Evaluation of metal levels in biota and sediments in the vicinity of Aberthaw

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Aberthaw FGD monitoring
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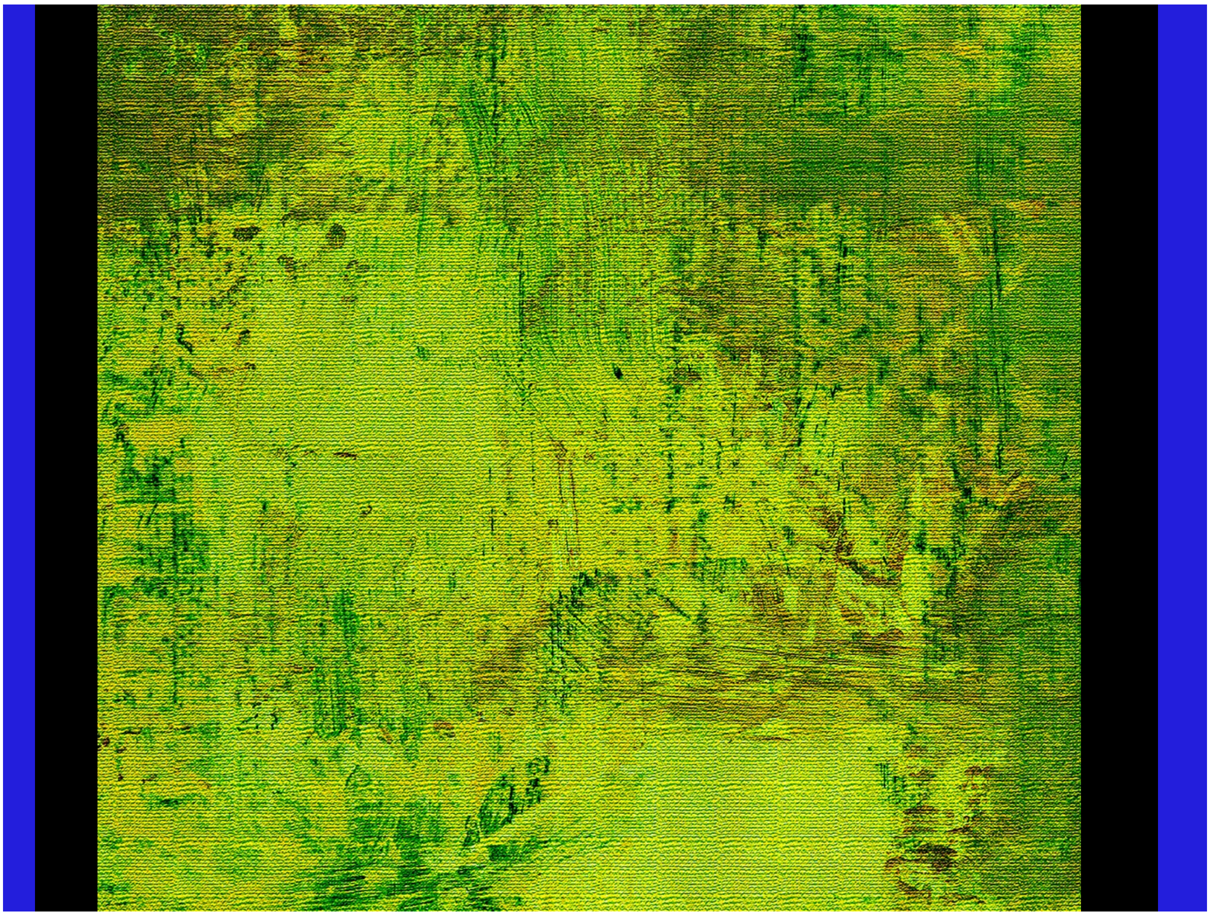

Executive summary

RWE Generation UK plc has installed a flue gas desulphurisation (FGD) plant at Aberthaw Power Station which was initially commissioned in spring 2008. The plant employed a seawater stripping process and following commissioning of the plant, the levels of all metals discharged in the effluent increased. A three-year (2007 to 2009) monitoring programme was undertaken to investigate metal levels in intertidal biota and sediments in the vicinity of Aberthaw Power Station, to determine both baseline (pre-commissioning) conditions and any changes following plant commissioning. Data from 2007 and 2008 described the baseline, pre-commissioning conditions, while the 2009 data described conditions after the first year's operation of the FGD plant. Subsequent post-commissioning monitoring was undertaken annually between 2010 and 2017. Initial monitoring included the sites EO, WO and CON, with the addition of the sites LW, LE and HO in 2010. The sites WO and CON were discontinued in 2013 and although mentioned, they are not the focus of the analysis. Following the end of operation of the plant in 2020 an additional survey was undertaken in 2022.

Serrated wrack (*Fucus serratus*), limpet (*Patella vulgata*)and dogwhelk (*Nucella lapillus*) were identified as target monitoring species on the basis of their documented use as bio-indicators and their relative abundance on the shores at Aberthaw. Between 2007 and 2009 biota and sediments were analysed for a suite of metals as per List I and List II of the Dangerous Substances Directive (76/464/EEC) (now repealed), most of which are now listed under the revised Priority Substances Directive (2013/39/EU) as priority substances, priority hazardous substances or under the Water Framework Directive (2000/60/EC) as specific pollutants. From 2010 onwards methylmercury (MeHg) was added to the suite of determinands for biological and sediment components.

It was deemed that any differences evident in the data from 2007 to 2008 was attributable to natural variability in background metal concentrations and is considered to provide an adequate baseline against which subsequent changes could be compared. Pre-commissioning metal levels in the tissue samples of the target species were consistent with results reported from elsewhere in the UK and it is considered that concentrations at Aberthaw, although elevated above the natural background levels, were not significantly high.

Post-commissioning, some spatial variability in metal concentrations was evident in the biota samples, although, with the exception of mercury, this was comparable to that reported in 2007 and 2008. In 2009, levels of mercury (dry weight) in *N. lapillus* were appreciably higher than pre-commissioning levels at the site immediately to the east of the outfall (EO). Despite mercury levels in 2010 being similar to those recorded in 2007/2008, 2011 concentrations at the site EO were again elevated and were higher than at other sites. This trend of appreciably higher concentrations of mercury at EO compared to pre-commissioning and to other sites, remained until a peak in the reported levels in 2013. From 2013, a general decline in mercury levels was observed at EO up to and including 2017 (no data were available for 2016 owing to there being insufficient material for analysis). In 2022 mercury was found to be below detectable limits for all species and all locations, however did show a reduction at site EO compared with 2017, showing a reversal in effect.

For all species, the concentrations of mercury recorded in the vicinity of the discharge (site EO) between 2010 and 2017 remained appreciably higher than those recorded at the more remote sites (LW, LE, HO). Levels of mercury in the tissue of the biota showed a general decline with distance from the outfall; the sites to the east of outfall were overall higher than those to the west. No such patterns were evident for other metals, at least not to the same notable extent as that of mercury.

This post-commissioning (2009 and 2017) distribution of mercury in biota suggested that the influence of the outfall orientation (directed south east) and the local hydrodynamics resulted in greater exposure to the FGD discharge water of habitats to the east of the discharge. However, as the target biota along this stretch of shore remain in sufficient numbers to permit sampling, with no obvious changes in community structure observed by the monitoring teams, there appears to be no evidence to indicate any toxic effects on the target species associated with the changes to the metals levels in the discharge. Post closure of the plant in 2022 mercury was found to be below detectable limits for all species and all locations though a reduction was apparent at EO.

The MeHg concentrations reported in both *N. lapillus* and *P. vulgata* indicate relatively consistent levels across the survey area. The proportion of total mercury represented by organic mercury (i.e. in MeHg) showed a clear spatial pattern in both species, with the lowest proportions (up to approximately 10%) occurring in the

vicinity of the outfalls to the east in all years from 2010, since MeHg was added to the monitoring suite. Post closure of the plant in 2022 MeHg was found to be below detectable limits for all species and all locations, however represented a decrease across all sites when compared with 2017.

Post-commissioning increases in the organic mercury concentrations recorded in biota samples may be indicative of bio-magnification. This increased tissue burden could result in elevated metal levels in commercial fish species feeding on the target species tested. However, a study by Jacobs (2013), on the levels of mercury and MeHg in muscle tissue taken from bass (*Dicentrarchus labrax*) caught in the vicinity of the Aberthaw outfalls, considered that any risks to potential human food sources were negligible.

In 2007 and 2008 mean sediment-bound concentrations of all metals (excluding cadmium and nickel) were between the Interim Sediment Quality Guideline and Probable Effect Level (ISQG and PEL). Throughout the study period, sediment bound concentrations of the majority of metals exceeded the relevant ISQG but were below the PEL. However, 2008 and 2013 mean levels of nickel were greater than the PEL, but these declined to close to the ISQG level in 2017 having been below this level only in 2009. In 2022 nickel levels increased compared to 2017 but were still below the PEL and the mean value for the whole study.

All cadmium concentrations were lower than the ISQG. When considered over the study period as a whole, mean concentrations of cadmium and mercury were less than the relevant ISQG while those for all other metals exceeded this lower limit. It is likely that any biological effects as a result of these exceedances would be chronic rather than acute.

Whilst there are limitations with the data collected during the 2022 study (higher limits of detection), the data does show that there is a reversal in levels of metals and methylmercury at the outfall site (EO). This therefore demonstrates a reversal in the effects of the FGD and supports cessation of the monitoring at this site.

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1. Introduction

1.1 Background

Aberthaw Power Station was a 1605 MW coal-fired plant located on the South Wales coast, 9 km west of Barry which ceased operation in March 2020. It abstracted cooling water (CW) from, and returned it to, the Bristol Channel via two outfalls at mean low water near Breaksea Point. In order to meet new air quality emissions limits for sulphur dioxide, RWE Generation UK plc (previously RWE npower plc) (RWE) commissioned a flue gas desulphurisation (FGD) plant using a seawater stripping process. After passing through the condensers a proportion of the CW flow was diverted to the FGD absorber tower where it stripped out most of the acidic gases from the flue gas, together with some dust and trace quantities of metals. This FGD stream was then mixed with the remaining CW flow and aerated before being discharged through the existing CW outfalls.

During the Environmental Permitting for Aberthaw an improvement condition (IC21) was set, under which a monitoring programme was required to investigate the potential impacts of FGD operation at the power station.

'*IC21. A written report shall be submitted to the Environment Agency for approval. The report shall describe the post-FGD operation local environmental quality with respect to List I and List II substance and selenium burdens in sediment and marine species as determined according to the plan submitted as further information to the application. Further reports shall be submitted annually, subject to review following completion of improvement condition reference IC22, and until notified in writing by the Environment Agency*'.

In accordance with this, monitoring has been undertaken at the site since 2006. Following commissioning of the plant, the levels of all metals in the CW discharge increased, although the concentrations of measured parameters complied with the discharge permit limits set by Natural Resources Wales (NRW) (previously Environment Agency Wales (EAW)).

FGD abatement of the first two power station units took place on the 8th March and 16th April 2008 respectively, while the third unit had initial abatement applied on the 20th January 2009. Following FGD commissioning and up to the closure of the plant in March 2020, the annual load factor of the power station is reported below as is the estimated amount of mercury discharged (Table 1.1).

Following closure of the plant and cessation of the FGD, an additional monitoring campaign was completed with the aim of showing a reversal in metals levels such that condition IC21 can be closed out.

Table 1-1. Annual load factor and mercury discharged by Aberthaw power station following FGD commissioning and subsequent plant closure (2009-2022).

Year	Annual load factor	Mercury discharge
2009-2012	40 – 70%	30 – 51kg
2013	73%	41.0kg
2014	59%	35.8kg
2015	55%	31.6kg
2016	51%	22.9kg
2017	19.9%	4.31kg
2018	4%	1.4kg
2019	3.5%	2.8kg
2020	0%	0kg

Year	Annual load factor	Mercury discharge
2021	0%	0kg
2022	0%	0kg

1.2 Monitoring aims

The principal aim of this report is to present the findings of the 2022 monitoring campaign with the view to cease monitoring entirely and close out condition IC21. Data are discussed in relation to results from previous surveys and to identify any reversal of effects from the FGD discharge.

1.2.1 Agreed monitoring

Following discussions between NRW and RWE a monitoring programme was agreed to investigate metal levels in intertidal biota and sediments in line with an improvement condition (IC21) on the Environmental Permit for the station. The metals to be monitored were originally included in the (now repealed) European Dangerous Substances Directive (76/464/EEC) as List I and II substances (Table 1-2). List I substances were those most toxic to aquatic life and are selected on the basis of their persistence, toxicity and bioaccumulation potential. List II substances were other materials which had a deleterious effect on aquatic life.

Table 1-2. Revised Priority Substances Directive and WFD designated pollutants studied at Aberthaw 2007-2022.

Revised Priority Substances Directive: Priority hazardous substances (PHS)	Revised Priority Substances Directive: Priority substances (PS)	WFD: Specific Pollutants (SP)
Cadmium	Lead	Arsenic
Mercury	Nickel	Chromium
		Copper
		Zinc

Of the original List I and II substances investigated in this study, cadmium, lead, mercury and nickel are designated as priority substances (PS) under the revised Priority Substances Directive, with cadmium and mercury also designated as priority hazardous substances (PHS). The Water Framework Directive (2000/60/EC) (WFD) aims to reduce the discharge of PS and to achieve the cessation of PHS discharges. In addition, arsenic, chromium, copper and zinc are also designated as specific pollutants (SP) under the WFD; these are considered toxic substances when discharged to water in significant quantities for which WFD requires environmental standards to be set. Selenium was originally included in the monitoring programme but does not fall under the above designations and the analysis has not been continued.

In addition to the suite of metals examined, levels of methylmercury (MeHG) were also examined within biota and sediments. MeHg is a highly toxic form of mercury which is readily taken up by organisms. It is formed primarily in sediments by sulphate reducing bacteria which take up mercury in its inorganic form and convert it to MeHg through metabolic processes. The bacteria may then be consumed by primary consumers or the MeHg may be excreted where it becomes bio-available to other sediment dwelling organisms.

The monitoring has been undertaken during 2007 and 2008 as pre-commissioning years and then in 2009 following commissioning. The 2009 surveys identified significant increases in mercury levels in the monitored biota in the vicinity of the discharge, consequently, further monitoring was requested and surveys have been conducted annually between 2010 and 2017 with this additional survey in 2022. The focus of the monitoring has been modified over the years in agreement with NRW. The 2014 and the 2022 surveys were a repetition of those completed in 2017 with assessments of biota and sediments only.

1.3 Description of the Aberthaw shore

The power station outfall structures are located at the seaward edge of an intertidal area of wave-swept limestone ledges (Figure 2.1). To the landward side there are sandflats, mainly of firm clean sand with one area of softer, sandy mud directly in front of the Limpert Bay car park. The rock platforms to the west of the outfalls are dominated by the brown alga *Fucus serratus*, with *Sabellaria alveolata* reefs and a variety of other algal and faunal species. The faunal communities here are generally sparse in nature with pockets of limpets, *Patella vulgata*, dogwhelks, *Nucella lapillus* and the periwinkle, *Littorina littorea* which were described by Bamber (1997) as “relatively normal for this kind of rocky intertidal habitat”. To the east, beyond Breaksea Point, the shore is more exposed and is characterised by boulders and cobbles which support a sparse flora and fauna. To the west of Limpert Bay towards Penry Bay the limestone ledges are also subject to greater levels of exposure than those in the immediate vicinity of the outfalls and support a correspondingly sparser community than that found closer to the outfalls.

2. Methods

An initial potential effects report (Jacobs, 2006) considered mechanisms by which metals could become concentrated in the sediments and biota at Aberthaw and proposed a monitoring programme. Subsequently, three areas of the shore were selected as suitable for sampling which would reflect the influence of the discharge on intertidal biota. Site EO (East Outfall) was located approximately 130 m to the east of the eastern outfall (NGR 301783 365674). Similarly, site WO (West Outfall) was located approximately 100 m to the west of the western outfall (NGR 301582 365856). EO and WO were selected as sites likely to show some influence of changes to the CW discharge following commissioning of the FGD plant. The control site (CON) was located approximately 400 m to the west of the outfalls towards the western side of Limpert Bay (NGR 301351 666005); this site was chosen as it was considered outside of the immediate area of influence of the discharge. Although a more remote location would have been preferable for a control site this was not initially considered as the habitat changed to cobble and boulder dominated shores to the east within a relatively short distance of the outfall; similarly, the shore became considerably more exposed to the west resulting in a sparser flora and fauna.

Owing to their high surface area to volume ratio, fine sediments have a natural propensity to adsorb metals and represent the ultimate sink for materials entering the marine system. Consequently, sediment-bound metal concentrations can provide a good indication of environmental contamination by heavy metals. As appreciable amounts of fine sediments (i.e. sandy muds) occur towards the top of the sedimentary shore in Limpert Bay, a single sampling site was located in this area (site SED, NGR 301867 166124).

Following the 2009 survey which indicated increased levels of mercury in biota at all three sites (EO, WO and SED), additional sample points were selected for subsequent surveys to increase the spatial extent of the study with the aim of better identifying the extent of the influence of the CW discharge. Reference was made to water temperature data collected by RWE which indicated that observable increases in water temperature above ambient dissipated within 1 km to the east and west of the outfalls. This was used as a marker for the short term thermal plume in which it was assumed the highest levels of discharged metals would occur. Consequently, additional sample sites were selected from within this area. As the 2009 data indicated that mercury levels in biota were greater to the east of the outfalls at EO, two further sites were selected to the east; LE (Limpert East, NGR 301900 165512), approximately 300 m from the outfall and HO (Historical Outfall, NGR 302386 165377), located immediately to the east of the old outfalls and approximately 800 m from the current discharge. One further site (LW, Limpert West, NGR 300765 166157) was selected approximately 900 m to the west of the current discharge. Generally, all six sites supported similar ecological communities, although those at the eastern (HO) and western (LW) extremes of the survey area had lower abundances than elsewhere.

Following the 2013 report and subsequent discussions between RWE and NRW, the scope of the 2014 monitoring programme was reviewed. It was considered that given the consistency of the data collected up to 2013 the continued inclusion of sites CON and WO would add little to the future findings and would not represent a cost effective use of resources. Consequently, it was decided that the 2014 survey would encompass biota sites HO, LE, EO and LW only while the sediment site SED site would also be retained; this amended approach was continued in 2015, 2016, 2017 and 2022. This report focusses on the sites currently sampled. Therefore, the full suite of data collected from sites CON and WO have not been presented graphically in the main body of this report; instead see the appendices and previous reports for CON and WO graphical data and more detailed discussion concerning these sites (e.g. Jacobs 2016).

All sites were located at mid to low shore height to ensure a consistent period of immersion of between eight to ten hours per tidal cycle.

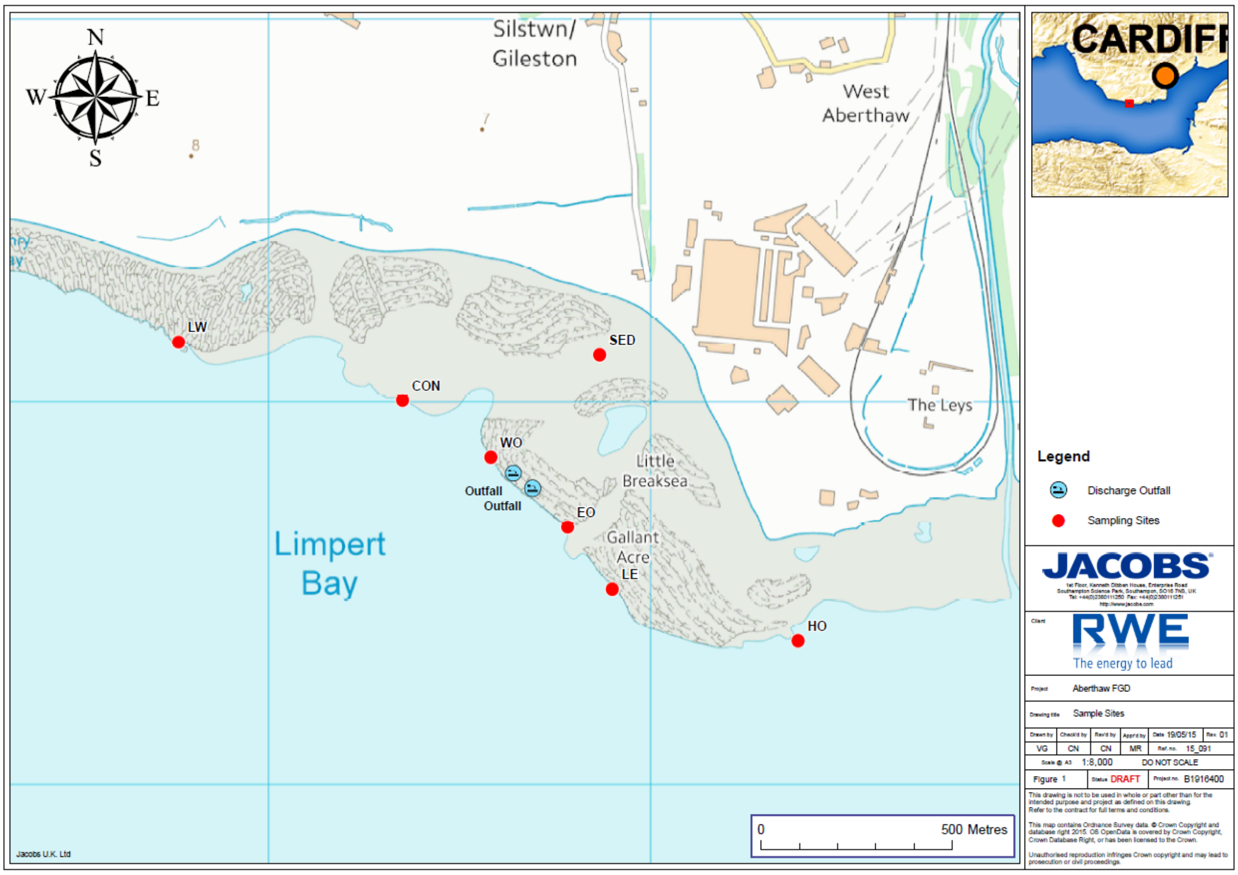


Figure 2-1. Sampling sites in Limpert Bay, including both current and discounted sites.

2.1 Biota

Fucus serratus, *Patella vulgata* and *Nucella lapillus* were identified as suitable target species on the basis of their documented use as bio-indicators in a wide range of metal bioaccumulation studies (Bryan *et al.*, 1985). In addition, their relative abundance on the shores of Limpert Bay (as indicated by the findings of the walkover survey conducted in September 2006) indicated that the populations would support an annual sampling programme. The range of selected target species will facilitate the assessment of metal contamination in the different chemical phases (e.g. dissolved, particulate etc.) and, as they encompass primary, secondary and tertiary trophic levels, any bio-magnification may also be indicated.

Annual sampling was undertaken in late March/early April with the sampling date standardised around the first spring tide in the month to facilitate safe access to the low shore. In 2022 the surveys were conducted on the 4th and 5th April and used the same sampling locations and methodology as in previous surveys. Sampling times and methodologies have been standardised to ensure consistency between each years' samples.

Samples were collected from mid to low shore within 50 m of the nominal sample sites, with three replicate samples of each target species being collected at each site. Each replicate of *N. lapillus* and *P. vulgata* comprised a minimum of ten individual animals within a standard size range (Figure 2.2). *N. lapillus* was picked by gloved hand and transferred to a labelled polythene jar. *P. vulgata* was dislodged by a swift tap with a stainless steel knife under the edge of their shell and then placed in labelled polythene jar. *F. serratus* was sampled by cutting off approximately 200 mm of the previous year's growth using stainless steel scissors. Each replicate comprised a minimum of 30 plants. This material was handled with nitrile gloves and transferred to sealable polythene bags.

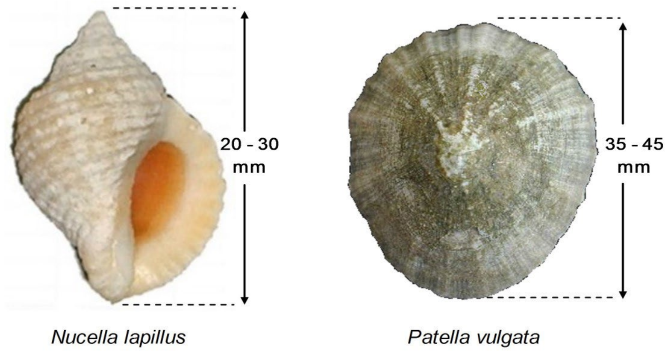


Figure 2-2. Standard size range for dogwhelk (*Nucella lapillus*) and limpet (*Patella vulgata*)

Analysis of biota for all metals was undertaken by Marchwood Scientific Services (UKAS accredited).

Biota tissue samples were analysed for the range of revised Priority Substances Directive (PS and PHS) metals, the WFD (SP) metals outlined in Table 1.2. Tissue samples were digested with nitric acid to extract metals and the digest analysed by inductively cooled plasma optical emission spectroscopy (ICP-OES). MeHg samples were digested by liquid-liquid micro-extraction (LLME) with analysis undertaken by atomic fluorescence spectroscopy.

It should be noted that in 2016, although normal *Nucella lapillus* samples were submitted as in previous years, the laboratory was unable to extract sufficient material to conduct analysis for PS, PHS, SP or selenium. The timing of the notification of this was such that the season had moved away from the normal biota sampling period; water temperature and animal sizes would have changed along with the associated body burdens making any comparison with the completed 2016 analysis unviable. In addition, re-sampling of all biota was not viable either, as the data would not be comparable to those collected in previous years due to the seasonal difference. As a result the only tissue analysis data obtained for *N. lapillus* in 2016 were for MeHg.

In 2022, no issues occurred with the collection of samples and subsequent laboratory analysis; the results of this have been subsequently presented in this report. Following the COVID-19 pandemic (during which many commercial laboratories changed focus to deal with medical samples) the availability of laboratories able to conduct the specialist analysis required for the Aberthaw samples was greatly limited. The available laboratories were not able to produce the same levels of detection for the required analytes as had previously been achieved. As a result of this the limit of detection for all analytes was higher than in other sampling years because of the change in laboratories used for the analysis. The limits of detection were <1mg/kg except for methyl-mercury which had a limit of detection of <100 µg/kg. Where it has only been possible to record a below limit of detection value for an analyte the actual value would fall between zero and the limit of detection and so the limit of detection value has been taken as a maximum for subsequent discussion of the results.

2.1.1 Biometric analysis

Of the specimens collected from each site, 30 randomly selected individuals were processed for biometric analysis prior to analytical analysis at Marchwood Scientific Services. On return to the laboratory, the live gastropods were processed for biometric data. This analysis was undertaken in order to give a numeric indication of the health of the gastropods and to align the FGD post-commissioning work with that of the pH variation report (Jacobs, 2016a). The maximum shell height, shell width and blotted tissue wet weight of each individual dogwhelk (*N. lapillus*) was measured. The same was completed for the limpets (*P. vulgata*), although shell weight was also recorded. All shell size measurements and wet weights were recorded to the nearest mm and 0.001g, respectively.

2.1.1.1 Data analysis – Biota health assessment

Condition Indices (CI) were calculated for all samples using a relationship between tissue weight and shell size, status. High values indicate a better condition or health. The calculation is given as (Lundebye *et al.*, 1997):

$$CI = \text{tissue dry weight (g)} \times 1000(\text{shell height(cm)})^3$$

As wet weight has been recorded for each specimen, the values were converted to dry weight. For *N. lapillus*, wet weight:dry weight ratios were derived from data for the related species *Buccinum undatum* given by Rumohr *et al.*, 1987, and for *P. vulgata*, values were converted using the average ratios for all gastropods detailed by Rumohr *et al.*, 1987.

2.2 Sediment

Between 2007 and 2022 sediment samples were collected at the sediment site (SED) (Figure 2.1). Five replicate samples (replicates A to E) were taken from within a ten-metre radius of the nominal site position with material taken from the surface to a depth of 1 to 2 cm. From 2010 to 2013 sediment samples were also taken at each of the six biota sites. However, as only limited amounts of suitable material were available at these sites, single replicate samples were collected.

Sediments were analysed for the range of metals detailed above and MeHg as well as particle size analyses (PSA). Analysis was undertaken by Socotec (UKAS accredited). Sediments were digested with nitric acid to extract metals and the digest analysed by inductively cooled plasma mass spectroscopy (ICP-MS) for all metals. MeHg samples were digested by liquid-liquid microextraction (LLME) with the digest analysed by liquid chromatography and mass spectrometry – mass spectrometry (LCMS/MS). Sediment PSA was also undertaken.

The level of enrichment of a metal within a sediment can be assessed by comparing the value of the metal:aluminium ratio to the Background Reference Concentration (BRC) for that metal. The BRC gives a range of values for the metal:aluminium ratio which would be expected in uncontaminated sediments as determined by OSPAR (OSPAR, 2000). Concentrations are considered to be close to background if the metal:aluminium ratio is less than twice the upper limit of the BRC.

In order to determine likely biological effects, sediment-bound metal concentrations were examined by the ISQG/PEL approach developed by Environment Canada (Canadian Council of Ministers of the Environment, 1995). The Interim Sediment Quality Guideline (ISQG) (referred to in previous reports as the Threshold Effects Level (TEL)) of a substance is the concentration below which sediment-associated chemicals are not considered to represent significant hazards to aquatic organisms. The Probable Effects Limit (PEL) represents the lowest concentration of a substance that is known to have had an adverse impact on aquatic organisms.

3. Results

3.1 Biota analysis

Metal concentrations recorded in biota are given in Appendix A with all data expressed on a dry weight basis. Mean and individual replicate concentrations for each metal are given for each of the three target biota. Standardised metal concentrations are also presented, i.e. the mean value for each site divided by the annual average.

3.1.1 Mercury in biota

3.1.1.1 Dogwhelk (*Nucella lapillus*)

Following little variation from 2007 to 2008 at site EO (Figure 3.1 (i)), mean mercury concentrations in *N. lapillus* showed a sharp increase in 2009. This rise in mean mercury levels was three times that recorded in the previous year (EO 2008 = 1.22 mg/kg, EO 2009 = 3.98 mg/kg), but was followed by a decrease in concentration from 2010 to 2012, similar to those prior to 2009. In 2013, the mean concentration at EO exceeded the previous highest mean concentration recorded in 2009, and using LW as a reference site, was almost seven times greater than that recorded at LW in the same year. Mercury levels continued to remain high in 2014 and 2015 but showed a general decrease in concentration, a trend which continued into 2017 where the mean concentration is closer to that of levels in 2007/2008 with 1.98 mg/kg (no data were available for 2016 owing to there being insufficient material for analysis). This decrease in mercury levels is apparent within the 2022 data at site EO with values reported as <1 mg/kg. Whilst the exact level of mercury is not available, Figure 3.1 clearly shows the decrease in mercury at site EO.

Patterns of mercury levels in 2022 are not clear at the other sites owing to the LoD within the labs, although values at LE were within ranges that have been observed within the monitoring programme, specifically 2012 to 2015. No dogwhelk were present at site HO therefore no data are available for comparison. There has been variability in the values of mercury in *N. lapillus* at all sites across the monitoring years, however this change is less well pronounced with increasing distance from the outfall with comparably almost no variance at site LW.

In comparison to the other sites, the highest concentrations of mercury were consistently recorded at EO, and a trend of decreasing concentrations with distance from the outfall can be observed. This trend is emphasised when comparing the mean mercury concentration values averaged between 2010 and 2017 at LW, EO, LE, HO (Appendix C; Figure C.1), where the highest concentration was at EO with 2.52 mg/kg, with declining values at LE, then HO and the lowest values occurring at the reference site LW with a concentration of 0.48 mg/kg. In 2017, the lowest individual replicate and mean concentration of mercury across all years and sites was recorded at LW (see Table 3.1) with values of 0.198 mg/kg and 0.263 mg/kg, respectively. For further evidence of the increases in mercury concentrations in *N. lapillus* at EO compared to other sites (discontinued sites CON and WO) see the standardised metals concentrations (against the annual mean) figures (Appendix B; Figure B.1).

3.1.1.2 Limpet (*Patella vulgata*)

In 2009 mercury levels in *P. vulgata* at EO showed an increase of over six times the levels of previous years and were significantly higher than at any other site (Figure 3.1 (ii)). The elevation of mercury at this site is further demonstrated by the standardised mercury concentration figure (Figure B.2) for *P. vulgata* within Appendix B. Although, mercury levels fell at EO in 2010 and 2011 the concentrations remained above pre-commissioning levels. In 2012 the concentration of mercury at EO showed a rise to a level intermediate between those recorded in 2009 and 2010 and remained higher than elsewhere. In 2013 the level of mercury in *P. vulgata* at EO was similar to that recorded the previous year and was again higher than at other sites. In 2014 mercury levels were lower at all sites sampled compared to the previous year, although the highest concentration was again recorded at EO. The highest mercury level continued to be recorded at EO in 2015 and 2016 at a concentration of over fifteen times greater than that recorded at site LW but showed a slight decline at EO in 2017. Despite the 2022 data being reported as below LoD, taking the LoD as the maximum possible value (1 mg/kg) presents a continued decline in mercury at EO when compared with 2017 values (2.12 mg/kg).

Overall, despite some inter-annual variability, a general trend of decreasing mercury values with time was observed at EO following a sharp rise in mean concentration in 2009. There was some variance over time recorded at LE, although there has been a consistent decline since 2016 and continuing into 2022 (values of <1 mg/kg, compared with 1.17 mg/kg in 2017). There has been little to no changeability observed at sites HO and LW, although 2022 cannot be confirmed owing to the change in limit of detection.

Between 2010 and 2017, mercury levels fell rapidly with distance from the outfall with a peak at EO, considerably lower concentrations at LE, and even lower values at HO where concentrations were comparable to the reference site, LW. For instance, the mean concentration of mercury averaged from 2010 to 2017 (Appendix C; Figure C.2) at EO was 3.12 mg/kg, which fell by almost two thirds at LE, and to 0.54 mg/kg at HO (HO being similar to LW with 0.20 mg/kg).

3.1.1.3 Serrated wrack (*Fucus serratus*)

Mean mercury concentrations in *F. serratus* at site EO, near the outfall, showed a marked increase post-commissioning, 2009 having values which were more than two orders of magnitude greater compared to 2007 (Figure 3.1 (iii)). Similarly, in 2008 (when sampling was conducted shortly after initial commissioning) the mean concentration was over one order of magnitude greater than that reported in the previous year. Despite a decrease in mean mercury concentration in 2010 to levels similar to those recorded in 2008, mercury values were appreciably higher in 2011. Since 2011, mean mercury concentrations have remained high at EO, although some variability has been recorded with decreases in 2012, 2014, 2016 and 2022. The 2015 mercury concentrations were no exception, EO mercury levels being 47.5 times greater than that recorded at the reference site LW in the same year. In 2016 mercury concentrations at EO fell to the lowest level since 2010 but increased again in 2017, though have subsequently decreased again in 2022. In contrast, there is little to almost no variation at the other sites sampled.

The mercury concentrations in *F. serratus* at EO were always higher than all other sites for each year. Furthermore, concentrations of mercury in *F. serratus* rapidly declined with distance from the outfall. Subsequently, although mercury levels were slightly elevated at LE, and although these were still very low they were an order of magnitude higher than the values recorded at the reference site LW. This is further demonstrated when the mean mercury concentrations are averaged from 2010 to 2017 for all sites (Appendix C; Figure C.3), EO having the highest overall mean of 1.24 mg/kg which is four times greater than that of LE (LE mean = 0.30) and almost 25 times greater than LW (LW mean = 0.05). The values for 2022 cannot be compared with these trends owing to the LoD within the lab.

See the standardised metal concentration figure in the appendix for further evidence of the highlighted increase in mercury in 2009 at EO (compared to the discontinued site CON and WO), as well as its continual elevation in subsequent years compared to all other sites (Appendix B; Figure B.3).

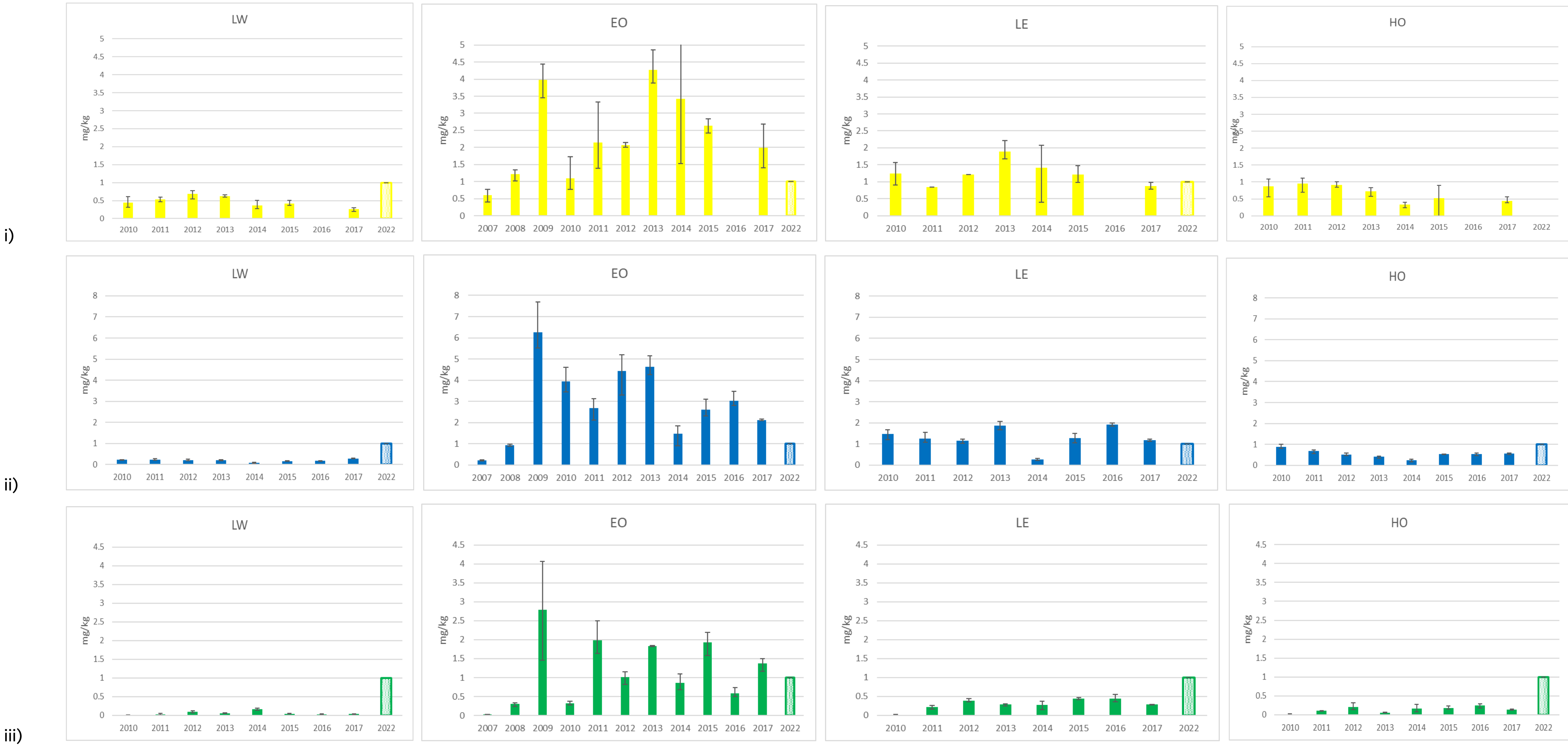


Figure 3-1. Mean concentrations (mg/kg dry weight) of mercury in the tissue of *Nucella lapillus* (i), *Patella vulgata* (ii), and *Fucus serratus* (iii) at each site (represented geographically west – east across the page). *N. lapillus* data not available for 2016 and for site HO in 2022. Error bars represent individual replicate concentration range. Data for 2022 were all below the limit of detection (1 mg/kg) and are represented as a patterned bar to provide an indication of potential levels.

3.1.1.4 Comparison with Natural Resource Wales Data

An investigation was made by NRW (previously EAW) in 2005 into the metal levels in the fucoid *Fucus vesiculosus* and in the sediment of the Severn Estuary. This study includes sites at Penarth and Berrow/Stert Flats, which are situated on the north and south banks of the mouth of the Severn Estuary, but also includes a site at Aberthaw. Mercury concentrations found in *Fucus serratus* from the present study have been compared to the data from the NRW sites, shown in Figure 3.2. Although two different fucoid species have been compared, it has been assumed that their rate of metal bioaccumulation would be similar, indicated by the similar levels at Aberthaw 2007/2008 in *F. serratus* with those in *F. vesiculosus* collected by NRW.

Prior to 2009 at site EO (and at the discontinued sites CON and WO, see Jacobs, 2016), mean levels of mercury in *F. serratus* were similar to those reported by NRW in 2005. However, levels of mercury at EO in 2009 are considerably higher than those reported within the Severn Estuary in 2005, being two orders of magnitude greater than that reported by NRW at Aberthaw in 2005 (EO 2009 (mean) = 2790.0, Aberthaw 05 = 23.8). Despite a decline in mercury levels at EO in 2010 to those seen prior to 2009, these values remained dissimilar to those reported by NRW. Levels of mercury remained high from 2011 to 2017 with little temporal variation. A decline in overall mean mercury values can be seen with distance from the outfall, and although levels in *F. serratus* at HO and LE are generally higher than the data from NRW, LW as the reference site, is comparable (LW (overall mean) = 57.0). The sites LE, HO and LW, similar to EO, also show a limited overall trend from 2011 to 2017, although an increase in values can be seen from 2010 to 2011. Data for all sites for 2022 were all below the lowest limit of detection (<1000 µg/kg) which is greater than previous years which makes identification of reductions harder to detect. Taking the LoD value as a maximum there was however a reduction at site EO in comparison to 2017 data.

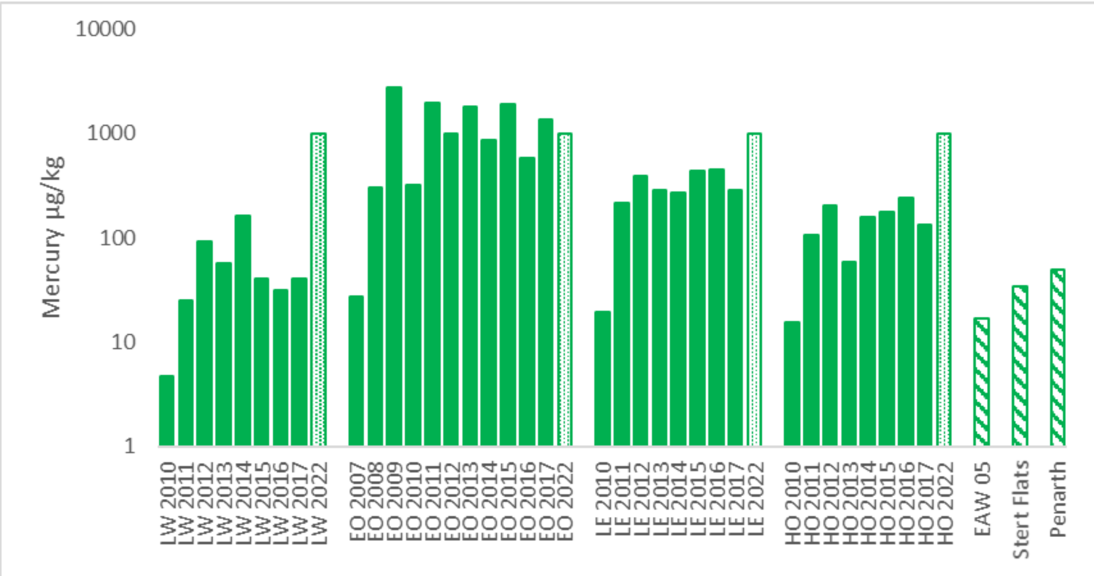


Figure 3-2. Concentration of mercury (µg/kg dry weight) in fucoids from Aberthaw and lower Severn Estuary. Solid and hatched bars represent Jacobs and NRW data, respectively. All values for 2022 were below limit of detection (1000 µg/kg) Note: data presented on a log scale.

3.1.1.5 Methylmercury in biota

Levels of MeHg were investigated from 2010 to 2022. In 2012, owing to low numbers of individuals present, no data were available for *N. lapillus* at LE, and similarly in 2022 due to low numbers of individuals no data was available for *N. lapillus* at HO. No data were available for mercury in 2016 at all sites. In 2011, 2014, 2015 and 2022 concentrations for *N. lapillus* at site LE were based on a single replicate only. Similarly, in 2015 and 2011, low numbers of *P. vulgata* at EO resulted in concentrations being based on single replicates. All faunal data are presented as dry weight (full results are presented in Appendix E).

Mean MeHg concentrations in *N. lapillus* sampled at the four sites in 2022, although being recorded as below LoD (<100 µg/kg), showed a reduction to those from 2017 which varied between 104 and 140 µg/kg. Taking the LoD of 100 µg/kg as a maximum the levels recorded in 2022 at LW and EO were the lowest recorded to date (since 2010), and LE being consistent with 2016 (Figure 3.3 A). No consistent pattern was evident in the distribution of MeHg in *N. lapillus* between years, although 2011 represented a peak in mean concentration at four of the six monitoring sites, nor was a pattern seen between sites. Some temporal variation was evident in the proportion of total mercury represented by mercury in MeHg in *N. lapillus* at each site (Figure 3.3 B). However, a consistent spatial pattern was evident in all years with the lowest proportions observed in the vicinity of the discharges with levels rising more rapidly at sites to the west of the discharges (LW having the highest proportions). Notably for 2017, site LW had the highest proportion of total mercury represented by mercury in MeHg in *N. lapillus* for any year, of 32 %.

Mean MeHg concentrations in *P. vulgata* sampled at the four sites in 2022, although being recorded as below LoD (<100 µg/kg), showed a reduction to those from 2017 which varied between 291 and 374 µg/kg. Taking the LoD of 100 µg/kg as a maximum the levels recorded in 2022 at LW and LE were the lowest recorded to date (since 2010), and EO and HP showing a large decrease compared with 2017 (Figure 3.3 A). The lowest proportion of total mercury represented by organic mercury in MeHg in *P. vulgata* generally occurred at sites in the vicinity of the outfalls in all years. The highest proportions were again reported at the westernmost sampling site (LW) (Figure 3.3 B).

Over the course of the monitoring period, mean MeHg concentrations were higher in *P. vulgata* compared to in *N. lapillus* for over two thirds of the samples.

Between 2010 and 2013 MeHg concentrations (wet weight) in *F. serratus* were consistently below the minimum reporting value (MRV) of 1 µg/kg, while in 2014 one replicate at sites HO, EO and LW returned values greater than the MRV with a maximum concentration of 10 µg/kg recorded at site EO. In 2015 one replicate at sites HO and LE returned values greater than the MRV with a maximum concentration of 15 µg/kg recorded at site LE. 2016 only returned three results greater than the MRV at EO and LW (1µg/kg) and at LE (2 µg/kg). In 2017, across all sites a total of six values were returned that were greater than the MRV, although the values were still very low, the maximum value recorded being 2 µg/kg at site LW. Patterns in 2022 are not detectable owing the difference in LoD.

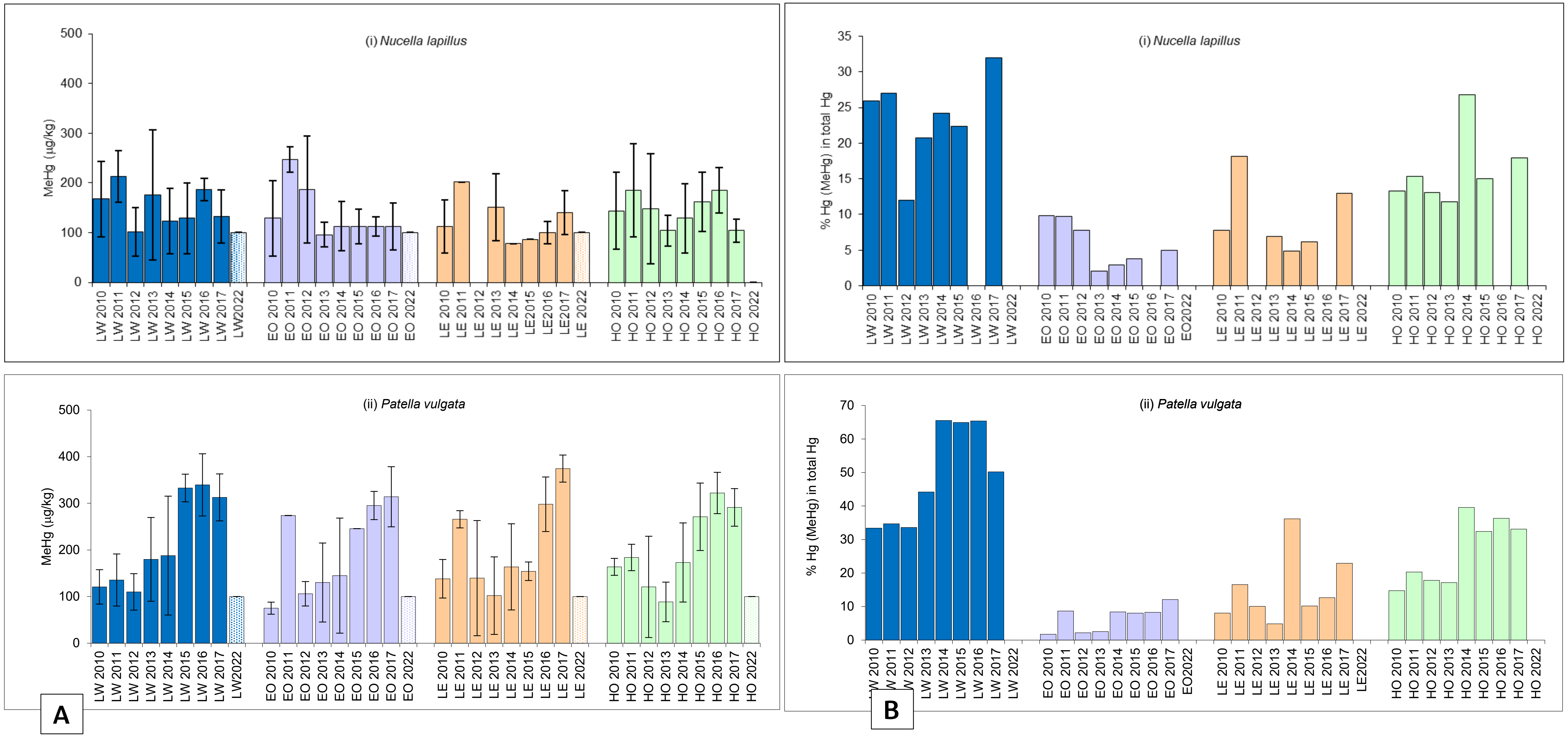


Figure 3-3. Mean methylmercury (MeHg) concentration (µg/kg dry weight) in *Nucella lapillus* and *Patella vulgata* 2010 - 2022. *N. lapillus* data not available for 2016 or for site HO in 2022. 2022 data are all below the limit of detection (100 µg/kg). Error bars represent standard deviation (no bar indicates no data). NB: the graphs are approximate as some organic mercury may be converted into inorganic mercury during the analytical process and therefore are included in the inorganic result (however, it is probable that most of it will not be included in the inorganic result).

3.1.2 Other metals in biota

The highest individual replicate concentrations of copper (Cu), zinc (Zn), arsenic (As) and selenium (Se) occurred in *N. lapillus*, while those for cadmium (Cd), lead (Pb) and chromium (Cr) were recorded in *P. vulgata*, although the maxima were not all recorded at the same site or in the same year. The highest individual replicate level of nickel (Ni) was found in *F. serratus* at site LW in 2014 (Table 3-1).

The highest mean concentrations of zinc, mercury, arsenic, chromium and nickel corresponded to the occurrence of the highest individual replicate concentrations. The majority of the lowest mean metal levels corresponded to the occurrence of the lowest individual replicate metal concentrations.

In 2022, there were no exceedances of individual replicate and mean metal concentrations in biota from 2007-2022 above the maxima for the monitoring period.

3.1.2.1 Dogwhelk (*Nucella lapillus*)

The mean metal concentrations for *N. lapillus* are reported in Figure 3.4, which shows very little spatial variation in the metals: cadmium; chromium and nickel. When comparing the mean metal levels averaged between 2010 and 2022 at HO, LE, EO and LW (Appendix C; Figure C.1), the greatest levels of zinc, lead, arsenic, cadmium, selenium and chromium occurred at EO, and levels of these metals generally remained higher to the east of the discharge compared to the west, showing decreasing levels of metal concentration with distance from the outfall. However, in comparison to mercury (Section 3.1.1), the between site differences were less marked. Copper showed a similar trend of spatial variation but the highest averaged mean concentration was recorded at site LE, although concentrations were still higher at the outfall than further away (e.g. sites HO and LW). The highest mean concentrations of copper, zinc, and lead occurred at EO in 2013 whilst that for selenium was at the same site in 2014 (Table 3.1). Highest mean concentrations of cadmium, arsenic and chromium were recorded at WO in 2007, and the highest mean nickel concentration was recorded in individuals collected from CON in 2013. Levels of metals in 2022 showed a decrease when compared to 2017 at all sites, with levels also being below pre-commissioning levels at site EO.

Despite there being some temporal variation in the concentration of metals across all sites, there were no apparent differences between the level of metals from pre and post-commissioning, as is evident from the standardised metal concentration figures (Appendix B; Figure B.1). No metal concentration values in *N. lapillus* were noticeable beyond the range of what has been previously recorded across the monitoring period in the data from the 2022 survey.

3.1.2.2 Limpet (*Patella vulgata*)

With the exception of nickel and chromium the highest mean metal concentrations in *P. vulgata* were recorded at EO with the highest mean copper and lead in 2010 and the highest zinc, arsenic and selenium in 2016 (Table 3.1). The highest mean cadmium concentration was recorded in 2013, although this level was elevated by a particularly high concentration in one replicate (214 mg/kg) which was confirmed as correct by the analysing laboratory. The highest mean nickel concentration was recorded in 2010 at LE and for chromium in 2011 at HO.

Mean metal concentrations in *P. vulgata* are presented in Figure 3.5. Across all metals there is only slight variation in concentration spatially, with no discernible trend. This is with the exception of zinc, arsenic, lead and selenium which were consistently higher at EO than the other sites when comparing mean metal concentrations averaged between 2010 and 2022 (Appendix C; Figure C.2), although this elevation is not markedly greater for zinc and lead. These metals generally showed greater concentrations to the east of the outfall but not to a greatly discernible extent. The standardised metal concentrations in *P. vulgata* (Appendix B; Figure B.2) clarifies these findings, showing arsenic to have a generally consistent elevation at site EO compared to other sites, although this increase was much less than for mercury. For other metals standardised concentrations were generally close to one with no consistent spatial patterns evident.

Across all metals, levels in *P. vulgata* showed changeability over time but with no apparent temporal trend. In particular there is no visible difference between pre and post-commissioning. In 2016, the metals arsenic, selenium and zinc all had the highest mean metal concentrations recorded across the monitoring period. For 2022, in most instances these concentrations had decreased to lower levels with no notably high metal concentrations found in *P. vulgata* compared to previously recorded values across all years.

3.1.2.3 Serrated wrack (*Fucus serratus*)

In *F. serratus* the highest mean zinc concentrations occurred at LW in 2014 and highest mean lead concentrations occurred at EO in 2015 (Table 3.1). The highest mean chromium occurred at LW in 2010 and the highest selenium concentrations were recorded at WO in 2010. The highest arsenic and nickel concentrations occurred at site LW in 2013 and 2014, respectively. The highest mean cadmium level was recorded at CON in 2007 and the highest copper was recorded at EO in 2016.

When averaged between 2010 and 2022 the highest mean metal concentrations for copper, lead, zinc, arsenic, chromium and selenium occurred at EO (Appendix C; Figure C.3). Copper, zinc and arsenic were the only metals that exhibited decreasing metal concentrations with distance from the outfall and higher concentrations to the east. However, these levels were only slightly higher and therefore no obvious trend could be determined. This is exemplified in the standardised metal concentrations figure for *F. serratus* (Appendix B; Figure B.3) where for all metals, values were generally close to one, indicating no consistent spatial patterns.

Temporal variation was detected across all metals, but trends were identified in the metals zinc, copper and cadmium. There was an almost consistent increase in zinc concentrations each year at HO and LE since monitoring began, with EO also exhibiting an (albeit less consistent) increase in concentrations particularly in the last four years. Copper also showed an increase in concentrations at EO, LE and HO. Cadmium showed similar trends at LE and HO (again, less so at EO). Of these metals that have been identified mean concentrations in 2022 have shown a decline since 2016 with no maximum mean or individual replicate concentrations reported in 2022 (Table 3.1).

Table 3-1. Range of individual replicate and mean metal concentrations (mg/kg dry weight) in biota 2007-2022 (2022 values in bold; no data for *Nucella lapillus* in 2016); overall maximum values for the monitoring period indicated in red; underlined italicised values represent pre-commissioning of the FGD. *=<0.1 for LE and CON 2013, EO 2012 and LW 2017. # = <1 LE, EO, LW and HO 2022.

Metal	*Fucus serratus*				*Patella vulgata*				*Nucella lapillus*			
	Individual replicate concentration		Mean concentration		Individual replicate concentration		Mean concentration		Individual replicate concentration		Mean concentration	
	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
Cu	**1.1 EO 2022**	16.6 EO 2016	**1.53 EO 2022**	14.7 EO 2016	**2.2 LE 2022**	31.3 LE 2012	**3.33 LE 2022**	23.8 EO 2010	1.71 LW 2010	868.0 LE 2014	**42 LE 2022**	718.7 EO 2013
Zn	**38 EO 2022**	681 LW 2014	**44 EO 2022**	660 LW 2014	**25 LE 2022**	276 EO 2010	**27.33 LE 2022**	245 EO 2016	**174 LE 2022**	2900 EO 2013	**174 LE 2022**	2426.7 EO 2013
Cd	**# See above**	*7.41 CON 2007*	**# See above**	*6.29 CON 2007*	**2.2 LE 2022**	214.00 EO 2013	**4.3 LE 2022**	100.8 EO 2013	6.03 HO 2014	*134.00 WO 2007*	**8.6 LE 2022**	*111.5 WO 2007*
Hg	0.003 LW 2010	4.07 EO 2009	0.008 CON 2010	2.79 EO 2009	0.057 LW 2014	7.68 EO 2009	0.092 LW 2014	6.25 EO 2009	0.198 LW 2017	5.04 EO 2014	0.263 LW 2017	4.273 EO 2013
Pb	0.29 LE 2010	2.55 EO 2014	0.33 LE 2010	2.03 EO 2015	0.454 LE 2014	15.50 LE 2013	0.95 LW 2014	10.99 EO 2010	0.63 CON 2010	4.57 EO 2013	0.92 LW 2010	3.87 EO 2013
As	**3.8 EO 2022**	95.7 LW 2013	**4.7 EO 2022**	77.0 LW 2013	**3.8 LE 2022**	41.6 EO 2016	**4.7 LE 2022**	40 EO 2016	**10 EO 2022**	*152.0 WO 2007*	**13 EO and LW 2022**	*122.7 WO 2007*
Cr	0.2 LE 2011	6.86 WO 2009	0.27 LE 2011	3.66 LW 2010	0.335 LW 2014	37.70 HO 2011	0.59 LW 2014	18.8 HO 2011	0.093 EO 2012	*6.25 WO 2007*	0.445 LE 2010	*4.38 WO 2007*
Ni	1.18 LE 2010	29.00 LW 2014	1.54 LE 2010	26.7 LW 2014	**1.00 LE 2014 and 2022**	18.00 HO 2011	**1 LE 2022**	12.5 LE 2010	<0.3 EO 2011	9.88 CON 2013	0.42 EO 2011	6.35 CON 2013
Se	* See above	1.13 LE 2010	* See above	0.65 WO 2010	0.30 LW 2014	6.49 EO 2016	0.40 LW 2014	5.93 EO 2016	*1.00 CON 2007*	*10.80 WO 2008*	*2.67 CON 2007*	7.7 EO 2014

Figure 3-4. Mean concentrations (mg/kg dry weight) of metals in the tissue of the dogwhelk *Nucella lapillus* at each site (represented geographically west-east across the page). Data not available for 2016 or for site HO in 2022. For 2022 data where the values are below the limit of detection (1mg/kg) are shown in yellow with a patterned fill. Error bars represent individual replicate concentration range: (i) Arsenic, (ii) Cadmium, (iii) Chromium and (iv) Copper;

v) vi) vii)

Figure 3-4 cont. (v) Lead, (vi) Nickel, (vii) Zinc.

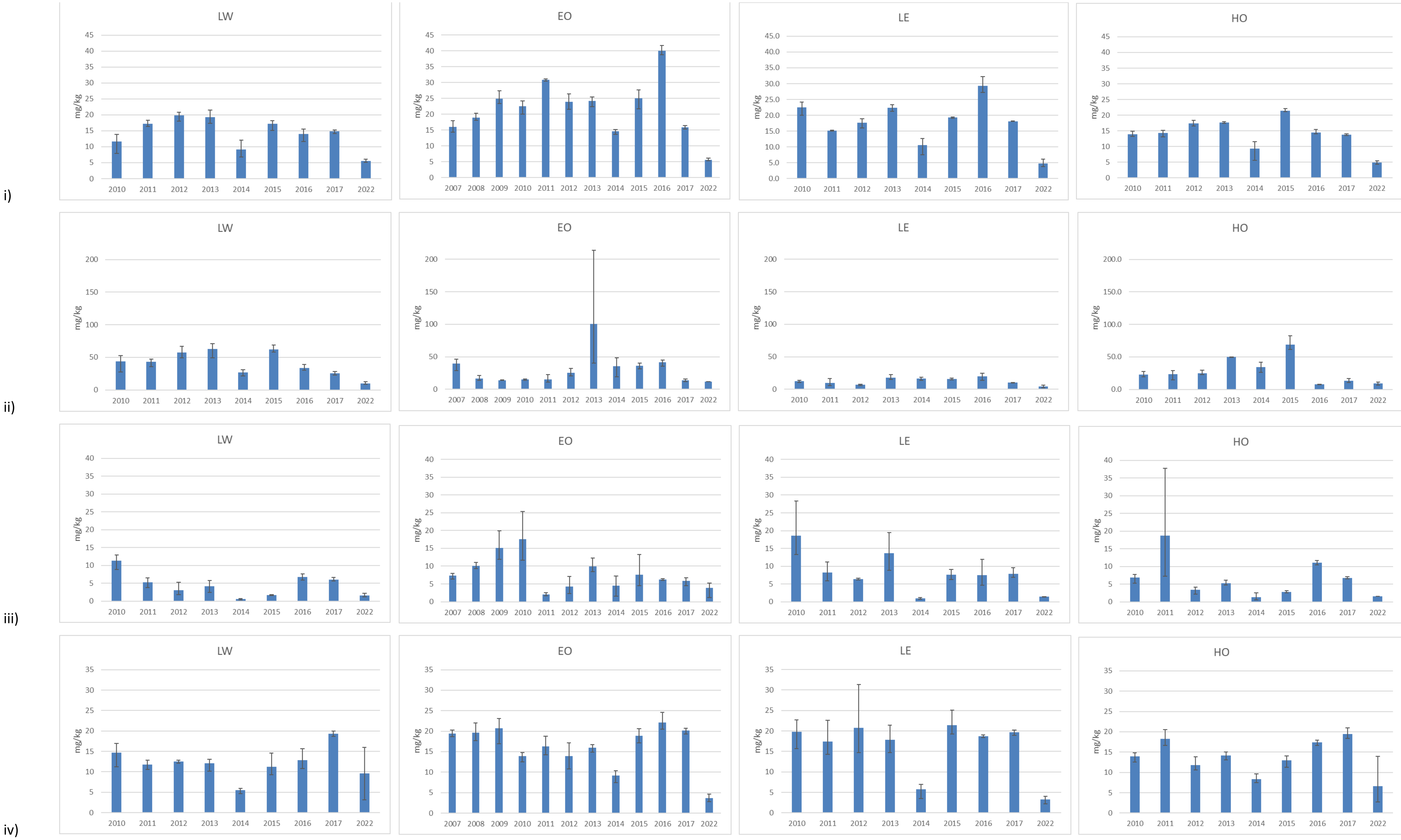
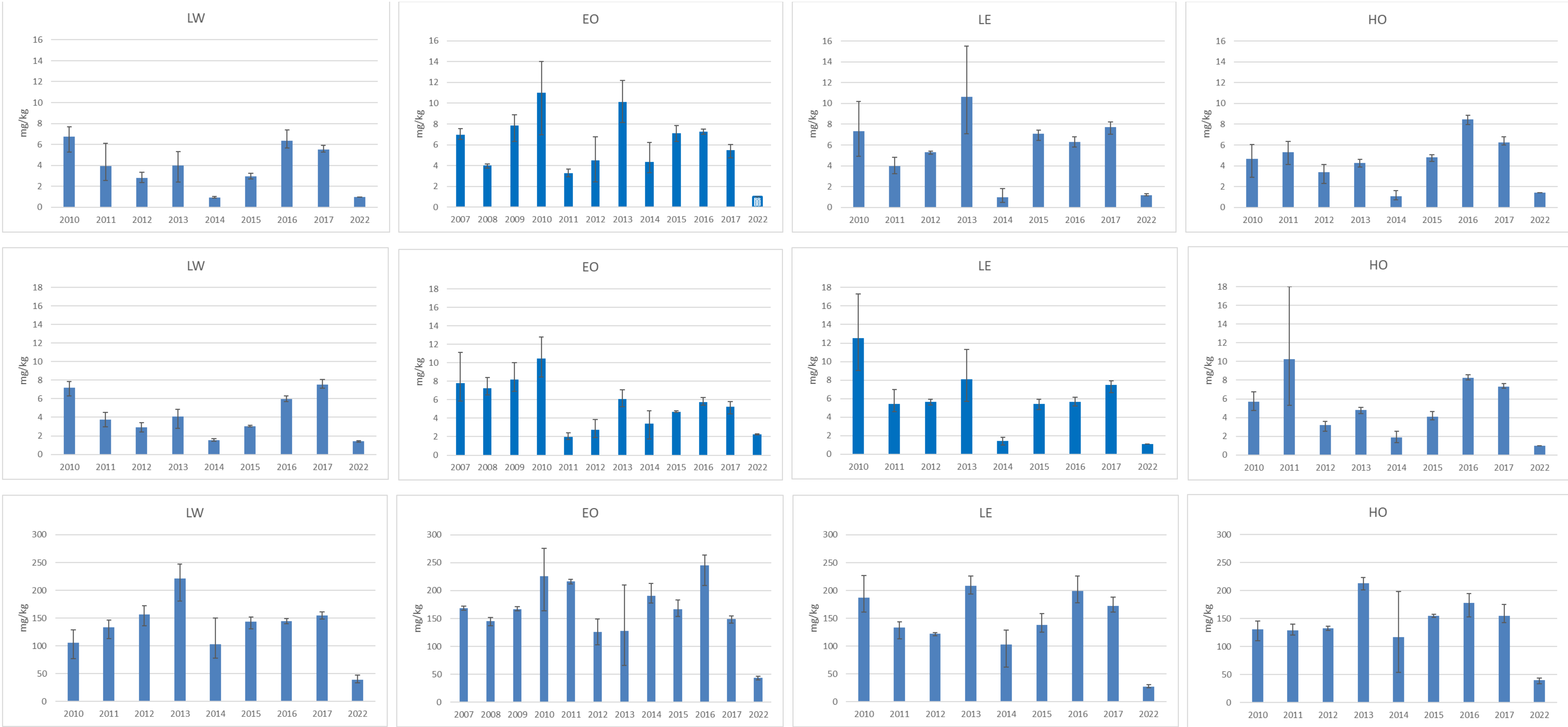


Figure 3-5. Mean concentrations (mg/kg dry weight) of metals in the tissue of the limpet *Patella vulgata* at each site (represented geographically west-east across the page). For 2022 data where the value is below the limit of detection (1mg/kg) this is shown in blue with a patterned fill. Error bars represent individual replicate concentration range: (i) Arsenic, (ii) Cadmium, (iii) Chromium, (iv) Copper;



v)

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vii)

Figure 3-5 cont. v) Lead, (vi) Nickel, (vii) Zinc.

Figure 3-6. Mean concentrations (mg/kg dry weight) of metals in the tissue of seaweed *Fucus serratus* at each site (represented geographically west-east across the page). For 2022 data where the value is below the limit of detection (1mg/kg) this is shown in green with a pattern fill. Error bars represent individual replicate concentration range: (i) Arsenic, (ii) Cadmium, (iii) Chromium, (iv) Copper;

v)

vi)

vii)

Figure 3-6 cont. v) Lead, (vi) Nickel, (vii) Zinc.

3.1.3 Biometric analysis

Biometric data is not available prior to 2017 for the Aberthaw FGD sampling sites as these data had not previously been collected. The mean conditional indices (CI) recorded at each site in 2017 and 2022 for both *N. lapillus* and *P. vulgata*, can be seen in Table 3.2.

Data from 2017 revealed (for both species studied) that site EO had significantly higher conditional indices than all other sites.

In 2022 data obtained for *N. lapillus* was analysed using an analysis of variance (ANOVA) having passed the normality and equal variance tests ($p = > 0.05$). The results showed a significant difference in conditional indices between sites in 2017 (ANOVA: $p = < 0.05$; d.f. = 3). The results of the post hoc test (Tukey Test) to identify at which sites there was a significant difference ($p = < 0.05$) revealed a significant difference between sites EO and LW accounted for by lower values for CI at site EO relative to LW. All other combinations of sites showed no significant difference ($p = > 0.05$).

The conditional indices for *P. vulgata* across the sampling sites in 2022 passed the test for normality and therefore an analysis of variance (ANOVA) was used to analyse the data. The results of the analysis showed a significant difference (ANOVA: $p = < 0.001$; d.f. = 3). The results of the post hoc test (Tukey Test) to identify at which sites there was a significant difference ($p = < 0.05$) revealed a significant difference between site LE and all other sites accounted for by lower values for CI at this site. No significant differences were recorded for sites LW, EO and HO.

Table 3-2. Mean conditional indices for *N. lapillus* and *P. vulgata* at each site, reported for 2017 and 2022 (brackets indicate standard deviation; values reported to 2 significant figures).

Species	Year	LW	EO	LE	HO
Nucella lapillus	2017	0.0045 (± 0.0014)	0.0057 (± 0.0011)	0.0043 (± 0.0011)	0.0035 (± 0.00085)
	2022	0.0045 (± 0.00096	0.0036 (± 0.00094)	0.0038 (± 0.0015)	0.0039 (± 0.00079)
Patella vulgata	2017	0.011 (± 0.0019)	0.013 (± 0.0027)	0.012 (± 0.0022)	0.011 (± 0.0023)
	2022	0.015 (± 0.0057)	0.012 (± 0.0058)	0.0085 (± 0.0017)	0.012 (± 0.0036)

3.2 Sediments

Sediment data collected in 2022 are presented in this section, along with some comparisons to previous years and data from other studies.

3.2.1 Sediment granulometry

Granulometry results reported between 2007 and 2022 can be seen in Figure 3.7 and the concurrent sediment classification diagrams in Figure 3.8. It must be noted that owing to the analytical method used by the laboratory (mechanical sieving only) between 2007 and 2011 and again in 2022, the finest particle size reported was <62.5µm. Therefore, between 2007 and 2011 the clay and silt fraction have been amalgamated.

For the 2022 data three fractions were recorded: gravel (>2mm), sand (63-2000 µm) and silt (<63µm). The granulometric analysis of the sediment samples from Limpert Bay, indicated that in 2022 the sediment was composed predominantly of silt and sand, the greatest proportion being silt (raw granulometric data for 2022 is given in Appendix D). Sand was the primary fraction that the sediment comprised from 2007 to 2017 with a smaller but still significant proportion of silt and clay, although the proportion of each size fraction varied across years. The majority of years were classified as muddy sand (2011, 2012, 2013, 2015, 2016) or sand (2008, 2009). There are however some exceptions such as in 2007, 2017 and 2022, where silt and clay represented over 50% of the sediment with the remainder of the material composed of fine and medium sand (classed as sandy mud). Similarly, 2010 and 2014 had even higher proportions of silt and clay, characterising over 90% of the material with fine sand (classed as mud and silt respectively) making up the remaining percentage. This is indicative of highly mobile sediments within an environment with strong tidal currents causing large amounts of sediment transport. As a consequence of this much of the sediment sampled is unlikely to have been resident in the area around Aberthaw for a long period of time.

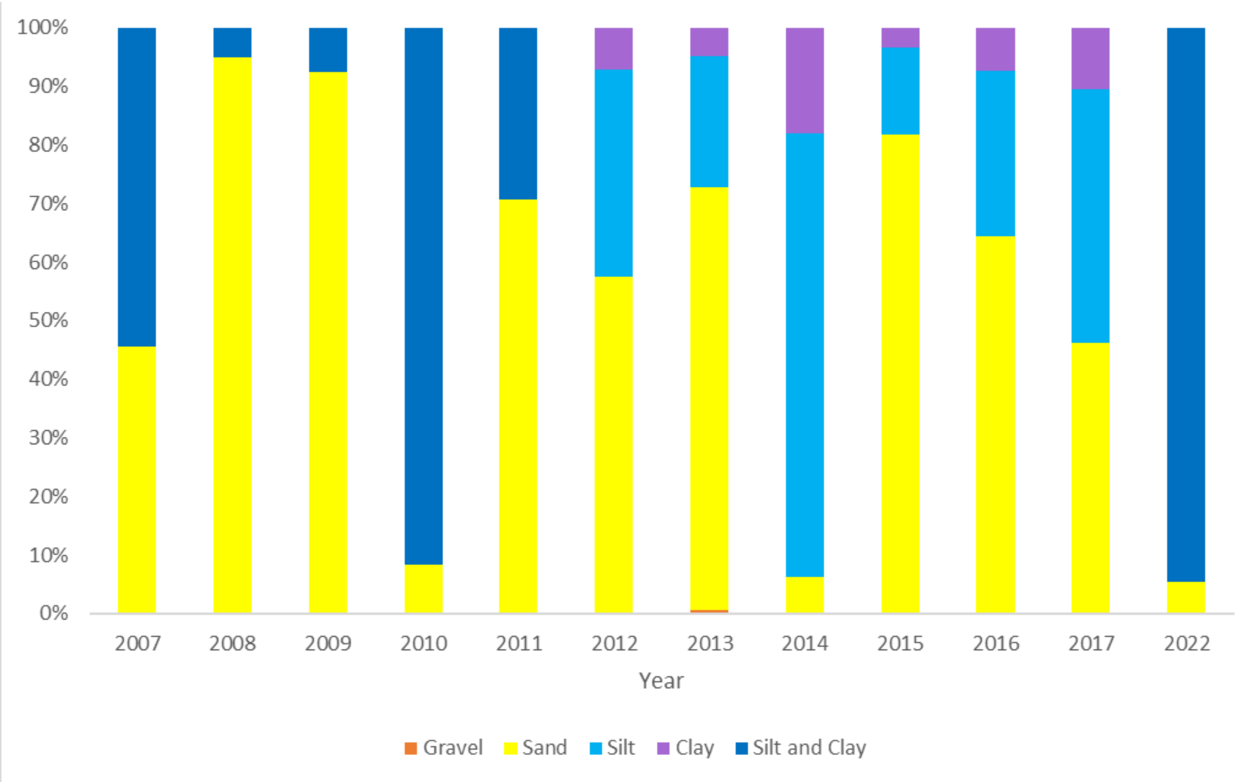

Figure 3-7. Mean particle size distribution reported from Aberthaw sediment sites between 2007 and 2022 (n = 5).

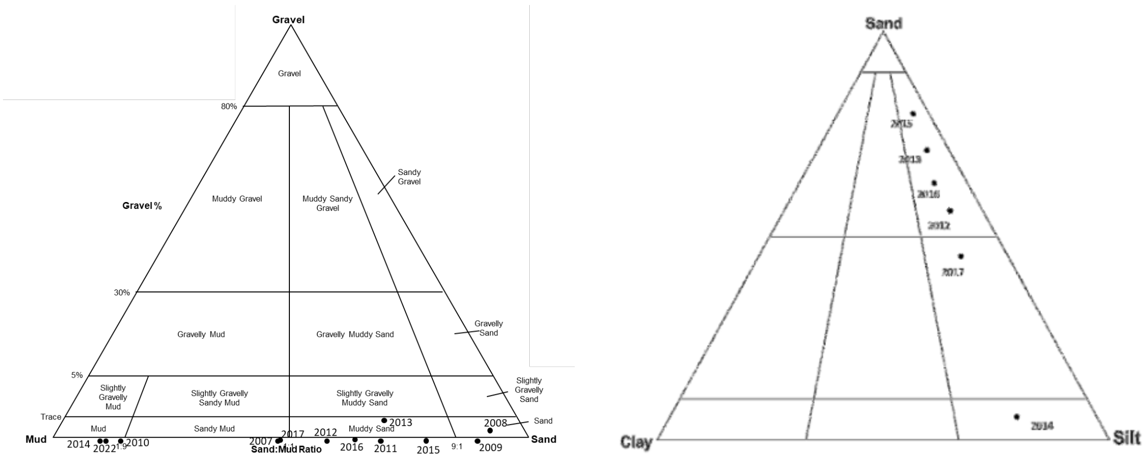

Figure 3-8. Mean particle size classification for Aberthaw sediment sites from 2007-2022 (n = 5) following the scheme and nomenclature devised by Folk (1954).

3.2.2 Sediment-bound metals

Sediment-bound metal concentrations (including mercury) at the sediment site are illustrated in Figure 3.9 and data are given in Appendix D. Some temporal variation was evident across all metals, although no clear trend could be detected. There is however an apparent difference in the level of some metals between pre and post-commissioning, the concentrations of metals from 2009 to 2022 tending to be lower than those of 2007 and 2008, particularly for copper and cadmium. The greatest range recorded over the study period was for copper where the highest recorded concentration was over 15 times the lowest; the narrowest recorded range was for arsenic where the highest concentration was less than three times the lowest. Throughout the study period, sediment bound concentrations of the majority of metals exceeded the relevant ISQGs but were below the PEL (Figure 3.9). However, in 2008 and 2013 mean levels of nickel were greater than the PEL. Subsequently, 2017 saw nickel concentrations much closer to the ISQG following a general decline each year since the 2013 peak. While values for nickel increased in 2022 they were still below the PEL and the mean value for the entire monitoring period. Conversely, cadmium exhibited maximum concentrations in 2007 which were over two times lower than the ISQG. Similarly, mean mercury concentrations post-commissioning were often below the ISQG, only exceeding this value on five occasions (including 2022) and never by twice as much. When considered over the study period as a whole, mean concentrations of cadmium and mercury were less than the relevant ISQG while those for all other metals exceeded this lower limit.

Overall, all mean sediment concentrations in 2022 were lower than the maxima recorded over the whole study period, showing an increase from the values recorded in 2017. This is with the exception of mercury and chromium which saw the mean concentration in 2022 decrease from that of 2017.

As sediments were not analysed for aluminium in 2007, metal:aluminium ratios could only be calculated for subsequent years (Table 3.3). Across the sampling period, very few metals had values more than twice the upper BRC throughout the study period with the exception of data from 2022 where copper, zinc, cadmium, mercury and lead exceeded twice the upper BRC (discussed in section 4.2). At the sediment site, values of the metal:aluminium ratio for mercury have exceeded more than twice the upper BRC since 2008; 2022 seeing an increase of 87% of that recorded in 2017. Likewise, except for 2012 and 2015 values for lead were considered elevated in all years. Values for the copper:aluminium ratio were greater than twice the upper BRC at the sediment site in 2008, while for cadmium the ratio was greater than twice the upper limit in 2009. All values for metals in 2022 showed a substantial increase in metal:aluminium ratio compared to 2017. At the same time aluminium concentration in the sediment decreased by approximately half between 2017 and 2022. Since 2014 aluminium levels in the sediment have decreased year on year (Figure 3.10).

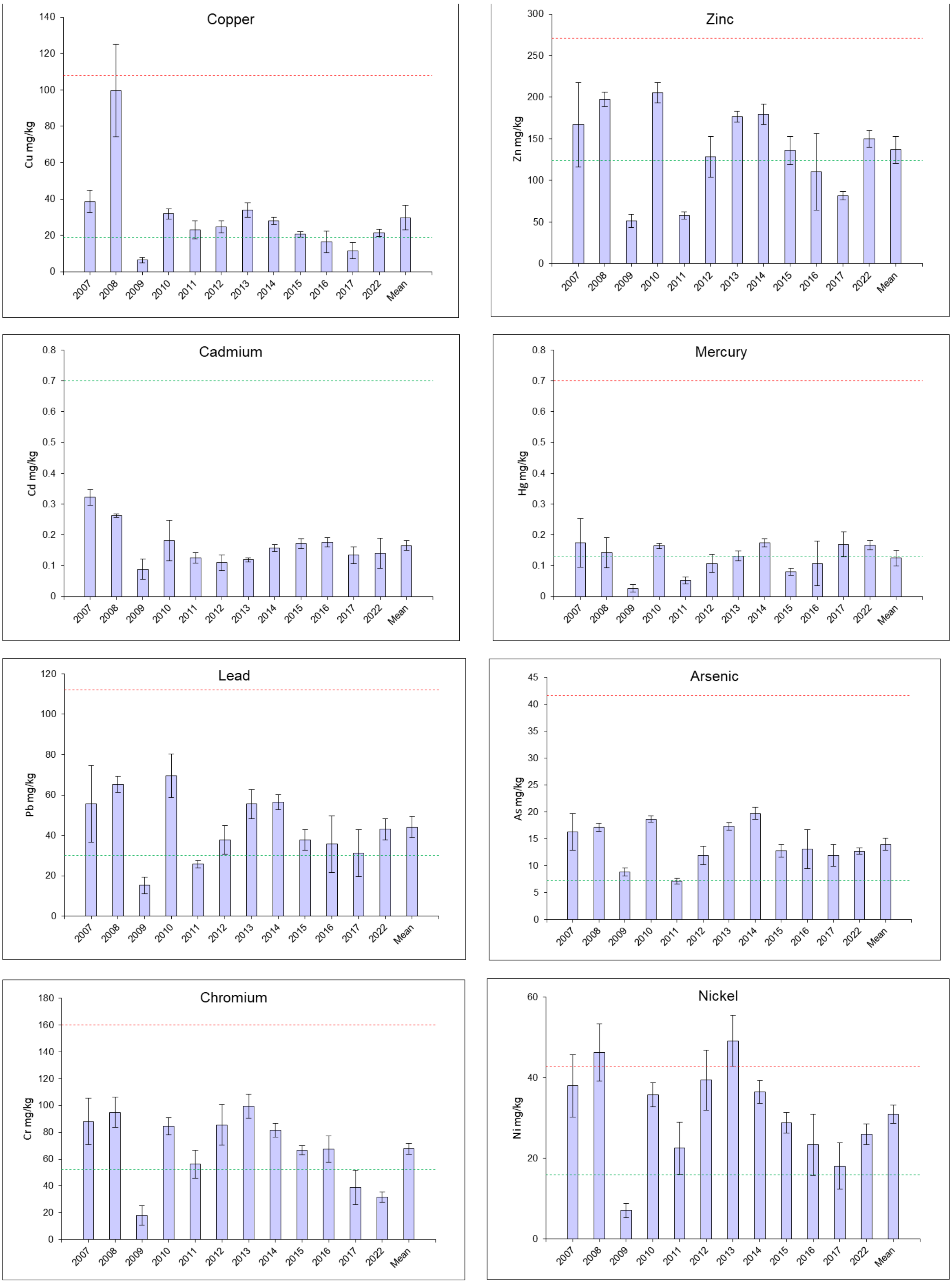

Figure 3-9. Mean sediment-bound metal concentrations (mg/kg) at the Aberthaw sediment site 2007 – 2022 (n= 5). Error bars represent standard deviation; dotted lines represent ISQG (green) and PEL (red). Where a dotted line is missing the ISQG and/or PEL is greater than the scale of the graphs.

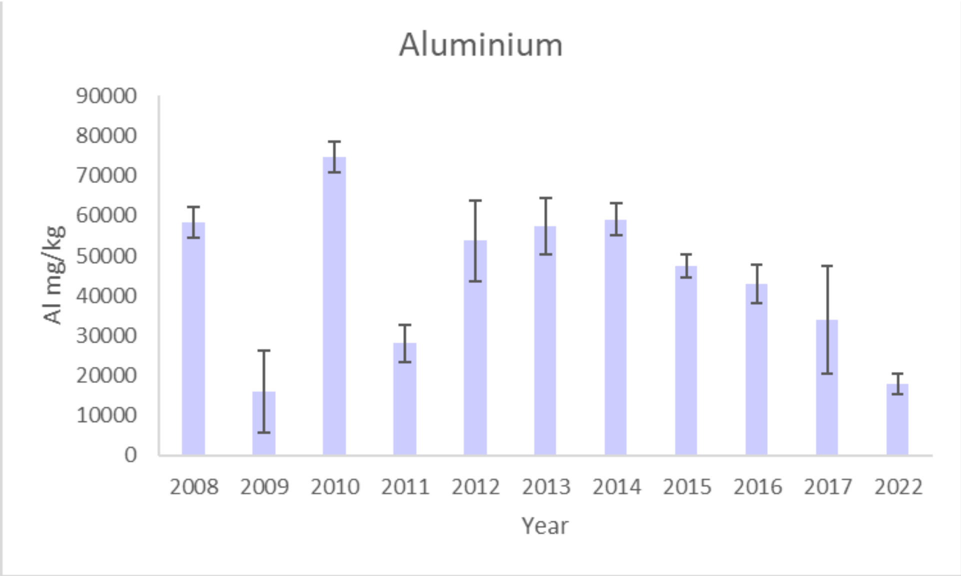

Figure 3-10. Mean sediment bound aluminium concentrations (mg/kg) at the Aberthaw sediment site 2007 – 2022 (n= 5). Error bars represent standard deviation.

Table 3-3. Background Reference Concentration (BRC) range for sediment-bound metals and values for metal:aluminium ratios x10[4] for sediments at Aberthaw, 2008 – 2022. Data from Environment Agency Wales (EAW) and other study data (Stert Flats and Penarth) also presented. Data in red denote a metal:aluminium ratio > 2x upper BRC (See section 2.2).

	BRC Range	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2022	EAW 05	Stert Flats	Penarth
Copper	2.2 - 4.5	17.3	4.0	4.3	8.2	4.6	5.9	4.7	4.3	3.8	3.4	12.0	15.6	6.9	6.9
Zinc	8.8 - 18	33.9	33.2	27.5	28.4	24.1	30.7	30.3	28.7	25.7	24.0	84.32	32.6	32.5	41.0
Cadmium	0.007 - 0.03	0.045	0.061	0.024	0.044	0.021	0.021	0.027	0.036	0.041	0.040	0.08	0.081	0.072	0.062
Mercury	0.0034 - 0.0066	0.0250	0.0158	0.0220	0.0186	0.0202	0.0229	0.0294	0.0168	0.0249	0.0498	0.0931	0.0170	0.0340	0.0500
Lead	1.8 - 4.0	11.2	9.6	9.3	9.2	7.1	9.7	9.5	7.9	8.3	9.2	24.17	10.4	10.7	13.6
Arsenic	2.0 - 4.5	2.9	5.8	2.5	2.5	2.2	3.0	3.3	2.7	3.0	3.5	7.13	6.0	2.4	2.8
Chromium	9.0 - 20	16.3	11.0	11.3	20.1	16.1	17.4	13.8	14.0	15.7	11.4	17.73	24.9	12.8	13.9
Nickel	4.4 - 9.1	8.0	4.6	4.8	8.1	7.4	8.5	6.2	6.1	5.4	5.3	14.59	13.7	6.3	7.5

3.2.3 Sediment-bound methylmercury

Levels of methylmercury (MeHg) in the sediments at the SED site has also been assessed as part of the FGD monitoring programme since 2010. From 2010-2014, the dry weight concentration of MeHg in the sediment at Aberthaw ranged between < 1 and 12 µg/kg and since 2015 the concentration of MeHg has ranged between < 1 and 2 µg/kg. These values are considered low, although MeHg levels were recorded as high as 12 µg/kg in a sample taken from site CON in 2010 (however, this site is no longer surveyed). In 2022 all samples were found to have values below the lowest limit of detection (<100 µg/kg).

3.2.3.1 Comparison with Natural Resource Wales data

The majority of metal:aluminium ratios recorded in sediments in the present study were consistent with those derived from sediment-bound metals data collected from the lower Severn Estuary (Table 3.3). Sediment bound metals in 2022 exhibited metals:aluminium ratios that were higher than those reported from the lower Severn Estuary, for all metals. Mercury metals:aluminium ratio for 2022 was five times greater than levels found by NRW at Aberthaw 2005 and was greater than levels reported at Penarth and Stert Flats. This is likely to have been caused more by a reduction in aluminium levels in the sediment rather than rising mercury levels. Aluminium values for 2022 (17820 mg/kg) continued a trend of reducing levels from a peak in 2010 (74680 mg/kg).

With the exception of data from 2022 during the rest of the study period most of the copper, zinc and arsenic concentrations were less than twice the upper BRC in the lower Severn Estuary, found in vicinity of Aberthaw. The elevated mercury and lead concentrations recorded from the sediments collected at Aberthaw were consistent with data from the lower Severn Estuary. Levels of cadmium in sediments from the lower Severn were clearly elevated, while at Aberthaw a similar level of elevation was only recorded in 2009 and 2022. For chromium and nickel all concentrations at Aberthaw and at sites sampled by NRW in 2005 were not considered as elevated.

4. Discussion

4.1 Survey area

The initial survey report by Jacobs (2006) described the limitations to any survey in the vicinity of Aberthaw. The report highlighted that as a consequence of the high background contaminant levels and high tidal flow velocities in the lower Severn Estuary and upper Bristol Channel, determining any patterns in relation to the FGD discharge may be problematical.

In the initial survey design the spatial extent of the study was influenced by the local coastal morphology with adjacent shores upstream and downstream of Aberthaw being markedly different from that within Limpert Bay. Consequently, the selection of a suitable reference site remote from the influence of the discharge was constrained somewhat by the distance at which such a site could be established. However, following the findings of the 2009 survey it was concluded that a greater spatial extent needed to be monitored to better determine patterns in relation to changes to the FGD discharge. Although this necessitated sampling from areas with some variability in substrate morphology, it was deemed that this would have little effect on the likely bioaccumulation by the selected target species.

Although the number of sites sampled was reduced in 2014, the survey encompassed the same spatial extent as that surveyed between 2010 and 2013. The removal of sites in the western half of the survey area was deemed appropriate due to the consistent patterns observed in the data. It was considered that their continued inclusion would add little to the study, while the continued use of the most westerly site (LW) would act as a suitable reference point. As the spatial patterns observed in 2014 were similar to those in previous years, a pattern reinforced by the 2022 results presented here, it was considered that this revised survey design is suitable for continued monitoring at Aberthaw.

4.2 Sediments

Both pre- and post-commissioning sediment-bound metal concentrations were consistent with data collected from the lower Severn Estuary (NRW unpublished data) and also historical data from other locations (Table E.1). Overall, metal concentrations can be considered to be elevated above background, although this is likely to be related to chronic anthropogenic inputs into the Severn Estuary as a whole (see Langston *et al*., 2010) rather than any specific point discharge. It is likely that while sediment-bound metal concentrations were elevated, the temporal variability evident for all metals is likely to be related to geochemical make-up of the sediments, which have shown variability between sampling seasons. Duquesne *et al*. (2006) found that the levels of metals bound to suspended particulates at a variety of sites in the Bristol Channel and Severn estuary were remarkably consistent reflecting the large mixing capacity of the estuary. Re-exposure of older sediments and redistribution within an estuarine environment subject to strong tidal influences (such as at Aberthaw) would be expected to cause variability in mobile sediment characteristics.

In 2022 while the levels of all metals recorded in the sediment were consistent with previously recorded levels metal;aluminium ratios were in almost all cases higher than those previously recorded. Rather than an actual increase in metal levels this has occurred due to the decreasing aluminium concentrations in the sediment. In turn this is likely to have been caused by changes to the geochemical and physical make-up of the sediments. Particle size analysis of the sediment samples showed that the 2022 samples were composed of substantially more clay and silt than in most previous years (with the exception of 2010 and 2014). This decrease in particle size would indicate a high degree of sediment transport both into and out of the sample site. As the sediment sample site is an area of shore where sediment deposition has taken place to produce a relatively shallow layer of sediment over the bedrock it is likely that there is a very high degree of sediment mobility. The observed results for metal levels may be more representative of the suspended sediments of the Severn estuary and Bristol Channel as a whole rather than being influenced by any local inputs.

With the exception of cadmium, (and mercury post-commissioning) the mean pre- and post-commissioning concentrations of sediment-bound metals would indicate the possibility of adverse biological effects as levels exceeded the relevant ISQGs. It is likely that any biological effects would be chronic rather than acute.

4.3 Dogwhelk (*Nucella lapillus*)

During the study period (2007-2022) the highest metal levels recorded each year in biota were often observed in *Nucella lapillus*, although there were some exceptions (discussed further in Section 4.6). Although this may be expected, with *N. lapillus* being the highest trophic level of the three target species, metal tissue levels are related less to trophic level and more to specific position in a food chain (Wang, 2002). The colour of *N. lapillus* shells is related to their prey, with those preying predominantly on barnacles having light coloured shells (Hyman, 1967, cited in O'Leary and Breen, 1997). The predominance of light coloured specimens indicates that the Aberthaw *N. lapillus* population preys predominantly on barnacles; this is supported by the absence of any other potential prey items on the shore in Limpert Bay. It has been reported that barnacles are particularly efficient accumulators of metals (Rainbow *et al*., 1990, Rainbow and Wang, 2001); consequently the relatively high metal levels in *N. lapillus* reported here are likely to be linked to their predation on barnacle populations. Generally, the concentrations of metals recorded in the present study are consistent with those reported elsewhere, with the exception of mercury at EO where levels in 2009 and between 2011 and 2017 were considerably higher than other reported values (Appendix E; Table E.1). It has been shown that levels of metal concentrations in the tissue of predatory gastropod molluscs tend to be greater than those found in their prey (e.g. Blackmore, 2000). Consequently body burdens in *N. lapillus* in the present study should give an indication of how levels are integrated over time and are biomagnified up the food chain.

The spatial pattern of mercury in *N. lapillus* between 2009 and 2017 indicated higher levels in individuals to the east of the discharge. Comparisons of *N. lapillus* with *P. vulgata* mercury concentrations over the monitoring period show a similar trend in increases and decreases in concentrations over the four sites. A trend of declining mercury concentrations from 2013-2017 was reported, although to levels still greater than pre-commissioning. According to data collected by RWE there is a similar pattern in relation to water temperature, which indicates a greater thermal influence to the east of the discharges and reflects the general extent of the plume. Metal uptake in *N. lapillus* has been demonstrated to increase with increasing temperature (Leung *et al*., 2000) (see also Section 4.8). However, if temperature was the factor driving the observed mercury distribution in *N. lapillus*, similar elevated patterns would be expected with other metals, although this was not the case. Consequently, it is considered that temperature is unlikely to be the main factor driving accumulation of mercury in *N. lapillus*. However, the observed spatial patterns would indicate some influence of the discharge on the uptake of mercury by *N. lapillus*.

In the 2022 survey sufficient quantities of *N. lapillus* were not found at site HO. At all other sites the levels of mercury were found to be below the limit of detection (LoD <1mg/kg). Taking the LoD as a theoretical maximum value for the samples at site EO this represents the lowest levels of mercury recorded since 2007. Mercury levels peaked at site EO in 2013 when the Aberthaw plant was at it's highest load factor (78%). Similarly the results at site LE show a peak value for 2013. The data from 2022 (<1mg/kg) shows mercury levels that are lower than in 2010 and 2012 to 2015. This would be consistent with lower levels of mercury in the environment following the closure of the plant at Aberthaw in 2020.

The closure of the Aberthaw plant has been responsible for a reduction in the concentration of mercury found in *N. lapillus* in the sites closest to the outfall.

4.4 Limpet (*Patella vulgata*)

The common limpet, *Patella vulgata*, is an important member of marine communities on exposed and moderately-exposed rocky shores. Being principally a microphageous grazer of diatoms and macroalgal spores, *P. vulgata* obtains metals predominantly via dietary sources. *P. vulgata* has also been reported as feeding on mature macroalgae when plants lie against the rocks during periods of emersion, which can result in total exclusion of macroalgae from limpet-occupied areas (Lorenzen, 2007). *P. vulgata* has an important role in the structuring of rocky shore communities and any changes in its population on the shore can result in appreciable modification of rocky shore community characteristics (Hill *et al*., 1998). As a primary consumer, metal levels in the flesh of *P. vulgata* give a good indication of the availability of metals at this level of the food chain in Limpert Bay. The increases in mercury concentrations in *P. vulgata* in 2009 indicates the likely influence of the increased levels of mercury being discharged as a consequence of the FGD, while the spatial patterns evident in subsequent years up to and including 2017, are clear evidence of the continued influence of the discharge. The consistent pattern of the reduction of mercury contamination with distance from the outfall mirrors the pattern evident in *N. lapillus*.

In 2022 all sites returned values that were below the lowest limit of detection (<1 mg/kg). Taking the LoD as a theoretical maximum value for the samples at site EO this represents the lowest levels of mercury recorded since 2008. Mercury levels peaked at site EO in 2009 a year after the FGD plant was commissioned and when the Aberthaw plant was at a high load factor. The results at site LE show a peak value for 2016. The data from 2022 (<1mg/kg) show mercury levels that are lower than in all previous years except for 2014. This would be consistent with lower levels of mercury in the environment following the closure of the plant at Aberthaw in 2020. While it is not possible to comment on the spatial distribution of mercury levels this does represent a decrease in mercury levels at sites EO and LE where previously the highest values were recorded.

The closure of the Aberthaw plant has been responsible for a reduction in the concentration of mercury found in *P. vulgata* in the sites closest to the outfall.

4.5 Serrated wrack (*Fucus serratus*)

As metals are absorbed by brown seaweeds by simple ion exchange across cell walls (Forsberg *et al*., 1988), concentrations of metals in seaweed are a good indication of concentrations in the surrounding water (Fuge and James, 1973). Mercury levels in fucoids have been shown to be proportional to average dissolved concentrations (Bryan and Gibbs, 1983). Cairrão *et al*., (2007) reported that where mercury levels in sediment and water were below international standards, levels in fucoids reached 0.3 mg/kg, which corresponds to levels reported throughout the present study with the exception of EO in 2009 (and to a lesser extent, 2011-2017). The post-commissioning pattern indicates some influence of the increased mercury content of the discharge, although it is unclear why the levels in fucoids were so elevated in 2009. However, in subsequent years, mercury concentrations in fucoids at EO remained an order of magnitude higher than levels recorded at sites further east and two orders of magnitude higher compared to those recorded to the west, indicating the continuing influence of the discharge. In 2022 all samples returned values for mercury below the lowest limits of detection (<1mg/kg).

Throughout the study period metal levels in *Fucus serratus* were appreciably higher than those which are considered as natural background concentrations for fucoids as reported by Riget *et al*. (1997). With the exception of mercury concentrations at EO in 2009 and between 2011 and 2017, most metal levels (with the exclusion of arsenic) were similar to those previously reported for fucoids from the south bank of the outer Severn Estuary and throughout Cardigan Bay (Appendix F; Table F.1).

In 2009 mercury levels in *F. serratus* at EO were an order of magnitude greater than those previously recorded in fucoid algae from this region, while concentrations recorded between 2011 and 2017 were also considerably higher than historical levels. Data for 2022 were all below the lowest level of detection (<1mg/kg).

Although some of these historical data refer to *Fucus vesiculosus*, rather than *F. serratus*, it has been reported that closely related species bio-accumulate metals to the same degree (Bryan *et al*., 1985; Barreiro *et al*., 2004). As the NRW data and pre-commissioning data reported here are of the same order of magnitude, the information can be considered as generally consistent. Consequently, it is considered that these historical data provide a suitable baseline against which post-commissioning data can be compared. When the data is paralleled with the historic data, the levels of mercury at EO (and to a smaller degree at sites LE and HO) were considerably higher than that of the baseline data. This is consistent with other findings within this report, i.e. that there is a likely effect from the outfall following FGD installation.

Since the closure of the site in March 2020 levels of mercury in the site are likely to have decreased. In the 2022 data all samples were below the lowest limit of detection (<1mg/kg) which represents a decrease in concentration for site EO. Site EO had previously had the highest concentrations of mercury in *F. serratus*. This would be consistent with lower levels of mercury being present in the environment following the closure of the plant at Aberthaw in 2020.

4.6 Methylmercury

Between 2010 and 2017 only the lower ends of the mean concentration ranges of methylmercury (MeHg) in both *N. lapillus* and *P. vulgata* were similar to concentrations reported elsewhere in comparative fauna of similar trophic levels. The levels at Aberthaw were generally higher and the upper limits reported have been increasing in *P. vulgata* each year (Appendix F; Table F.2). It should be noted that the comparative fauna include those from areas outside of the UK, and therefore they may be subject to different temperature regimes and other environmental variables, that may affect the concentrations of metals within their flesh. Data from 2022 was, for all sites and species, below the lowest limit of detection (<100 µg/kg). This represents a decrease in concentration for *N. lapillus* and *P. vulgata* at all sites and *F. serratus* for site EO, LE and HO.

The MeHg concentrations in both *N. lapillus* and *P. vulgata* have shown relatively consistent levels across the survey area. The proportion of total mercury represented by organic mercury (i.e. in MeHg) showed a clear spatial pattern in both species, with the lowest proportions (up to approximately 10%) occurring in the vicinity of the outfalls to the east in all years. At the more remote sites, proportions were similar to those recorded elsewhere in invertebrates of between approximately 15 and 70 % (see Ipolyi *et al*., 2004; Claisse *et al*., 2001; Di Leo *et al*., 2010). Mikac *et al*. (1987) reported that the proportion of organic mercury in mussels sampled from an area subject to high mercury contamination in the Adriatic Sea was lower than in animals taken from areas of low contamination. A similar pattern was observed by Coelho *et al*. (2008) in shore crabs from a coastal lagoon in Portugal contaminated by inorganic mercury. The study concluded that the major route by which animals are exposed to total mercury in uncontaminated areas was through their diet, while in contaminated areas environmental exposure was the primary uptake route.

At Aberthaw it is interesting that the concentrations of MeHg are higher in *P. vulgata*, a lower trophic level than *N. lapillus*, particularly as the levels in *F. serratus* (which may be grazed upon by *P. vulgata*) are very low. The reasons for this are unclear, and it is unlikely that *N. lapillus's* prey (e.g. barnacles) are accumulating MeHg at much lower rate than *P. vulgata* as their prey are filter feeders. Rainbow and Wang (2005) state that owing to factors such as high ingestion and assimilation rates, accumulated trace metal concentrations in barnacles are one of the highest in marine invertebrates. Furthermore the trend in the year on year increase in the upper limit of MeHg levels in *P. vulgata* is not mirrored in *N. lapillus*.

As the primary source of MeHg is dietary at Aberthaw, the consistent spatial pattern of MeHg levels in both species indicates that the MeHg source is relatively consistent over the survey area. However, as there is a negative relationship between the ratio of organic mercury and total mercury it is evident that the source of total mercury in both species varies over the survey area. Therefore, it is considered that the disparity between the levels of inorganic mercury present in the target species is clearly related to variability in bioavailability across the survey area. Hence, it is suggested that the greater levels of inorganic mercury that are available to the east of the outfalls are potentially a consequence of the discharge itself.

Sediments at the sites sampled are likely to be subject to continual re-suspension and, as a consequence, will be characterised by aerobic conditions thus reducing the activity of sulphate reducing bacteria which are prevalent in anaerobic conditions and primarily responsible for methylation of mercury (Compeau and Bartha, 1985). Consequently, the levels of sediment-bound MeHg in the present study do not indicate the influence of gross inputs of mercury; these inputs will have dropped to zero since the closure of the plant in March 2020 and mercury levels (and thus MeHg levels) will be representative of levels in sediments in the wider environment. It is possible that natural environmental conditions may result in low conversion rates of inorganic mercury into MeHg, with the 2017 mean proportion of total mercury represented by organic mercury (from MeHg) at Aberthaw of 0.68%, below what is considered as a typical contribution to total concentration in bottom sediments (i.e. between 1 and 1.5%) (Ullrich *et al*., 2001, cited in Boszke *et al*., 2003). This is a strong indication that the sediments sampled are more representative of suspended sediments and not bottom sediments.

4.7 Mercury in biota

It has been stated in Section 4.3 that different species accumulate metals at different rates (e.g. Bryan *et al*., 1985; Ostapczuk *et al*., 1997). Hence, it is not surprising that the highest metal concentrations reported here tended to occur in *N. lapillus* and the lowest generally in the seaweed *F. serratus*; a pattern probably related to the trophic level of the species (i.e. bioaccumulation).

As outlined earlier, there were some spatial differences in the levels of metals found within each of the three target species, although there was no consistent pattern and, with the exception of mercury, any differences observed were relatively small. The spatial and temporal patterns observed in the present study indicate that mercury levels in biota are related to the concentration of available metal, which, in the context of the present study area was potentially influenced by the FGD discharge. Since the closure of the plant in March 2020 the influence of the discharge will progressively have waned.

It has been demonstrated that some species can develop physiological traits to eliminate or detoxify elevated levels of metals in dietary sources (Rainbow *et al*., 1999, cited in Rainbow *et al*., 2009). This can lead to increased tolerances to metal exposure, leading to an equilibrium between metal inputs and bioaccumulation which may result in a diminution of the initial post-commissioning patterns observed. However, the rate at which any such equilibrium is reached and becomes evident at the population level is unclear. Boisson *et al*. (1998) discussed the importance of cellular, genetic and biochemical levels of adaptation in relation to biota dealing with increased exposure to metals. It has also been demonstrated that these traits can be inherited in subsequent generations (see Grant *et al*., 1989). However, Paterson *et al*. (2006) suggested that changes in mercury levels in biota following an increase in mercury loading to the aquatic environment may take at least 10 to 30 years to reach a steady state. Consequently, although post-commissioning spatial patterns in mercury levels in all three target species indicated an influence of the discharge, the temporal variability in the levels at sites directly to the east of the outfalls may have indicated that any adaptation of biota to increased mercury availability may still be ongoing at Aberthaw.

The revised Priority Substances Directive set an EQS for mercury concentration in fish tissue of 0.02 mg/kg wet weight, although it states that the EQS may be applied to other taxa as long as an equivalent level of protection is provided. It should be noted that the species studied here may not be adopted for statutory monitoring purposes; consequently, the following discussions should be considered as indicative rather than specific.

Converting results from the present study for *N. lapillus* using a shell-free dry weight (SFDW) to shell-free wet weight (SFWW) conversion ratio of 0.243 (derived from data for the related species *Buccinum undatum* given by Rumohr *et al*., 1987), resulted in mercury levels exceeding the value associated with the EQS at all sites in all years, i.e. both pre- and post-commissioning and at all distances from the discharge. The greatest exceedance was observed at EO in 2014 (Appendix E; Table E.3). Between 2010 and 2017 the highest values were recorded in the vicinity of and to the east of the CW discharges. Mercury levels in *P. vulgata* were converted using an average SFDW:SFWW (shell-free dry weight:shell-free wet weight) ratio obtained for species of gastropod (not specifically Patella) taken from Rumohr *et al*. (1987). All *P. vulgata* wet weight mercury concentrations were above the value associated with the EQS, with the greatest value recorded at EO.

It should be noted that the revised Priority Substances Directive allows for the designation of a mixing zone, where these standards can be exceeded with the proviso that conditions of the water body out with the mixing zone complies with the Directive standards. Although all of the data values presented here are greater than the value associated with the EQS, the pattern of reduction in mercury levels with distance from the outfalls to pre–commissioning levels at the remote sites indicates that the influence of the FGD discharge in relation to exceedances of the value associated with the EQS is spatially restricted. These data can be compared with average mercury levels in the harbour ragworm *Hediste diversicolor* from the Severn Estuary/upper Bristol Channel recorded by NRW of 0.10 mg/kg and in the edible mussel *Mytilus edulis* from Cardiff of 0.04 mg/kg (NRW unpublished data). This indicates that mercury levels reported in the biota at Aberthaw show a similar relationship to the stated EQS, as concentrations in other invertebrate species from the same region.

The 2009 data indicated a post-commissioning increase in availability of mercury to biota at two of the three sites sampled, although some spatial variability was evident with the greatest increases evident to the east of the outfalls at EO. As sites EO and WO are equidistant from the outfall it would be expected that any effects associated with changes to the discharge would be comparable in biota at each site. However, it is evident that increased levels of mercury in the biota collected at EO were proportionally higher than at site WO. Between 2010 and 2017 mercury levels in biota fell with distance from the discharges although concentrations were higher to the east of the discharge. Such patterns may be related to the orientation of the discharge. At both outfalls the discharge is from the eastern side of the caissons which, in the case of the eastern outfall results in the discharge flowing directly over the sampling area of EO for the majority of the tidal cycle, particularly on a rising tide. Water discharged from the western outfall will initially flow away from the sampling area of WO at any state of the tide, and is only likely to impinge directly on the sampling area on a falling tide between high and mid water; at all other states of the tide the influence of the discharge at WO is likely to be low. Furthermore, above mid water on a rising tide, cooling water from the western outfall will also pass over the sampling area at EO. These observations are supported by water temperature monitoring carried out by RWE.

Consequently, any changes to the discharge are likely to be experienced to an appreciably greater extent by biota at EO compared with elsewhere. Therefore, it would appear that the distribution of mercury in all three target species between 2009 and 2022 is related to changes in the discharge, increased mercury loadings post-commissioning and reduction in mercury loadings post-closure. Despite elevated body burdens of mercury, the field observations of sustained populations remaining along the same shore each year, the abundances of which allow for harvesting for analysis, indicate no observable effects from operation of the power station (both alone in terms of metals, temperature increases and pH, and in combination with natural stressors) at the population or community level. Furthermore, it was found that at EO, both *N. lapillus* and *P. vulgata* had a better condition (or health) at site EO compared to the other sites, indicated by the conditional indices results for 2017. Data for 2022 (2 years post closure of the plant) showed that both *N. lapillus* and *P. vulgata* had a better condition (or health) at site LW with site EO having a decreased condition index for both species. In the 2016 Jacobs pH variation assessment Aberthaw report (Jacobs, 2016a) it was hypothesised that the higher temperatures at the outfall acted as a buffer to increased stresses over winter, leading to higher conditional indices at site EO. However, caution must be taken with this analysis as there is a lack of data and therefore this information is not conclusive and just for reference.

The power station load factor remained consistent in the three years following commissioning, with the level of mercury being released remaining relatively stable. However, the increased load factor recorded between 2012 and 2014 may have influenced the level of mercury being discharged. It is also possible that changes in the fuel used at the station may have led to some variability in emissions. Although it appears that the post-commissioning variability in mercury levels in the target species is related to biotic factors, the post-commissioning increase and distribution of mercury levels observed in the biota indicate that changes owing to natural variability or other sources are unlikely and that the power station discharge was a major factor. At the closure of the power station discharges ended and it ceased to be a major factor in the study site.

4.8 Impacts on biota

The potential for impacts on biota as a result of changes in metal concentrations of the type inferred is discussed below. Gray (1979) states that biological impacts can be separated into two categories: disturbance, i.e. where taxa are physically destroyed or removed from the area (lethal effects); and stress, i.e. where the productivity of an individual is reduced (sub-lethal effects). These two factors can be seen in the time-related sequence of effects listed by Blackstock (1984) where the initial response of an organism to a pollutant is detection by sensory receptors, followed by behavioural and metabolic reactions. Mobile fauna may simply migrate away from the affected area, while the response of sessile, sediment dwelling animals unable to use the escape response, may be hormone controlled metabolic changes designed to aid their survival. In some instances metabolic equilibrium may be restored, showing acclimatisation, or individuals may be genetically selected to survive in the conditions, indicating adaptation (these can be considered as sub-lethal effects). Conversely, the impact may cause serious impairment of normal functions, and subsequently death, leading to changes in populations and community structure (i.e. lethal effects).

Adaptive traits developed by invertebrates for survival in contaminated areas may involve physiological processes such as the storage of metals in insoluble forms within distinct bioaccumulation structures or in inert chemical forms which pose no toxic threat to the host (Rainbow *et al*., 2009; Geffard *et al*., 2004). However, such material may be bioavailable to any potential predators which can result in possible toxic effects and/or the transfer and magnification of contaminants up the food chain. Where taxa are exposed to temporary increases of a contaminant, accumulation may be observed followed by a fall to normal levels when contaminant concentrations return to normal (Clark, 1999). Although such processes may allow for increased accumulation of metals within the biota, there may be a physiological price to pay. For instance, it has been demonstrated that increased contaminant concentrations and the subsequent bioaccumulation of materials can lead to reduced growth (Widdows *et al*., 1995).

In a review of the effects of temperature and metal pollution on aquatic organisms, Sokolova and Lannig (2008) stated that increasing temperature can have a positive effect on the rate of accumulation of mercury and MeHg, while the elimination of metals is generally not thought to be influenced by rising temperature. The review also discusses the positive correlation between increased temperature and toxicity of trace metals to aquatic invertebrates. It is further discussed that an increase in the accumulation rate of and sensitivity to metals of organisms in relation to elevated temperature may significantly impact the integrity of populations and potentially alter the transfer rates of metals through the food chain. Such factors can influence the success of a population and also have implications for potential predators in relation to food availability and exposure to contaminants.

In the populations of the target species investigated at Aberthaw metal levels recorded were analogous with those reported elsewhere in the region in areas remote from thermal discharges, therefore it is unlikely that the discharge was increasing uptake of metals in biota. However, as uptake of other metals is not affected by the thermal influence of the discharge it is considered that uptake of mercury was similarly unaffected in this manner and that any such changes were related to increased mercury loading associated with post-commissioning changes to the discharge.

4.9 Trophic transfer to humans

Commission Regulation (EC) no 466/2001[1] set maximum levels for mercury in edible fish of 0.5 mg/kg wet weight (1.0 mg/kg for European sea bass (*Dicentrarchus labrax*)) which corresponds to 4.9 mg/kg dry weight. Mercury levels in both *N. lapillus* and *P. vulgata* were below this level for all sites aside from EO, where the level was exceeded in most years from 2009 onwards. As this limit refers to fish for human consumption, the high mercury levels recorded here raise concern in relation to the possibility of transfer of mercury up the food chain, ultimately into species directly consumed by humans. The United States Food and Drug Administration set an action level (i.e. the level deemed unfit for human consumption) for MeHg of 1 mg/kg wet weight which corresponds to 10.2 mg/kg dry weight for gastropod molluscs (using conversion factor wet weight to dry weight conversion factor for gastropod molluscs given by Rumohr *et al*., 1987). All MeHg wet weight concentrations were considerably less than this limit with no single value exceeding 0.1 mg/kg. Blackmore and Wang (2004) stated that MeHg has a high potential for trophic transfer in the intertidal rocky shore food chain, while Kehrig *et al*., (2002) demonstrated an order of magnitude increase in MeHg levels between mussels and carnivorous fish. Consequently, although MeHg concentrations reported in the vicinity of the Aberthaw discharge are low, the rate of biomagnification to higher trophic levels could be potentially significant. This and the implications for possible transfer of material into the human food chain are discussed below.

As material is passed upwards between each trophic level, persistent contaminants are biomagnified and it is commonly accepted that longer food chains result in greater levels of biomagnification (Rasmussen *et al*., 1990). The upper trophic level in the marine food chain is generally represented by fish species which only metabolise mercury compounds slowly and tend to accumulate mercury in proteins, particularly in muscle tissue. Mercury is primarily stored in fish tissue as MeHg which in turn can be readily accumulated by humans who can only eliminate it from the body at a relatively slow rate (Harris and Snodgrass, 1993). Consequently, increased availability of mercury through the marine food chain in the vicinity of Aberthaw could represent a potential hazard for human consumption of fish caught in the area.

Limpert Bay is a popular site for anglers with sea bass a particular target species. Bass are known to be attracted to warm water outfalls and the vicinity of the Aberthaw power station outfall was recognised and designated as a nursery ground for this species. As bass are known to feed on small fish and invertebrates the patterns of contamination in the target species at Aberthaw provide an indication of levels of mercury available to fish feeding in the vicinity of the discharges. A survey was undertaken in 2013 to make a preliminary assessment of metals in bass muscle tissue in relation to the Aberthaw power station. Although there were no clear patterns in relation to most of the metal concentrations and the size of bass, mercury concentrations were positively correlated to fish size. Mercury concentrations in all of the bass sampled were in excess of the EQS for biota and levels in all but one individual were considered to be above Background Reference Conditions (Jacobs, 2013). It has been reported that at increased temperatures the rate of MeHg accumulation is elevated in fish consuming MeHg contaminated prey while growth rate is supressed (Dijkstra *et al*., 2013). Such a pattern could potentially exacerbate any risk in relation to human consumption of contaminated fish. However, when comparing the levels of mercury and MeHg recorded in the bass muscle tissue from Aberthaw to the maximum levels set for edible fish for human consumption, it was considered that levels recorded did not present a risk to human health.

Following the closure of the Aberthaw plant in 2020 mercury input to the environment will have ceased. Data collected in 2022 has shown a decrease in mercury and MeHg levels in biota in the area. Consequently it is likely that mercury and MeHg levels will also have decreased in fish found in the area.

[1] COMMISSION REGULATION (EC) No 466/2001of 8 March 2001setting maximum levels for certain contaminants in foodstuffs.

4.10 Summary

- Spatial variability in metal concentrations was evident for all three target species for all metals, although, with the exception of mercury, levels were considered to be generally consistent between 2007 and 2022.
- With the exception of mercury, pre- and post-commissioning metal concentrations recorded in all taxa were comparable with those recorded previously from elsewhere in the Bristol Channel and Severn Estuary.
- Between 2009 and 2017 mercury levels were generally higher in all three target species in the vicinity of the discharges, particularly at EO. Following closure of the plant, mercury levels in 2022 were below 1mg/kg for all species at all sites. At site EO this represents a substantial decrease in mercury levels.
- Between 2010 and 2017 the pattern of mercury distribution in biota showed a decrease with distance from the discharge, although concentrations to the east of the survey area remained higher than those to the west. The elevated levels of mercury recorded in biota to the immediate east of the outfall were considered to be related to the post-commissioning increase in levels of mercury in the discharge. This was not observed in 2022 (following closure of the plant) as mercury levels were below 1mg/kg for all species at all sites. This represents a decrease compared with values recorded in previous years.
- The data indicate that the total mercury in the discharge influences the mercury levels in the mollusc species sampled. Since the plant closed in 2020 and discharge ceased mercury levels in the mollusc species have reduced.
- The elevated levels of mercury in biota above the EQS are comparable with levels in other biota elsewhere in the wider region indicating the relatively low impact the FGD discharge had on the waterbody as a whole. Since the closure of the plant the mercury levels in biota at Aberthaw have decreased.
- In 2022 mean MeHg concentrations were below the lowest limit of detection for all species and all sites. This represents a decrease in concentration for *N. lapillus* and *P. vulgata* at all sites and *F. serratus* for site EO, LE and HO.
- Throughout the monitoring period sediment bound metal concentrations have stayed broadly similar to sediments found elsewhere in the lower Bristol Channel. Data from the 2022 survey was consistent with this.
- It is considered that sediment-bound metal concentrations are unlikely to be having any significant detrimental effects on the marine ecology in the vicinity of station.

5. Conclusions

This report seeks to make the case that the terms of Aberthaw Power Station Permit number EPR/RP3133LD improvement programme requirement IC21 have been met and that this requirement may now be closed out.

With the exception of mercury, post-commissioning metal concentrations in the tissue of all three target species at Aberthaw are consistent with pre-commissioning levels and data from other studies and are not considered as particularly elevated. Post-commissioning mercury concentrations in all three taxa at EO indicated an appreciable increase in levels and were considerably higher than those in similar species reported from the Bristol Channel, Severn Estuary and other locations. Following closure of the power station all discharges from the FGD plant have now ceased and the 2022 results have shown a decrease in mercury levels in all biota at EO, therefore showing a reversal of effects from FGD operation.

Pre- and post-commissioning sediment conditions are consistent with other areas in the Severn Estuary and Bristol Channel and, although some levels of metals were elevated above background conditions, they showed no specific influence of the discharge from Aberthaw and the levels reported are not considered to be harmful to sediment-dwelling benthos. During the monitoring period there have been variations in sediment type and metal concentrations which are consistent with highly mobile sediments likely to be found in an environment like the Bristol Channel with extensive tidal flow. Variations observed in metal concentrations are therefore likely to result from wider anthropogenic factors, rather than resulting from power station operation.

It should be noted that the EQS value set for mercury in biota by the revised Priority Substances Directive was exceeded in *N. lapillus* and *P. vulgata* in pre- and post-commissioning populations. While the 2022 survey still showed values in excess of the EQS, values had decreased since the previous survey in 2017.

There is only likely to be a low risk of transfer of mercury up the food chain due to the limited extent of elevated mercury levels in biota allied to the mobile nature of predator species which will spend limited time feeding in the vicinity of Aberthaw. Consequently, it is unlikely that the FGD discharge represented a significant source of mercury for transfer and magnification up the food chain. Since the closure of the power station and the ending of discharges the risk of transfer of mercury up the food chain has substantially decreased.

Whilst there are limitations with the data collected during the 2022 study (higher limits of detection), the data does show that there is a reversal in levels of metals and methylmercury at the outfall site (EO). This therefore demonstrates a reversal in the effects of the FGD and supports cessation of the monitoring at this site.

6. References

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Appendix A. Metal concentrations in biota, 2022

Given as mg/kg dry weight (ug/Kg for methylmercury).

Table A-1. Dogwhelk (*Nucella lapillus*)

	LE1	EO1	EO2	EO3	LW1	LW2	LW3
Arsenic	14	15	10	14	11	13	15
Cadmium	8.6	12	8.4	14	11	14	16
Chromium	<1	<1	1.1	<1	<1	<1	<1
Copper	42	55	52	51	61	79	82
Mercury	<1	<1	<1	<1	<1	<1	<1
Nickel	<1	<1	<1	<1	<1	<1	<1
Lead	<1	<1	<1	<1	<1	<1	<1
Zinc	174	202	197	180	207	192	278
Methylmercury	<100	<100	<100	<100	<100	<100	<100

Table A-2. Limpet (*Patella vulgata*).

	HO1	HO2	HO3	LE1	LE2	LE3	EO1	EO2	EO3	LW1	LW2	LW3
Arsenic	4.9	4.3	5.5	3.8	4.1	6.2	5.4	5.3	6.1	6.1	5.1	5.4
Cadmium	11	6.8	9.3	2.2	6.1	4.6	12	12	12	8.5	8.3	13
Chromium	1.6	<1	<1	1.4	1.4	1.5	5.3	1.2	5.1	1.2	2.2	1.5
Copper	3.3	14	2.7	4.0	3.8	2.2	3.5	2.8	4.6	9.7	16	3.1
Mercury	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Nickel	1.0	<1	<1	1.1	1.1	1.1	2.3	<1	2.2	<1	1.5	1.3
Lead	<1	1.4	<1	1.1	1.3	1.1	<1	<1	<1	<1	1.0	<1
Zinc	41	34	44	31	25	26	46	45	40	47	35	34
Methylmercury	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100

Table A-3. Serrated wrack (*Fucus serratus*)

	HO1	HO2	HO3	LE1	LE2	LE3	EO1	EO2	EO3	LW1	LW2	LW3
Arsenic	8.6	9.1	7.3	5.9	9.2	7.3	9.3	6.7	8.4	8.8	8.8	8.9
Cadmium	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Chromium	4.7	2.8	1.7	2.9	4.4	2.2	<1	2.9	1.9	1.4	1.3	1.1
Copper	4.3	11	5.6	1.7	2.2	3.6	1.5	1.1	2.0	2.0	1.8	3.0
Mercury	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Nickel	4.0	3.5	2.1	2.6	4.4	3.5	2.5	2.6	2.1	2.5	2.0	2.5
Lead	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1
Zinc	67	57	52	45	59	54	53	38	41	49	44	53
Methylmercury	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100

Appendix B. Standardised metal concentrations in biota

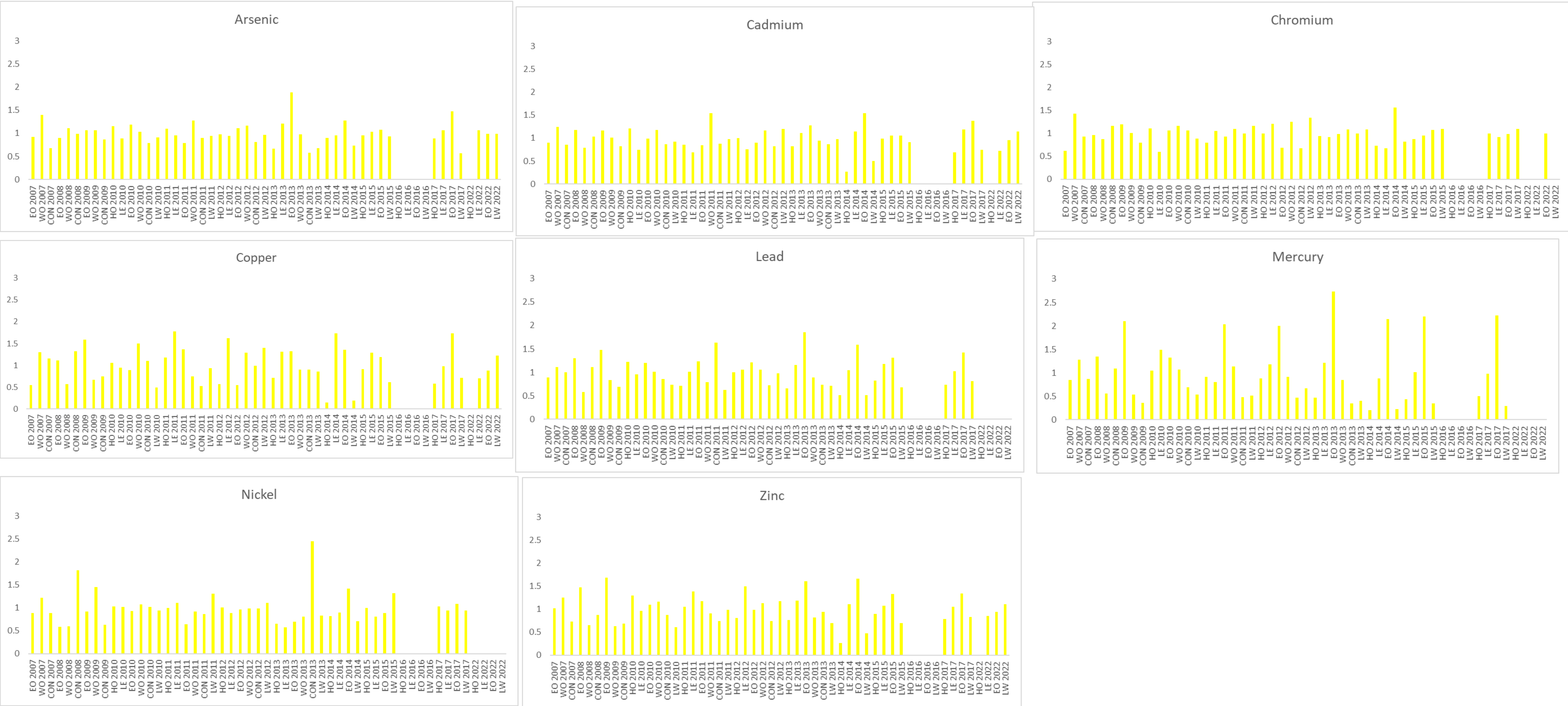

Figure B-1. : Standardised metal concentrations (SM) in the tissue of the dogwhelk *Nucella lapillus* at each site in each year (sites CON and WO sampled 2007-2013 only, data not available for 2016).

Figure B-2. Standardised metal concentrations in the tissue of the limpet *Patella vulgata* at each site in each year. (Sites CON and WO sampled 2007-2013 only).

Figure B-3. : Standardised metal concentrations in the tissue of the seaweed *Fucus serratus* at each site in each year. (Sites CON and WO sampled 2007-2013 only).

Appendix C. Mean metal concentrations, averaged 2010-2022

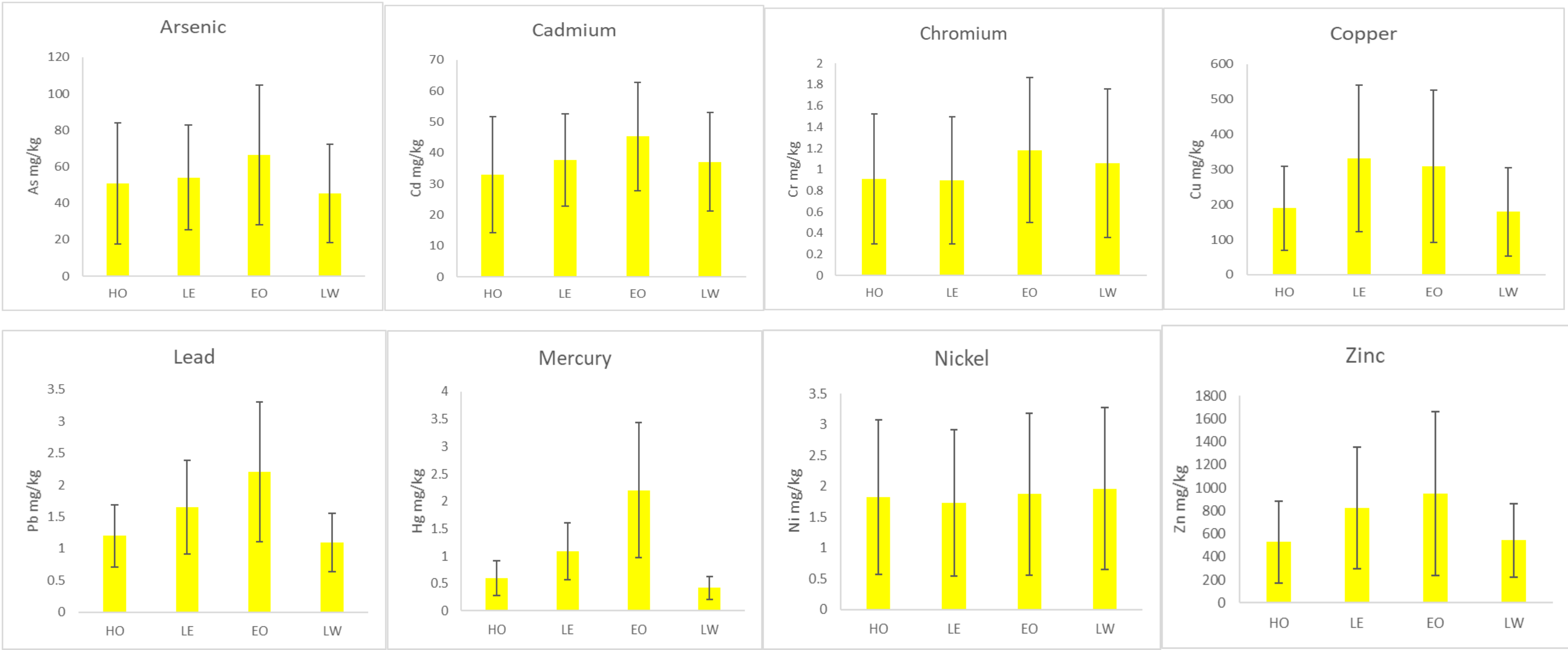

Figure C-1. Mean metal concentrations (mg/kg dry weight) in the tissue of the dogwhelk *Nucella lapillus* averaged between 2010 and 2022 at sites sampled in 2022. Data not available for 2016. Error bars represent standard deviation.

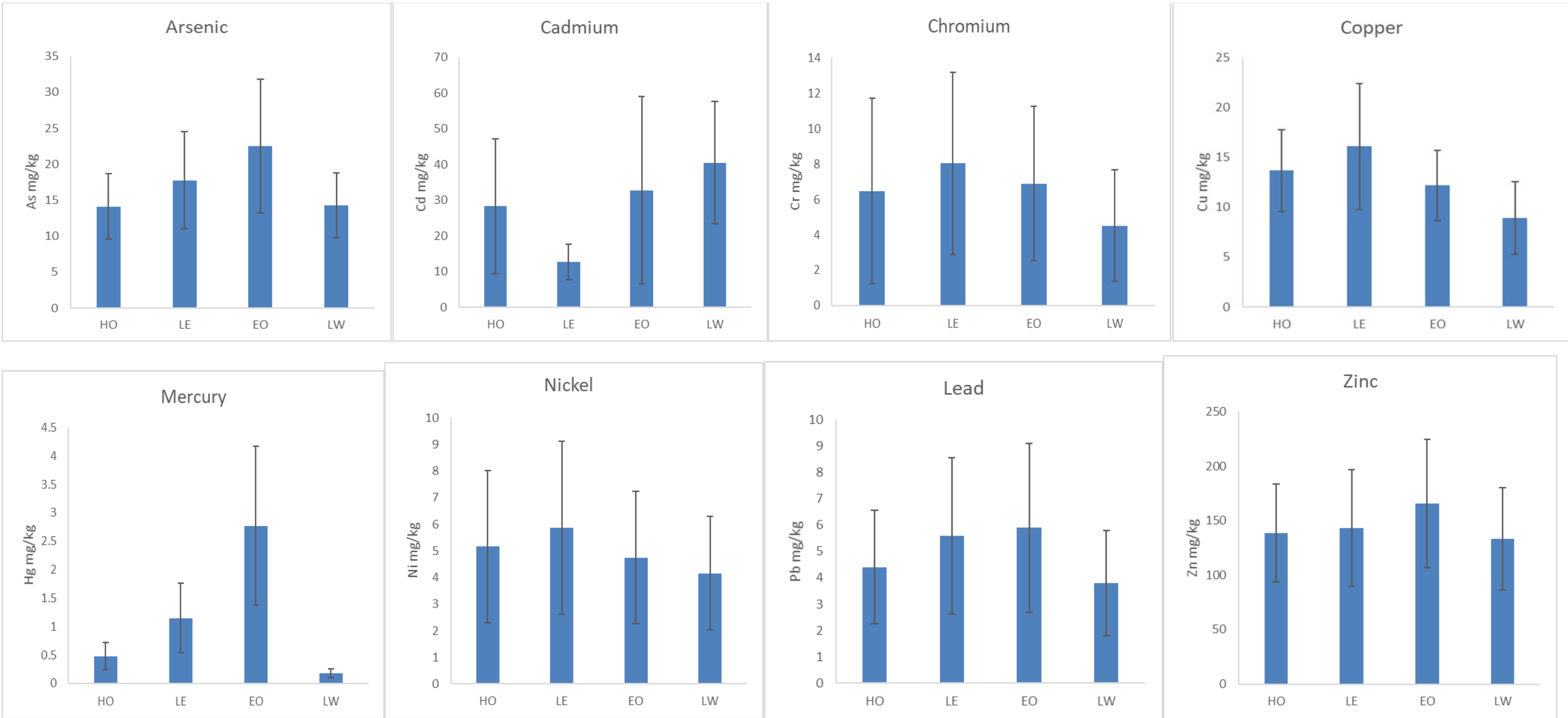

Figure C-2. Mean metal concentrations (mg/kg dry weight) in the tissue of the limpet *Patella vulgata* averaged between 2010 and 2017 at sites sampled in 2017. Error bars represent standard deviation.

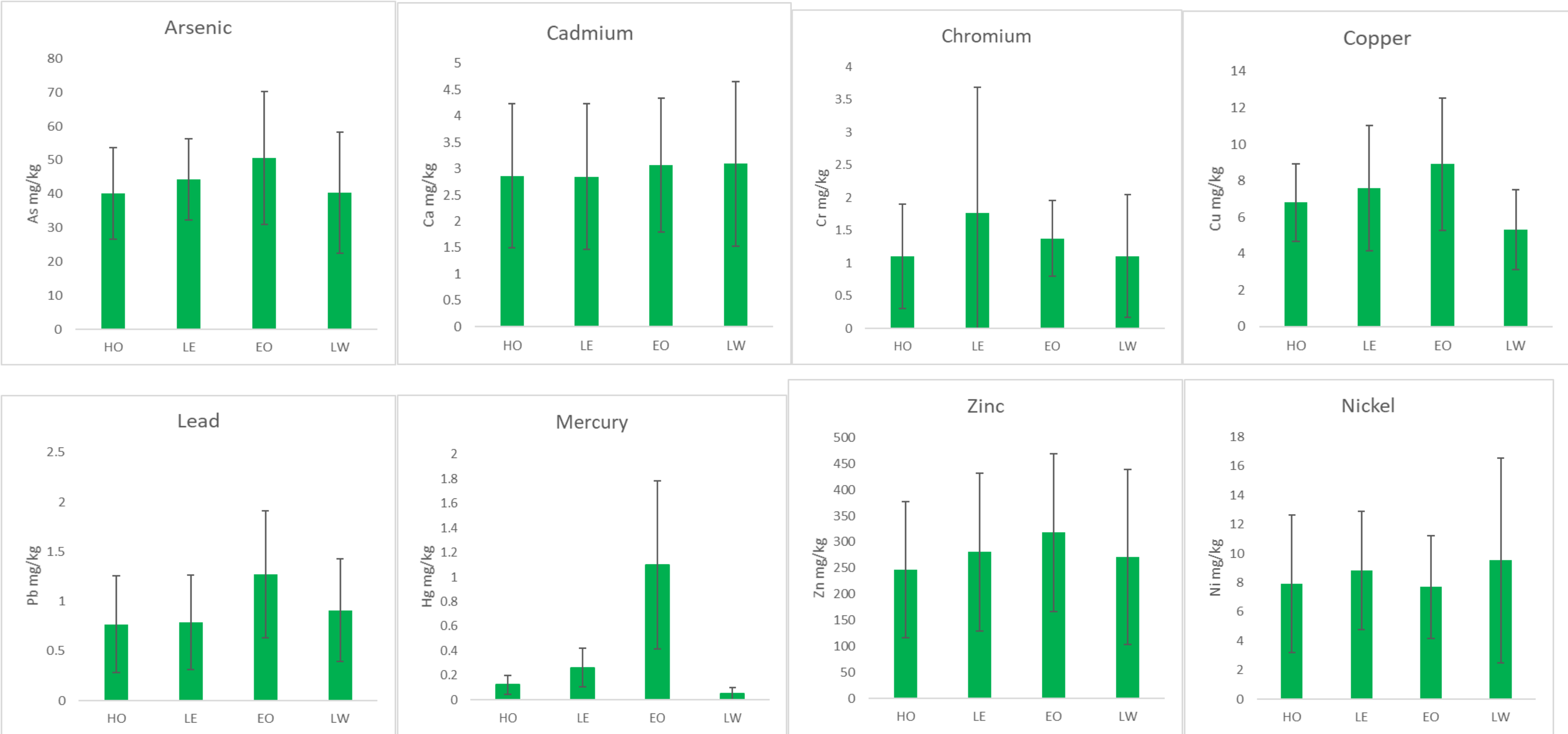

Figure C-3. Mean metal concentrations (mg/kg dry weight) in the tissue of the seaweed *Fucus serratus* averaged between 2010 and 2022 at sites sampled in 2022. Error bars represent standard deviation.

Appendix D. Sediment data, 2022

Table D-1. Granulometric data (Grain size fractions given as a percentage).

	Replicate A	Replicate B	Replicate C	Replicate D	Replicate E
Grain Size Fraction (µm)					
<62.5	99.7	99.7	80.6	97.3	95.7
63 - 2000	0.26	0.28	19.4	2.75	4.35
>2000	0	0	0	0	0

Table D-2. Sediment-bound metal concentrations (mg/kg dry weight)

	Replicate A	Replicate B	Replicate C	Replicate D	Replicate E
Copper	22.4	23.3	17.7	22	21.5
Cadmium	0.2	0.1	0.1	0.2	0.1
Zinc	164.5	158.4	136.6	143	148.8
Mercury	0.17	0.18	0.14	0.18	0.16
Lead	47	48.6	33.6	43.7	42.5
Arsenic	13.4	13.4	12	12.5	12.2
Chromium	34.6	36.2	24.9	31.3	31
Nickel	27.7	28.8	21.3	26	26.2
Aluminium	19900	20100	13100	18000	18000

Appendix E. Methylmercury concentrations, 2022

Table E-1. Methylmercury concentrations (mg/kg wet weight) in Dogwhelk (*Nucella lapillus*)

	HO	LE	EO	LW
Replicate 1	<100	<100	<100	<100
Replicate 2	<100	<100	<100	<100
Replicate 3	<100	<100	<100	<100

Table E-2. Methylmercury concentrations (mg/kg wet weight) in limpet (*Patella vulgata*)

	HO	LE	EO	LW
Replicate 1	<100	<100	<100	<100
Replicate 2	<100	<100	<100	<100
Replicate 3	<100	<100	<100	<100

Table E-3. Methylmercury concentrations (mg/kg wet weight) in serrated wrack (*Fucus serratus*)

	HO	LE	EO	LW
Replicate 1	<100	<100	<100	<100
Replicate 2	<100	<100	<100	<100
Replicate 3	<100	<100	<100	<100

Table E-4. Methylmercury concentrations (µg/kg dry weight) in Sediments

Replicate A	Replicate B	Replicate C	Replicate D	Replicate E
<100	<100	<100	<100	<100

Appendix F. Comparison with previous studies

Table F-1. Mean metal concentrations (mg/kg) in biota and sediment at Aberthaw (all sites) and reported values from other studies. Shading indicates FGD post-commissioning data from Aberthaw.

Location	Species	Cu	Zn	Cd	Hg	Pb	As	Cr	Ni	Reference
Barry	*F. vesiculosus*	14.3	209	15.8	-	-	-	-	26.2	Fuge & James (1974)
Sand Point	*F. serratus*	16.3	740	22.7	-	7.33	-	-	51.2	Martin *et al.* (1997)
Cardigan Bay	*F. serratus*	3.7	175	2.9		-	12.1	-	-	Fuge & James (1973)
Severn Estuary	*F. vesiculosus*	8.4	73.5	1.6	0.01	1.64	14.4	1.74	5.18	EA unpublished data
Greenland	*F. vesiculosus*	2.1	7.2	2.1	-	0.26	0.3	0.6	-	Riget *et al.*(1997)
Aberthaw	*F. serratus*	6.1	258.9	3.7	0.08	1.09	37.2	1.47	9.57	Present Study 2007/08
Aberthaw	*F. serratus*	6.1	226.3	2.9	1.03	0.76	31.7	1.83	7.9	Present Study 2009
Aberthaw	*F. serratus*	3.9	117.2	2.2	0.06	0.57	46.9	1.3	2.55	Present Study 2010
Aberthaw	*F. serratus*	5.7	226.4	3.1	0.39	0.74	47.6	1.54	5.83	Present Study 2011
Aberthaw	*F. serratus*	6.8	248.4	3.3	0.33	0.7	45.5	0.51	7.54	Present Study 2012
Aberthaw	*F. serratus*	7.1	347.6	3.6	0.42	0.89	61	1.16	8.72	Present Study 2013
Aberthaw	*F. serratus*	9.6	396.5	3.4	0.38	1.54	41.5	0.89	13.54	Present Study 2014
Aberthaw	*F. serratus*	9.1	341.1	3.8	0.64	1.63	46.8	1.41	9.34	Present Study 2015
Aberthaw	*F. serratus*	10.2	459.3	4.2	0.32	1.06	54	1	13.96	Present Study 2016
Aberthaw	*F. serratus*	7.5	312.3	3.1	0.45	1.25	46	0.95	10.42	Present Study 2017
Shannon Estuary,	*N. lapillus*	44.5	213.8	-	-	-	-	5.11	-	O'Leary and Breen
Barents Sea	*N. lapillus*	66	553	24	-	1.9	-	-	2.3	Zauke *et al.* (2003)
Weston-Super-Mare	*N. lapillus*	114	1836	114	-	19	-	1.38	3.4	Bryan *et al.* (1985)
Aberavon	*N. lapillus*	93	667	47	0.33	18	-	11	-	Portman (1979)b
Amroth	*N. lapillus*	77	263	32	0.13	8.7	-	6	-	Portman (1979)b
Aberthaw	*N. lapillus*	222	848	79	0.8	2.15	87.6	3.08	2.47	Present Study 2007/08
Aberthaw	*N. lapillus*	257	978.4	53	1.89	1.35	86.7	0.62	2.21	Present Study 2009
Aberthaw	*N. lapillus*	113	543.4	44.9	0.83	1.26	68.5	0.75	3.93	Present Study 2010
Aberthaw	*N. lapillus*	179	724.8	45.6	1.05	1.67	77.9	1	1.49	Present Study 2011
Aberthaw	*N. lapillus*	146	830.1	49.9	1.03	1.49	83	0.86	1.39	Present Study 2012
Aberthaw	*N. lapillus*	542	1512.9	58	1.56	2.09	50.7	1.53	2.6	Present Study 2013

Location	Species	Cu	Zn	Cd	Hg	Pb	As	Cr	Ni	Reference
Aberthaw	*N. lapillus*	250	948.3	36.2	1.38	1.88	96.2	1.4	2.33	Present Study 2014
Aberthaw	*N. lapillus*	374	735	41.3	1.28	2.05	44	1.27	3.07	Present Study 2015
Aberthaw	*N. lapillus*	253	665.1	29.4	0.89	1.71	53.6	2.08	2.67	Present Study 2017
Shannon Estuary, Eire	*P. vulgata*	5.6	87	-	-	-	-	-	0.03	O'Leary and Breen (1997)
Portishead	*P. vulgata*	10.8	117	116.9	-	-	-	-	-	Noël-Lambot, *et al.* (1980)
Portishead	*P. vulgata*	35	312	289	-	6.2	-	1.66	1.07	Bryan *et al.* (1985)
Weston-Super-Mare	*P. vulgata*	41	279	239	0.12	10.3	15	3.59	4.5	Bryan *et al.* (1980)
Looe Estuary	*P. vulgata*	18	145	5.6	0.26	30	33	0.5	2.3	Bryan *et al.* (1980)
Aberthaw	*P. vulgata*	17.7	144	39.7	0.34	5.7	17	34.3	26.1	Present Study 2007/08
Aberthaw	*P. vulgata*	16.7	161.7	31.8	2.51	4.64	19	9.14	6.62	Present Study 2009
Aberthaw	*P. vulgata*	17.2	152.9	31.1	1.22	6.63	16	12.5	8.01	Present Study 2010
Aberthaw	*P. vulgata*	15.4	147.4	30.2	1.07	4.59	18	9.41	5.83	Present Study 2011
Aberthaw	*P. vulgata*	14.1	130.3	34	1.25	4.43	19	4.74	3.79	Present Study 2012
Aberthaw	*P. vulgata*	15	174.3	48	1.34	7.05	19	9.06	6.09	Present Study 2013
Aberthaw	*P. vulgata*	7.2	128.4	28.2	0.51	1.85	10	1.86	2.06	Present Study 2014
Aberthaw	*P. vulgata*	16.1	150.7	45.7	1.14	5.49	20	4.96	4.31	Present Study 2015
Aberthaw	*P. vulgata*	17.8	191.6	25.6	1.41	7.08	24	7.88	6.4	Present Study 2016
Aberthaw	*P. vulgata*	19.7	157.8	15.9	1.03	6.24	15	6.66	6.9	Present Study 2017
Severn Estuary	*Sediment*	43	215	71	0.4	84	38	15	0.24	EA unpublished data
Milford Haven	*Sediment*	19	126	38	0.23	56	23	11	0.12	NMMP site 647 data
Lower Thames Estuary	*Sediment*	48	120	55	0.36	96	31	13	0.35	NMMP site 455 data
Avon (Severn)	*Sediment*	39	287	0.92	0.55	104	8.6	-	-	Langston *et al.*(2003)
Usk	*Sediment*	53	288	0.86	0.41	93	9.2	-	-	Langston *et al.*(2003)
Weston-Super-Mare	*Sediment*	33	252	0.7	0.42	88	7.8	41.4	33.3	Bryan *et al.* (1980)
Aberthaw	*Sediment*	69	182	0.29	0.01	60	16	91	42	Present Study 2007/08
Aberthaw	*Sediment*	6.4	51.3	0.09	0.02	15.2	8.8	18	7.06	Present Study 2009
Aberthaw	*Sediment*	31.9	205.6	0.18	0.16	69.4	18	84.6	35.7	Present Study 2010
Aberthaw	*Sediment*	23.1	79.6	0.12	0.05	25.7	7.1	56.2	22.5	Present Study 2011

Location	Species	Cu	Zn	Cd	Hg	Pb	As	Cr	Ni	Reference
Aberthaw	*Sediment*	24.6	128.4	0.11	0.1	37.7	11	85.5	39.3	Present Study 2012
Aberthaw	*Sediment*	33.9	176.4	0.12	0.13	55.5	17	99.6	49	Present Study 2013
Aberthaw	*Sediment*	28	179.2	0.16	0.17	56.4	19	81.4	36.4	Present Study 2014
Aberthaw	*Sediment*	20.5	136	0.17	0.07	37.6	12	66.7	28.8	Present Study 2015
Aberthaw	*Sediment*	16.4	-	0.18	0.1	35.7	13	67.3	23.3	Present Study 2016
Aberthaw	*Sediment*	11.6	81.4	0.13	0.16	31.2	11	38.8	18	Present Study 2017

Table F-2. Methylmercury (MeHg) concentrations (µg/kg) in biota (wet weight) and sediment (dry weight) at Aberthaw, and reported values from other studies. *cited in Trefry *et al*. (2002)

	Biota/Location	Concentration µg/kg	Reference
Biota	*Hediste diversicolor*	3,8	Muhaya *et al.* (1997)
	Ruditapes philippinarum	15 - 38	Trombini *et al.* (2003)
	Mytilidae	17 - 116	Ipoyi *et al.* (2004)
	Mytilidae	48 - 223	Kehrig *et al.* (2002)
	Mytilidae	3.4 - 7.1	Claisse *et al.* (2001)
	Mytilidae	5.4 - 12.5	Di Leo *et al.* (2010)
	Nucella lapillus	15-59	Present study 2010
	Nucella lapillus	17 - 70	Present study 2011
	Nucella lapillus	12 - 72	Present study 2012
	Nucella lapillus	11 - 78	Present study 2013
	Nucella lapillus	11 - 47	Present study 2014
	Nucella lapillus	18 - 51	Present study 2015
	Nucella lapillus	19 - 56	Present study 2016
	Nucella lapillus	18 - 46	Present study 2017
	Patella vulgata	15 - 45	Present study 2010
	Patella vulgata	21 - 73	Present study 2011
	Patella vulgata	9 - 72	Present study 2012
	Patella vulgata	10 - 72	Present study 2013
	Patella vulgata	14 - 80	Present study 2014
	Patella vulgata	35 - 91	Present study 2015
	Patella vulgata	59 - 100	Present study 2016
	Patella vulgata	61 - 101	Present study 2017
	Scheldt Estuary, Belgium	0.8 - 6	Muhaya *et al.* (1997)

Sediment	Pialassa Baiona, Italy	0.13 - 45	Trombini *et al.* (2003)
	Bohai Sea, China	35	Wang *et al.* (2009)
	Lavaca Bay, USA	783	Bloom *et al.* (1999)*
	Aberthaw	3 -12	Present study 2010
	Aberthaw	1 - 7	Present study 2011
	Aberthaw	<1- 9	Present study 2012
	Aberthaw	<1- 9	Present study 2013
	Aberthaw	<1 - 4	Present study 2014
	Aberthaw	<1 - 2	Present study 2015
	Aberthaw	<1 - 2	Present study 2016
	Aberthaw	<1 - 2	Present study 2017

Table E-3. Mercury levels in *Nucella lapillus* and *Patella vulgata* expressed as mg/kg wet weight. EQS for mercury set as 0.02 mg/kg wet weight (note this value is for fish tissue). n-d= no data.

Nucella lapillus												
	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2022
LW	-	-	-	0.108	0.130	0.168	0.152	0.087	0.101	-	0.063909	<1
EO	0.145	0.296	0.966	0.267	0.520	0.501	1.038	1.074	0.641	-	0.48195	<1
LE	-	-	-	0.300	0.205	0.296	0.460	0.344	0.295	-	0.212139	<1
HO	-	-	-	0.212	0.232	0.222	0.176	0.079	0.208	-	0.086865	-
Patella vulgata												
	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2022
LW	-	-	-	0.056	0.059	0.050	0.052	0.023	0.042	0.045	0.072	<1
EO	0.055	0.231	1.552	0.980	0.666	1.099	1.153	0.365	0.648	0.755	0.527	<1
LE	-	-	-	0.365	0.310	0.287	0.461	0.067	0.315	0.476	0.291	<1
HO	-	-	-	0.219	0.167	0.129	0.099	0.061	0.131	0.130	0.136	<1