* (2016) noted that *Atrina fragilis* was most abundant in the Sound of Canna, which had the highest habitat complexity (ruggedness) in their study. The Sound of Canna is a deep-sided channel deepened by glaciation with a complex benthic profile including glacial moraines and deep water dredge disposal site (Stirling *et al.*, 2016).

Ramsay *et al.* (1998) suggested that *Amphiura* spp. may be less susceptible to beam trawl damage than other species like echinoids or tube-dwelling amphipods and polychaetes. For example, Bergman & Hup (1992) found that beam trawling in the North Sea had no significant direct effect on small brittlestars. Holtmann *et al.* (1996) reported a decrease in the abundance of the fragile burrowing heart urchins and the brittlestar *Amphiura filiformis* in areas of the southern North Sea between 1990 and 1995. These trends suggest that fishing activity may have been the main cause of these changes. Several species of brittlestar were reported to increase in abundance in trawled areas (including *Ophiocomina nigra*), however, Bradshaw *et al.* (2002) noted that the relatively sessile *Ophiothrix fragilis* decreased in the long-term in areas subject to scallop dredging. Bradshaw *et al.* (2002) also noted that the brittlestars *Amphiura filiformis* had increased in abundance in a long-term study of the effects of scallop dredging in the Irish Sea. Uthicke *et al.* (2009) also suggested that the removal of predators by fishing activities, together with eutrophication, may have contributed to the long-term increase in *Amphiura filiformis* population density in the southern North Sea.

Sensitivity assessment. The effect of fishing activities on brittlestars varies depending on the type of gear and the species. The evidence suggests that *Atrina fragilis* can survive low levels of abrasion e.g. trampling in the shallow sublittoral and possibly pots and creels that damage the exposed shell. However, any passing chains or fishing gear that could remove individuals, or objects placed on the substratum temporarily (e.g. legs of jack-up barges) are likely to cause *Atrina* mortality. Therefore, a resistance of **'Low'** is suggested. Resilience is probably **'Low'** and sensitivity is assessed as **'High'.**

Penetration or disturbance of the substratum subsurface	None Q: High A: Medium C: Medium	Very Low Q: Medium A: Medium C: Medium	High Q: Medium A: Medium C: Medium

Sensitivity assessment. The evidence above (see 'abrasion') suggests that *Atrina fragilis* can survive low levels of abrasion. However, penetrative gear such as beam trawls, Rapido trawls and scallop dredges are likely to cause severe mortality in *Atrina fragilis* and possibly other characteristic species. Therefore, a resistance of **'None'** is suggested. Resilience is probably **'Very low'** and sensitivity is assessed as **'High'.**

Changes in suspended solids (water clarity)	Medium Q: High A: Medium C: Medium	Low Q: Medium A: Medium C: Medium	Medium Q: Medium A: Medium C: Medium

Pinnids are adapted to a sedimentary lifestyle and possess a unique ciliated waste canal for the removal of sediment from the mantle cavity (Yonge 1953). However, increased siltation will require increased metabolic demand on filtration and a likely decrease in growth and reproductive capacity. *Pinna bicolor* and *Pinna nobilis* occur in sheltered areas of low turbidity. However, juveniles settle in the boundary layer and grow rapidly to escape the high levels of sediment and it is likely that Pinnids are tolerant of suspended sediment. The absence of *Pinna* sp. from areas of severe sediment disturbance (Bulter *et al.* 1993) suggests that the populations in areas of high sediment availability will be adversely affected by increased siltation.

Thrush *et al.* (1999) demonstrated a decrease in the biochemical condition in *Atrina zealandica* with increasing sediment load in the Mahurangi Estuary, New Zealand. Ellis *et al.* (2002) examined the effects of the addition of sediment in laboratory experiments, at a range of turbidity treatments that represented the range of values (23-512 mg/l) experienced in the Mahurangi Estuary, where the normal background turbidity ranged from 12-90 mg/l but were much higher in storm associated resuspension of sediment or runoff from forestry. The initial addition of suspended sediment increased clearance rates, in the same way, that increased seston (phytoplankton) was found to increase filtration rates (Ellis *et al.*, 2002; Safi *et al.*, 2007). Clearance rates increased with increasing suspended sediment until a threshold of ca 120 FTU (Formazin Turbidity Unit) at which clearance rates declined (Ellis *et al.*, 2002). Clearance rates continued to decrease over the duration of the experiment (12 days) in all of the sediment addition treatments. Negative effects on the condition of *Atrina zelandica* became apparent after only three days of exposure to increased suspended sediment levels, compared to controls with 'no' sediment added (Ellis *et al.*, 2002). In transplantation experiments, Ellis *et al.* (2002) found that *Atrina* transplanted to the area closest to the mouth of the estuary (lower suspended sediment flux) improved in condition over the three months of the experiment. But *Atrina* transplanted to upper estuary sites (with high suspended sediment flux, equivalent to 108 g dry weight of sediment per month in sediment traps) lost condition. No *Atrina* occurred naturally at this upper estuary site, which may represent the upper limit of its tolerance of suspended sediment. *Atrina* also lost condition at intermediate sites (e.g. at 49 g dry weight of sediment per month) (Ellis *et al.*, 2002). It may be that *Atrina zelandica* found in areas with naturally high sediment loading are adapted to cope better with increases in suspended sediment than those from areas with lower background sediment concentrations. Nonetheless, very large increases in suspended sediment are still likely to be detrimental to *Atrina zelandica* (Hewitt & Pilditch, 2004). Hewitt & Pilditch (2004) examined the response of feeding in *Atrina zealandica* to 0-500 mg/l for ca one day. *Atrina* was able to reject filtered particles (75-100%) but maintain high organic absorption efficiencies. However, they identified site-specific differences in response that they suggested were due to prior history of exposure to suspended sediments at each site (Hewitt & Pilditch, 2004). Arrieche *et al.* (2010) noted that juvenile *Atrina maura* withstood high seston levels of ca 900 mg/ l for 10 consecutive days in their flume experiments.

The characteristic brittlestars are suspension-feeders but can switch to deposit-feeding or become detrivores in stagnant water or low flow so could avoid the 'clogging' effects of high suspended sediment loads.

Sensitivity assessment. *Atrina* spp. are probably well adapted to a sedimentary habitat and the occasional resuspension of sediment due to storms, as they are able to cleanse themselves quickly. Short-term (i.e. 3 day) increases in suspended sediment, similar to that created by storms and storm runoff, are likely to result in a loss of condition but not mortality. However, an increase in turbidity from, for example, 'clear' to 'intermediate' (100-300 mg/l) or turbid (>300 mg/l) for a period of a year (see benchmark) may be detrimental. Therefore, a resistance of **'Medium'** is recorded. Resilience is probably **'Low'** and sensitivity is assessed as **'Medium'.**

Smothering and siltation rate changes (light)	Medium Q: Low A: NR C: NR	Low Q: Medium A: Medium C:	Medium Q: Low A: Low C: Low

Last *et al.* (2011) buried *Ophiura ophiura* individuals under three different depths of sediment; shallow (2 cm), medium (5 cm) and deep (7 cm). The results indicated that *Ophiura ophiura* is highly tolerant of short-term (32 days) burial events, with less than 10% mortality of all buried specimens. This was largely a reflection of the ability of the species to re-emerge from all depths across all sediment fractions tested. Survival of specimens that remained buried was low, with 100% mortality of individuals that remained buried after 32 days. The experiments utilized three different fractions of kiln dried, commercially obtained marine sediment: coarse (1.2-2.0mm diameter), medium fine (0.25-0.95mm diameter) and fine (0.1-0.25mm diameter). Trannum *et al.* (2010) investigated how sedimentation from water-based drill cuttings could affect benthic communities, in comparison with natural sediment deposition. The authors concluded there was no effect of adding natural test sediment up to 2.4 cm but a significant reduction in the number of taxa, abundance, biomass and diversity of fauna with an increasing layer of thickness of drill cuttings (3-24 mm), suggesting other mechanisms affecting the fauna other than sedimentation, possibly lower contents of nutrients, toxicity and oxygen depletion. *Amphiura filiformis* was amongst the species to be absent from treatments under 6, 12 and 24 mm of artificial sediment, possibly due to its surface deposit feeding habits.

Atrina fragilis cannot burrow upwards through sediment (Yonge, 1953). However, one-third to one-half of the animal can protrude above the surface which, in adults, can be up to 10-15 cm above the sediment surface. Therefore, adult specimens may not be affected by smothering by 5 cm of fine sediment (see benchmark). Pinnids are adapted to a sedimentary lifestyle and exhibit a powerful exhalent current and a unique ciliated waste canal to remove sediment from the mantle cavity, as would be expected from occasional smothering due to storms (Yonge, 1953). Clearance of sediment from the mantle constitutes a metabolic cost that may reduce their reproductive ability (Butler *et al.*, 1993). Individuals are likely to cleanse themselves relatively quickly.

Sensitivity assessment. *Atrina fragilis* is known from weak to moderately strong currents. For example in Knightstown, Valentia Island, the population is exposed to >2 knots (ca >1 m/s) on spring tides (Dan Minchin pers. comm.). Stirling *et al.* (2016) reported that Scottish records, including this biotope) occurred in areas with peak currents, during a mean spring tide, of 0.65 m/s on average and a maximum of 1.24 m/s. Therefore, the deposition of 5 cm of fine sediment might be removed or redistributed in a relatively short period. Therefore, the characteristic brittlestars may survive and reposition themselves at the top of the sediment, while adult *Atrina fragilis* would not be smothered, depending on size. However, small juveniles may be smothered and resistance is assessed as **'Medium'**. Resilience is probably **'Low'** and sensitivity is assessed as **'Medium'**.

Smothering and siltation rate changes (heavy)	None Q: Low A: NR C: NR	Very Low Q: Medium A: Medium C: Medium	High Q: Low A: Low C: Low

Last *et al.* (2011) buried *Ophiura ophiura* individuals under three different depths of sediment; shallow (2 cm), medium (5 cm) and deep (7 cm). The results indicated that *Ophiura ophiura* is highly tolerant of short-term (32 days) burial events, with less than 10% mortality of all buried specimens. This was largely a reflection of the ability of the species to re-emerge from all depths across all sediment fractions tested. Survival of specimens that remained buried was low, with 100% mortality of individuals that remained buried after 32 days. The experiments utilized three different fractions of kiln dried, commercially obtained marine sediment: coarse (1.2-2.0mm diameter), medium fine (0.25-0.95mm diameter) and fine (0.1-0.25mm diameter). Trannum *et al.* (2010) investigated how sedimentation from water-based drill cuttings could affect benthic communities, in comparison with natural sediment deposition. The authors concluded there was no effect of adding natural test sediment up to 2.4 cm but a significant reduction in the number of taxa, abundance, biomass and diversity of fauna with an increasing layer of thickness of drill cuttings (3-24 mm), suggesting other mechanisms affecting the fauna other than sedimentation, possibly lower contents of nutrients, toxicity and oxygen depletion. *Amphiura filiformis* was amongst the species to be absent from treatments under 6, 12 and 24 mm of artificial sediment, possibly due to its surface deposit feeding habits.

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Litter	Not Assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR	Not assessed (NA) Q: NR A: NR C: NR

Not assessed

Electromagnetic changes	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

No evidence was found.

Underwater noise changes	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant. *Atrina fragilis* probably reacts to localised vibration but is unlikely to react to the noise from passing vessels etc.

Introduction of light or shading	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant. The characteristic species *Atrina fragilis* or *Amphiura* spp. are suspension feeders, feeding on phytoplankton. Artificial light or localised shading is unlikely to alter phytoplankton productivity to any significant level, especially in deep waters and/or where currents supply food to the fan mussel.

Barrier to species movement	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant. This pressure is considered applicable to mobile species, e.g. fish and marine mammals rather than seabed habitats. Physical and hydrographic barriers may limit the dispersal of larval stages or propagules. However, the dispersal of larval stages or propagules is not considered under the pressure definition and benchmark.

Death or injury by collision	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant to seabed habitats. NB. Collision by interaction with bottom towed fishing gears and moorings are addressed under 'surface abrasion' above.

Visual disturbance	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

Not relevant. The characteristic species probably react to localised shading but are unlikely to react to the visual disturbance from passing vessels etc.

Biological Pressures

	Resistance	Resilience	Sensitivity
Genetic modification & translocation of indigenous species	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Liu *et al.* (2011) identified five lineages of mtDNA in *Atrina pectinata* along the coast of China and one location in Japan. The linages corresponded to six morphotypes although intraspecies hybridization between lineages obscured the differences. However, no evidence of translocation, breeding or hybridization with other species was found.

Introduction or spread of invasive non-indigenous species	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

Crepidula sp. may have had some impact on near-shore populations of *Atrina fragilis* on the south coast of England (Dan Minchin pers comm.). But no further evidence was found.

Introduction of microbial pathogens	No evidence (NEv) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	No evidence (NEv) Q: NR A: NR C: NR

The Pinnids are parasitized by the pea crab (Pinnotheridae) (Yonge 1953). Butler *et al.* (1993) state that *Pinna bicolor* and *Pinna nobilis* harbour macroscopic commensals or parasites of unknown effect, although an unidentified parasitic microbe has been recorded as causing castration of *Pinna nobilis*. Any parasite is likely to reduce the condition of the host but no information on mortality rates (if any) was found.

Maeno *et al.* (2006) examined specimens of *Atrina pectinata* after the mass mortality (60-90%) of the pen shell in the fishing grounds of Ariake Bay, Japan between 2003 and 2004. The authors concluded that novel virus-like particles found in the kidney and gill tissues of moribund specimens were the probable cause of mortality. Subsequent, experimental infection of *Atrina lischkeana* in the laboratory and the field (Ariake Bay, Japan) resulted in necrosis of kidney, gill and mantle tissue and 20% mortality in the laboratory but 80-100% mortality of infected specimens in the field Maeno *et al.* (2012; abstract only). Therefore, the authors concluded that the virus-like particles were the causal agent.

Echinoderm populations have been massively reduced by sudden outbreaks of epidemic disease. Cases include the mass mortality of the sea urchin *Diadema antillarum* throughout the Caribbean as a result of infection by a water-borne pathogen (Lessios, 1988), and the decimation of urchin populations in the North Atlantic by parasitic amoebae and nematodes (Hagen, 1997). Brittlestars have symbiotic sub-cuticular bacteria. The host-bacteria association can be perturbed by acute stress and changes in bacterial loading may be used as an indicator of sub-lethal stress (Newton & McKenzie, 1995).

Sensitivity assessment. The evidence from Japan suggests that *Atrina* is sensitive to the disease documented. However, no evidence of the disease outside Japan was found. Similarly, no evdicne of the effects of diseases in UK brittlestar populations was found. Hence, an assessment of **'No evidence'** is given for UK populations until further evidence is found.

Removal of target species	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR	Not relevant (NR) Q: NR A: NR C: NR

In Spain, pinnids may be collected for consumption, used as bait, or for use as souvenirs. In the Bay of Naples, the byssus threads were historically used for making glues. In the Pacific, declines in production have occurred as a result of the exploitation of other species of pen shell (Cardoza-Velasco & Maeda-Martinez, 1997). However, *Atrina fragilis* is not targeted by any commercial fishery in the UK.

Removal of non-target species	None Q: High A: Medium C: Medium	Very Low Q: Medium A: Medium C: Medium	High Q: Medium A: Medium C: Medium

Scallop dredging and demersal trawling have been implicated in the decline in populations of *Atrina fragilis* and other pinnids (Anon, 1999c; Hall-spencer et al., 1999; Solandt, 2003; Šimunović et al., 2001; Fryganiotis *et al.*, 2013; Chavaz-Villabla *et al.*, 2022). In the UK, *Atrina fragilis* was more common in scallop beds in the early 1900s than at present. Presumably trawling and dredging of these formerly populated regions is the reason for the decline of this species (Minchin pers. comm.). Dredging of a *Pecten maximus* bed off Glengad Head, Ireland, after 1975, removed many live specimens of *Atrina fragilis* in scallop dredges and the population of fan mussels is thought to have been destroyed by subsequent dredging (Anon 1999c). Solandt (2003) noted anecdotal records where 'considerable fragments of *Atrina* shells were collected by scallop trawlers', and large individuals caught in the Celtic Sea in the 1970s with 'decks covered with the broken fragments of this species'. Solandt (2003) also reported anecdotal records from the diving community of considerable numbers of *Atrina fragilis* found in areas where scallop trawlers and dredgers cannot set gear.

In the Adriatic queen scallop (*Aequipecten opercularis*) trawl fishery, *Atrina fragilis* incurred more damage as a result of the fishing and sorting process than any other species of bycatch (Pranovi *et al.*, 2001). In the Adriatic, Fryganiotis *et al.* (2013; Fig 2) reported that the density of fan mussels in trawled areas (ca 0.03 individuals/km) was sparse compared to the areas in which bottom trawling was prohibited for 25 years (ca 5.5 individuals/km). Rapido trawling (a form of beam trawl) for scallops in the Gulf of Venice resulted in the removal of organisms from the top 2 cm of sediment and an 87% reduction in *Atrina fragilis* abundance in the trawl tracks. Some specimens were speared on the trawl teeth and pulled from the sediment (Hall-Spencer *et al.*, 1999). Pinnids in the Mediterranean are associated with seagrass beds, the removal of which has been linked to the decline in pinnid populations (Richardson *et al.*, 1999).

Sensitivity assessment. The above evidence suggests that populations of *Atrina fragilis* are vulnerable to demersal fisheries. Therefore, a resistance of **'None'** is suggested. Resilience is probably **'Very low'** and sensitivity is assessed as **'High'**.

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