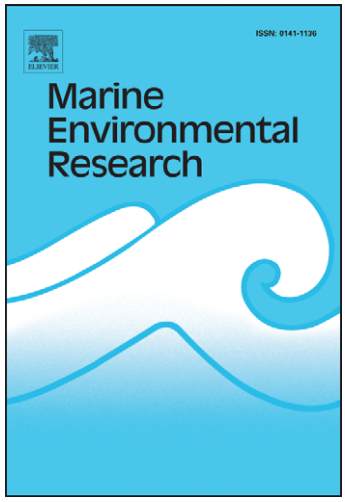
Hepatic biomarkers of sediment-associated pollution in juvenile turbot, L

Mark G. J. Hartl, Michael Kilemade, David Sheehan, Carmel Mothersill, John O'Halloran, Nora M. O'Brien, Frank N.A. M van Pelt



Hepatic biomarkers of sediment-associated pollution in juvenile turbot, *Scophthalmus maximus* L.

Mark G. J. Hartl, Michael Kilemade, David Sheehan, Carmel Mothersill, John O'Halloran, Nora M. O'Brien, Frank N. A. M van Pelt

PII: S0141-1136(07)00016-5
DOI: 10.1016/j.marenvres.2007.01.002
Reference: MERE 3095

To appear in: *Marine Environmental Research*

Received Date: 12 May 2006
Revised Date: 20 December 2006
Accepted Date: 5 January 2007

Please cite this article as: J. Hartl, M.G., Kilemade, M., Sheehan, D., Mothersill, C., O'Halloran, J., O'Brien, N.M., A. M van Pelt, F.N., Hepatic biomarkers of sediment-associated pollution in juvenile turbot, *Scophthalmus maximus* L., *Marine Environmental Research* (2007), doi: 10.1016/j.marenvres.2007.01.002

Hepatic biomarkers of sediment-associated pollution in juvenile turbot, *Scophthalmus maximus* L.

Running title: Ecotoxicology of contaminated estuarine sediment

Mark G. J. Hartl[a, b, §], Michael Kilemade[a, b], David Sheehan[a, c], Carmel Mothersill[a, d], John O'Halloran[a, b], Nora M. O'Brien[a, f] and Frank N. A. M van Pelt[a, e *]

[a] Environmental Research Institute, University College Cork, Ireland

[b] Department of Zoology, Ecology & Plant Science, University College Cork, Ireland

[c] Department of Biochemistry, University College Cork, Ireland

[d] Radiation and Environmental Science Centre, Dublin Institute of Technology, Dublin, Ireland

[e] Department of Pharmacology & Therapeutics, University College Cork, Ireland

[f] Department of Food and Nutritional Sciences, University College Cork

[§] Present address: School of Life Sciences, Heriot-Watt University, Edinburgh EH14 4AS, UK

* Corresponding author; Fax: +353 21 4343211. e-mail : f.vanpelt@ucc.ie

Abstract

Hatchery-reared turbot (*Scophthalmus maximus* L.) were exposed for three weeks, under laboratory conditions, to sediment collected from polluted sites in Cork Harbour and a reference site at Ballymacoda, Co. Cork, Ireland. The potential of surficial sediment for inducing hepatic biomarkers was assessed at two levels of biological organisation: expression of cytochrome P450 [Western blotting analysis and 7-Ethoxy-Resorufin O-Dealkylase (EROD), 7-Benzoxy Resorufin O-Dealkylase (BROD), 7-Methoxy Resorufin O-Dealkylase (MROD), 7-Pentoxy-Resorufin O-Dealkylase (PROD) activities] and DNA integrity (Comet assay). Positive controls were generated, either by exposing turbot to cadmium chloride spiked seawater (Comet assay) or to ß-naphthaflavone by intra-peritoneal injection (cytochrome P450 induction).

The induction of cytochrome P450 activity (EROD, MROD and PROD) in animals following a 7-day exposure to contaminated sediments was significantly higher than those exposed to reference site sediment and remained elevated thereafter; BROD was not induced. DNA single-strand breaks were also significantly higher following exposure to contaminated sediments throughout the experiment. Although no direct correlation between induction of alkoxyresorufin O-dealkylase activities and a particular chemical class was established, the induction of MROD and PROD activities in fish exposed to sediments containing complex contaminant mixtures, appeared to be more sensitive than conventional EROD activity assays.

We conclude from the present laboratory study that *S. maximus* is a suitable sentinel species for the assessment of moderately contaminated sediments and therefore allows for the further development of this model for future, ecologically relevant, field studies.

Key words: sediment; cytochrome P450; Western blotting; Comet assay; Pleuronectiformes

1. Introduction

Post-metamorphic Pleuronectiformes, such as turbot, maintain a close association with sediments for food and cover and are therefore more likely to be exposed to sediment-associated organic pollutants, including polychlorinated biphenyls (PCBs) (Courtney & Langston, 1980), organotins (Hartl *et al.*, 2001)and polycyclic aromatic hydrocarbons (PAHs) (Besselink *et al.*, 1998). Cytochrome P450-dependent monooxygenases play an important role in the synthesis and degradation of many biological molecules and the transformation of potentially carcinogenic foreign compounds (Stegeman, 1989).

Laboratory studies with collected sediment and harbour sludge have revealed induction of cytochrome P4501A1 in various benthic fish species, such as European flounder (Eggens *et al.*, 1996; Besselink *et al.*, 1998) greenback flounder (Mondon *et al.*, 2001) and plaice (Eggens *et al.*, 1996). In addition *in situ* caging experiments with European flounder (Lindstrom-Seppa *et al.*, 1992; Eggens *et al.*, 1995; Beyer *et al.*, 1996; Eggens *et al.*, 1996; Besselink *et al.*, 1998), plaice (Eggens *et al.*, 1995; Eggens *et al.*, 1996), cod (Husoy *et al.*, 1996) and dab (Sleiderink *et al.*, 1995) have demonstrated that cytochrome P4501A1 induction is an environmentally relevant biomarker.

PAHs have been shown to bind to DNA in fish, causing DNA instability and potentially pre-mutagenic damage (Aas *et al.*, 2001; Myers *et al.*, 2003). However, PAHs require enzymatic bioactivation in order to produce potentially damaging DNA adducts (Stegeman, 1981). The alkaline single-cell gel electrophoresis assay (Comet assay) can determine such pre-mutagenic DNA damage in the form of single-strand breaks (Singh *et al.*, 1988). Accordingly, the Comet assay has been applied to a range of fish species using aqueous exposure regimes, both *in vitro* (Devaux *et al.*, 1997;

Mitchelmore & Chipman, 1998a; Frenzilli *et al.*, 1999) and *in vivo* (Belpaeme *et al.*, 1996; Belpaeme *et al.*, 1998). Kammann *et al.* (2000) used the Comet assay to examine the genotoxicity of sediment extracts to *Cyprinus carpio* leukocytes *in vitro* and Nacci *et al.* (1996) determined DNA damage in flounders exposed to sediment spiked with benzo(a)pyrene. In turbot, a wide variety of substrates have been found to induce cytochrome P450-containing monooxygenases activity and cause DNA damage, whereby the route of exposure has either been intraperitoneal injection (Peters & Livingstone, 1995) or aqueous suspension (Peters *et al.*, 1997; Camus *et al.*, 1998).

Following metamorphosis, turbot display a predominantly benthic life style and maintain intimate contact with sediments, where they seek shelter, waylay their prey (Aarnio *et al.*, 1996; Beyst *et al.*, 1999), and are therefore likely to be exposed to sediment-associated contaminants. This is of particular interest, because although similar data from field experiments using other pleuronectiforme species exist (Rice *et al.*, 1994; Eggens *et al.*, 1995; Myers *et al.*, 2003), there is no comparable data available for turbot, *S. maximus*. Hatchery production of turbot has lead to an increased availability of individuals with a known exposure history, which is an essential element in any toxicological study (Boisson *et al.*, 1998).

The purpose of this study was to assess the effect of sediment exposure in hatchery-reared *S. maximus* under controlled laboratory conditions by means of two complimentary biomarkers on two levels of biological organisation, biochemical (cytochrome P450 induction) and genetic (DNA instability), and to evaluate the potential of this species as a model organism for the detection of sediment-associated pollutants in the environment.

2. Material & Methods

2.1. Site characteristics

For this study, two contaminated sites Whitegate and Aghada, in Cork Harbour and a reference site, Ballymacoda Estuary, outside of Cork Harbour, were chosen (Fig. 1). Previous investigations, including our own, have shown that, although sediments from Ballymacoda Estuary do contain metals, they are comparatively free of organic contaminants, whereas those from Cork Harbour are more heavily contaminated with trace metals and PAHs (Boelens *et al.*, 1999; Byrne & O'Halloran, 1999; Kilemade *et al.*, 2004a) Table 1 summarises the chemical analysis of the sediment sampled for the present study.

2.2. Collection of sediment

Approximately 100kg of surface sediment (the top oxygenated 0.5-1 cm) were collected in June 2002, using a plastic trowel at low tide, mixed thoroughly and transported to the laboratory, where sub-samples were immediately frozen for chemical analysis.

2.3. Sediment analyses and characterisation

The <63µm fraction of the sediment sub-samples were analysed by ERGO Unweltinstitut GmbH, Dresden, Germany (DIN EN 45001 and DIN EN ISO 9002-accredited) for PAH analysis (Soxhlet extraction, GC-MS detection) and by RIVO, Netherlands Institute for Fisheries Research, The Netherlands (EN/ISO 17025, STERLAB accredited) for brominated flame retardants (BFR), organotin (Soxhlet extraction, GC-MS detection), PCB and organochlorine pesticide (OCP) (Soxhlet extraction, GC-ECD detection). The heavy metals copper, lead, cadmium and zinc were analysed by atomic absorption spectrometry, following acid digestion by Mercury Analytical Ltd., Limerick, Ireland, who conform to ISO 9002.

2.4. Fish husbandry

Juvenile turbot (0+; average weight 15.4 ± 2.77 g) were obtained from a hatchery (Turbard Iarthar Chonamara Teo) in County Galway, Ireland. The fish were acclimatised to aerated seawater (16°C; salinity 35; pH 8; 74 % O_2; NO_3^-: <4 mg l^{-1}; NO_2^-: 0.25 mg l^{-1}) in 500 litre polyethylene fish-farming tanks (stocking density: 100 per tank) at the Aquaculture & Fisheries Development Centre in Cork for at least two weeks. The water was changed every three days and the fish were fed *ad libitum* on commercially available turbot pellets during the acclimation phase, but starved during the experiment.

2.5. Experimental design

The sediment from each site was divided evenly between two 500 l seawater-seasoned fish farming tanks (2 tanks for each site, 6 tanks in total) to form a layer covering the floor approximately 10 cm thick. The tanks were then filled with seawater and the sediment was left to settle overnight. 30 acclimated turbot were added to each tank. Four turbot were sampled from the stock population at t_0 (without exposure to the sediment) and then a further 4 per tank in 7-day intervals ($t_{7\text{-days}}$, $t_{14\text{-days}}$, $t_{21\text{-days}}$) for three weeks thereafter. The fish were sacrificed by an overdose of CO_2, followed by the destruction of the brain. Half the fish (two) from each tank were used for the P450 analysis and the other half for the Comet assay. All handling during tissue dissection, dissociation, and preparations were performed on ice.

2.6. Cytochrome P450 analysis – sample preparation

The livers were removed, weighed, immediately shock frozen in liquid nitrogen and stored at –80°C for later analysis. Individual livers were thawed on ice, homogenised in ice-cold Trizma buffer (50mM, pH 7.7) and the resulting homogenate centrifuged at 12,500xg (4°C). Total protein was measured according to Bradford (1976) .

2.6.1. *Cytochrome P450 analysis - protein detection*

A Western blot analysis was performed to detect expression of the cytochrome P4501A1 protein. Samples were diluted in water and mixed with an equal volume of sample buffer [8% Sodium dodecyl sulphate (SDS) (w/v), 20% (v/v) glycerol, 0.002% (w/v) bromophenol blue, 0.125 M Tris (pH 6.8) and dithiothreitol (6 mg ml^{-1})] to give a final protein concentration of 1.3 mg ml^{-1} and boiled for three minutes. SDS-PolyAcrylamide Gel Electrophoresis (PAGE) was carried out using a mini vertical electrophoresis unit (Sigma) with a 5% stacking gel/ 10% resolving gel. Samples were loaded (40 µg $well^{-1}$) and run at 15 mA/minigel at 4°C. The resolved proteins were transferred onto nitrocellulose at 4°C using a transfer buffer containing 15.7 mM Tris, 120 mM Glycine (pH 8.3) and 20% (v/v) methanol for 1 hour at 200V using a mini-wet blotter (Sigma). The nitrocellulose membrane was incubated overnight at 4°C in blocking buffer [Phosphate Buffered Saline (PBS) containing 5% (w/v) non-fat milk powder] with gentle agitation. The membrane was then incubated with 1° antibody diluted in blocking buffer (1:3,000 polyclonal rabbit-anti-fish P4501A1, CP-226 Biosense, Norway) for 3 hours at room temperature with gentle agitation. Unbound antiserum was removed and the membrane was washed three times for 10 minutes each in PBS containing 0.02% (w/v) Tween 20. Following this, the membrane was incubated with the 2° antibody diluted in blocking buffer (1:2000 anti-rabbit IgG HRP-linked antibody and 1:1000 Anti-biotin HRP-linked antibody #7075, Cell Signalling) for two hours at room temperature with gentle agitation. The blots were washed again as before. Bound antibodies were visualised by enhanced chemiluminescent detection using Pierce SuperSignal® Substrate (Pierce, USA) according to manufacturers instructions. Control samples were generated by intra-peritoneal injection of either 40 mg kg^{-1} β-naphthflavone (BNF, positive control) in

Dimethyl Sulfoxide (DMSO) or DMSO (a vehicle control). Livers were sampled 96 hours after exposure and processed as described above.

2.6.2. *Cytochrome P450 analysis - enzyme activities*

Ten µl of supernatant containing the postmitochondrial fraction were added to the wells of a black fluorometric plate containing 200 µl of NAPDH (0.25 mg ml^{-1}) and incubated at 37°C for 10 minutes. The reaction was then started by adding 1 µl of the substrates ethoxyresorufin (for EROD), benzyloxyresorufin (for BROD), methoxyresorufin (for MROD) or pentoxyresorufin (for PROD) (250µM in DMSO) and the fluorescence measured in a micro plate reader (Tecan) for 20 minutes (ex: 535nm; em: 590nm) according to Burke & Meyer (1974). *ß*-Napthaflavone, a know inducer of Cytochrome P450 activity in turbot (Arukwe & Goksoyr, 1997) was used as a positive control. The activity was expressed as pmol resorufin mg $protein^{-1}$ $minute^{-1}$.

2.7. Comet assay

Following dissection, livers for the Comet assay were prepared according to (Kilemade *et al.*, 2004b). The livers were immediately washed 3 times with phosphate buffered saline (PBS), gently minced with two fresh scalpel blades, transferred to fresh tubes, incubated in 10 ml 0.25 % trypsin-EDTA and placed on a rotating disc at room temperature for 10 minutes. The enzymatic digestion was halted by the addition of an equal volume of foetal calf serum to each tube. The resultant cell suspensions were subsequently decanted into fresh tubes, leaving behind the larger undigested tissue pieces and centrifuged at 800x*g* for 10 minutes in a cooled bench-top micro-centrifuge. After centrifugation, the supernatants were decanted, and the cell pellets resuspended in 10 ml fetal calf serum. Following this procedure, the cell viability was determined by the fluorescein diacetate/ethidium bromide (F.Da/Et.Br.) assay of

(Anderson *et al.*, 1994). The Comet assay was performed according to Coughlan *et al.* (2002) , adapted from Woods *et al.* (1999) . Briefly, cells were sandwiched in 1% low-melting agarose onto frosted microscope slides, lysed in a high salt buffer [2.5M NaCl, 10mM Tris, 100mM EDTA, 1% (v/v) Triton X-100 and 10% (v/v) DMSO, pH 10.0, in the dark at 4°C] for 90 minutes, immersed in an alkaline solution (0.3 M NaOH, 1 mM EDTA; pH 13) for 30 min at 4°C in order for the DNA to unwind and a current (25V, 300 mA) applied for 25 minutes in a horizontal electrophoresis tank. The pH was neutralized with Tris buffer (0.4 M Tris-HCl, pH 7.4), the DNA stained with ethidium bromide and the nucleoids analysed using an epifluorescence microscope (Nikon EFD-3). DNA damage was determined using the imaging analysis software package Komet 4.0. (Kinetic Imaging Ltd). DNA damage was expressed as percentage tail DNA (% tDNA), which is defined as the percentage DNA that has migrated from the head (Hartmann *et al.*, 2003).

In order to substantiate the Comet assay, juvenile turbot were exposed to seawater spiked with cadmium chloride ($CdCl_2$), an agent known to cause oxidative DNA damage (Risso-de Faverney *et al.*, 2001; Valverde *et al.*, 2001). 50 l tanks of seawater were spiked with 4 and 40 µM $CdCl_2$ respectively. A control tank consisted of seawater only. Following a 40h exposure fish were sacrificed, and liver single cell preparations were carried out as detailed above.

2.8. Data and Statistical analysis

Comet assay: 400-500 nuclei per treatment were analysed, i.e. 4-5 fish per duplicate treatment tank – 1 slide per fish - 4-5 slides per duplicate treatment tank – 8-10 slides per treatment - 50 nuclei per slide.

Cytochrome P450: livers from 2 fish per duplicate treatment tank.

The data from each group (time and site) were pooled and analysed using a non-parametric Kruskal-Wallis ANOVA on ranks followed by a Dunn's pair-wise multiple comparison test. A $p < 0.05$ was considered significant (Fry, 1993).

3. Results

3.1. Chemical analysis of sediment

The sediments used were analysed for a range of organic compounds and metals (Table 1). The concentrations of some pollutants, especially total polycyclic aromatic hydrocarbons (ΣPAHs) and to a lesser extent heavy metals, were substantially higher in the polluted sites, Aghada and Whitegate, compared to the reference site at Ballymacoda. All other compounds analysed were at or below the limit of detection at all sites.

3.2. Cytochrome P450 induction

3.2.1. Cytochrome P450 protein detection

The Western blot analysis showed a strong immune response for cytochrome P4501A1 in lanes BNF (positive control $t_{4\text{-days}}$), A (Aghada, $t_{21\text{-days}}$) and W (Whitegate, $t_{21\text{-days}}$), whereas no protein recognition by the antibody was observed in lanes C (control $t_{21\text{-days}}$), t_0 (pre-exposure) and B (Ballymacoda, $t_{21\text{-days}}$) (Fig. 2). This agreed with the EROD activities for both positive control (BNF) and exposure experiment (Figs. 3 & 4). No cross-reactivity with other proteins was observed with the antibody on the Western blots.

3.2.2. Cytochrome P450 activity - Positive control

A significant up-regulation of EROD (cytochrome P4501A1) activity was observed in turbot 96h post-intraperitonial injection with 40 mg kg^{-1} Beta-Naphthoflavone (BNF; Fig. 3).

3.2.3. Cytochrome P450 enzyme activities-exposure experiments

The average baseline rate of EROD, BROD, MROD and PROD activity in juvenile turbot at t_0, prior to sediment exposure, were 25.88 ± 15.6, 0.6 ± 1.02, 2.07 ± 0.91 and 1.09 ± 2.0 pmol mg protein^{-1} min^{-1}, respectively (Fig. 4). Following 7 days of exposure ($t_{7\text{-days}}$) to sediments from the reference site at Ballymacoda, the EROD activity increased significantly above background levels (t_0). However, this increase was substantially lower than the increase in EROD activity observed in fish exposed to sediment from the contaminated sites at Aghada and Whitegate. At $t_{14\text{-days}}$ and $t_{21\text{-days}}$, the EROD activity in turbot exposed to sediment from the reference site dropped, but remained significantly above background levels, whilst those from both contaminated sites remained elevated (Fig. 4).

MROD activities of fish exposed to contaminated sediments from Whitegate and Aghada increased significantly above the baseline rates at t_0 (Fig. 4). With the exception of $t_{21\text{-days}}$, MROD activity in fish exposed to sediment from Ballymacoda did not change and were consistently significantly lower than activities of contaminant-exposed fish.

The patterns of PROD activity for fish exposed to sediment from the reference site and the contaminated sediments from Aghada, were similar to those for MROD – a significant increase in PROD activity throughout the experiment for Aghada and Whitegate, but no change for Ballymacoda. Although the PROD activity from Whitegate was higher than Aghada at $t_{21\text{-days}}$, there was no statistically significant difference between the two. BROD activity did not change throughout the experiment.

3.6. DNA integrity

3.6.1. Positive control

A Comet assay performed on liver cells from turbot exposed for 40h to 0, 4 and 40 µmol $CdCl_2$ in aqueous suspension, showed a clear dose-related genotoxic response (Fig. 5).

3.6.2. Comet assay

The cell viability following preparation of single-cell suspensions from turbot liver averaged 73 %. The average background % tail DNA at t_0, before exposure to sediment, was 7.79 % ± 0.35 (Fig. 6). Following 7 days of exposure ($t_{7\text{-days}}$) to sediments from the polluted sites at Aghada and Whitegate, the % tail DNA values increased significantly and were also significantly higher than those from the reference site at Ballymacoda (Fig. 6). This pattern continued largely unchanged throughout the remainder of the three-week experiment.

4. Discussion

In the present study the examination of cytochrome-P450 monooxygenase induction and DNA single-strand breaks were applied, in combination, to assess the *in vivo* effects of exposure to sediment-associated contaminants on the liver of juvenile turbot. The availability of hatchery-reared turbot from a single cohort with known exposure history allowed the determination of baseline levels for these complimentary biomarkers, with the aim of developing this species as a potential model organism for the contamination status of natural sediments in the field.

4.1. Exposure conditions

The predominantly benthic behaviour of turbot, together with turbulence-induced agitation of the sediment surface, facilitates the re-introduction of sediment-associated contaminants to the sediment-water interface and the likely exposure of wild turbot to

potentially harmful metals and organic compounds (Anderson *et al.*, 1987; Long *et al.*, 1998; Schiff & Allen, 2000). Although turbot were exposed under controlled laboratory conditions, experiments were designed on the basis of these observations in order to simulate field conditions as closely as possible. For this reason, only the thin oxygenated surface layer of sediment (1-2 cm), the layer with which turbot are most likely to interact with, was collected and used in the present study.

4.2. Cytochrome P450 induction

A variety of PAHs and PCBs introduced to the aquatic environment are known to induce cytochrome P450 in turbot (Peters & Livingstone, 1995; Peters *et al.*, 1997; Boleas *et al.*, 1998; Aas *et al.*, 2000). Baseline cytochrome P450 activity may vary considerably and depend on a variety of factors, such as age, sex, sexual maturity, season, exposure history and diet. This is reflected in the baseline EROD activity of juvenile turbot (15 g) measured in the present study (25 ± 15 pmol min^{-1} mg $protein^{-1}$), compared with the previously reported values of 10.8 ± 2 pmol min^{-1} mg $protein^{-1}$ for 90 day old juveniles, 12.3 ± 4 pmol min^{-1} mg $protein^{-1}$ for sexually mature adults (Peters & Liviingstone, 1995).

In this study the 21-day exposure of juvenile turbot to polluted sediments showed EROD activities significantly above baseline values and also above those from fish exposed to sediments from a reference site (Fig. 4). Significant EROD induction was observed after 7 days and the activity remained elevated thereafter. Although there are no comparable data available for turbot, exposure experiments of other benthic fish species to contaminated whole sediments, in both field or laboratory studies, have yielded similar results: *Limanda limanda*, wild catch field study (Sleiderink *et al.*, 1995); *Platichthys flesus*, whole sediment mesocosm study (Besselink *et al.*, 1998); *P. flesus* and *Gadus morhua*, wild catch and hatchery-reared caged field studies (Beyer

et al., 1996; Husoy *et al.*, 1996); *P. flesus* and *Pleuronectes platessa*, wild catch and caged field studies (Eggens *et al.*, 1995; Eggens *et al.*, 1996); *Rhombosolea tapirina*, hatcery-reared whole sediment laboratory study (Mondon *et al.*, 2001); *Cottus cognatus*, wild catch whole sediment laboratory study (Tetreault *et al.*, 2003).

A variety of PAHs, planar PCBs, chlorinated dioxins and furans are known inducers of cytochrome P4501A1 (Stegeman & Hahn, 1994).

In complex chemical mixtures, such as those found in the polluted sediments from Aghada and Whitegate, any cytochrome P450 upregulation is likely to be the net result of additive, synergistic or antagonistic chemical interactions. Although the precise nature of these interactions was beyond the scope of the present study, the prevalence of Σ PAHs (Table 1) in the sediments samples suggests that PAHs were probably mainly responsible for the observed increase in EROD activity.

Although background levels of EROD activity were detected in t_0 and reference site samples (Fig. 4), on the corresponding Western blot there is no band for t_0 and only a very faint band for the reference site (Fig. 2). This apparent discrepancy is due to the fact that protein expression in samples from Aghada and Whitegate, as well as the BNF samples, was very high and the antibody dilutions and the exposure time during film development were optimised accordingly. The MROD activity also showed a significant increase in turbot exposed to sediment from Cork harbour (Fig. 4). In contrast to mammals, knowledge of multiple cytochrome P450 enzymes in fish is limited and controversial. It has been suggested that unlike mammals, where there is a clear distinction between cytochrome P4501A1 and cytochrome P4501A2, there is only one cytochrome P4501A enzyme in fish (Stegeman, 1989). Smeets *et al.* (2002) reported MROD activity in *P. flesus, L. limanda*, *Oncorhynchus mykiss* and *Microstomus kitt*, but concluded from strong correlations between EROD and MROD

activities, that these were linked, suggesting, that there was only one cytochrome P4501A enzyme involved. In contrast cytochrome P4501A2-like induction has been reported from a variety of fish species, such as *P. platessa* (Leaver *et al.*, 1988), *Pagrus major* (Mizukami *et al.*, 1994), *P. flesus* and *Anguilla Anguilla* (Rotchell *et al.*, 2000). The correlation between EROD and MROD activities from turbot in the present study (r = 0.81) and reports by (Celander & Forlin, 1992) and Berndtson & Chen (1994) of two cytochrome P4501A enzymes in *O. mykiss* suggest that the induction of cytochrome P4501A2 in fish may indeed be species specific.

PROD activity was also significantly increased in turbot exposed to Cork Harbour sediment (Fig. 4). PROD in mammals is associated with cytochrome P4502B. PROD activity has been detected in several marine and freshwater fish species exposed to water-borne pollutants, including *Cyprinus carpio* (Machala *et al.*, 1997), *O. mykiss* (Lindstrom-Seppa *et al.*, 1992; Haasch *et al.*, 1994), and immunological detection of P4502B has been demonstrated for *Stenotomus chrysops* (Stegeman, 1989). To our knowledge this is the first time induction of PROD activity has been recorded in fish exposed to sediments containing complex mixtures of contaminants. Typical PROD inducers are barbiturates, non-planar PCBs and DDT. Failure by Yawetz *et al.* (1998a) to detect cytochrome P4502B induction in *Mugil capito* exposed to cytochrome P4501A1 inducers indicates a certain degree of substrate specificity of this enzyme in fish. However, as the above-mentioned compounds were barely detectable in sediments from Aghada and Whitegate (Table. 1), it is unclear what caused the induction of cytochrome P4502B in the present study.

BROD activity shows a broader specificity and is a known marker for cytochromes P4501A, P4502B and P4503A in mammals. Cytochrome P4503A is also known to occur in various fish species, such as *M. capito* (Yawetz *et al.*, 1998b), *G. morhua*,

(Husoy *et al.*, 1996) and *Fundulus heteroclitus* (Celander & Stegeman, 1997) and can be induced by cytochrome P4501A inducers (Yawetz *et al.*, 1998b). Failure to detect BROD activity in the present study suggests that the induction of this enzyme is not only less specific but may also be less sensitive than EROD, MROD and PROD and therefore presumably less suitable as a biomarker for sediment contamination in turbot.

Baseline values for MROD and PROD activities in fish have generally been found to be much lower than those for EROD (Yawetz *et al.*, 1998a; Kennedy *et al.*, 2003) and this was also observed in the present study (Fig. 4). Although the EROD activities presented here are approximately three to four times higher following sediment exposure than those for MROD and PROD, the relative increase in activity above baseline is much higher for MROD and PROD (Table 2). This indicates that induction of MROD and PROD, as well as EROD in liver tissue samples can be suitably adopted to evaluate the effects of exposure of turbot to sediment-associated organic contaminants. However, a direct correlation of these activities with specific isoforms of cytochrome P450 in turbot remains to be established.

4.3. DNA integrity

In order to confirm that the Comet assay can detect DNA damage in turbot exposed to a known genotoxic agent, a control study was first carried out exposing a sub-sample of the turbot employed in the sediment-exposure to seawater spiked with $CdCl_2$, a substance known to produce DNA damage (Pruski & Dixon, 2002). Figure 5 clearly demonstrates $CdCl_2$-induced single-strand breaks above background levels in liver cells of exposed turbot.

In the present study the background values in juvenile turbot averaged 4.10 % ± 0.18 tail DNA. These results agree well with the 4 % and 5 % background reported in

isolated trout hepatocytes by Mitchelmore and Chipman, (1998a, b), respectively, and 4 % in liver cell suspensions from juvenile turbot (Belpaeme *et al.*, 1998).

The increase in DNA single strand breaks in turbot was significant following a 7-day exposure to contaminated sediments from both Aghada and Whitegate (Fig. 6). The observation, that % tail DNA values remained elevated after 14 and 21 days and that the values in fish exposed to the contaminated sediments remained consistently higher than those exposed to sediments from the reference site at Ballymacoda, suggest that beyond a seven day exposure, their genotoxic response was not time-related. This seems to support the findings of Deventer (1996) who reported that DNA damage in isolated blood cells of *Brachydanio rerio*, exposed to methyl methane sulphonate, increased initially and leveled off at around 96 hours. Furthermore, Belpaeme *et al.* (1998) reported significant DNA damage in turbot liver cell preparations exposed to ethyl methane sulphonate occurring only after 7 days exposure.

As mentioned above, many organic contaminants, including PAHs, undergo bioactivation by the monooxygenase system (Stegeman, 1981), producing reactive intermediates, including free radicals, which are known to form DNA adducts and cause oxidative DNA damage (Canova et al., 1998, Nacci *et al.*, 1992; Nacci *et al.*, 1996; Mitchelmore *et al.*, 1998, Xue and Warshawsky, 2005,), that can lead to an increase in the level of single strand breaks. In addition, organic contaminants, including PAHs, may cause an induction of the cytochrome P450 enzyme activity in exposed organisms (Myers *et al.*, 2003). However, an up-regulation of cytochrome P450, following exposure to complex contaminant mixtures, does not necessarily imply DNA damage, because some cytochrome P450 inducers, such as certain dioxins, are not genotoxic. Furthermore, not all genotoxic compounds are cytochrome P450 inducers (e.g. $CdCl_2$). Thus, although the observed effects in both assays used in

this study are probably mainly caused by the elevated levels of PAHs, these endpoints should be regarded as complementary.

5. Conclusions

The level of contamination in polluted sediments used here was, by comparison with other sites in Ireland and the British Isles, relatively low (Widdows *et al.*, 2002). Therefore, the present laboratory study demonstrates that *S. maximus* could be used as a sentinel species for the assessment of even moderately contaminated sediments and therefore allows for the further development of this sensitive model for future field studies.

Although no direct correlation between XROD induction and a particular chemical class was established here, the induction of MROD and PROD activities in fish exposed to sediments containing complex contaminant mixtures, were shown to be more sensitive than conventional EROD activity.

Acknowledgements

This work was funded by The Higher Education Authority under the Programme of Research in Third Level Institutions (Cycle 2), and the National Development Plan, as part of the Environmental Research Institute, University College Cork The authors would like to thank Brendan McHugh and Evan McGovern from the Marine Institute, Dublin, for organising the chemical analysis.

References

Aarnio, K., Bonsdorff, E. & Rosenback, N. (1996). Food and feeding habits of juvenile flounder *Platichthys flesus* (L), and turbot *Scophthalmus maximus* L in the Aland archipelago northern Baltic Sea. Journal of Sea Research, 36, 311-320.

Aas, E., Baussant, T., Balk, L., Liewenborg, B. & Andersen, O. K. (2000). PAH metabolites in bile, cytochrome P4501A and DNA adducts as environmental risk parameters for chronic oil exposure: a laboratory experiment with Atlantic cod. Aquatic Toxicology, 51, 241-258.

Aas, E., Beyer, J., Jonsson, G., Reichert, W. L. & Andersen, O. K. (2001). Evidence of uptake, biotransformation and DNA binding of polyaromatic hydrocarbons in Atlantic cod and corkwing wrasse caught in the vicinity of an aluminium works. Marine Environmental Research, 52, 213-229.

Anderson, D., Yu, T. W., Phillips, B. J. & Schmezer, P. (1994). The effect of various antioxidants and other modifying agents on oxygen-radical-generated DNA-damage in human Lymphocytes in the comet assay. Mutation Research, 307, 261-271.

Anderson, J. W., Birge, W. J., Gentile, J., Lake, J. & Rodgers, J. J. (1987). Biological effects. Bioaccumulation and ecotoxicology of sediment-associated chemicals. In Dickson, K. L. & Maki, A. W., Fate and Effects of Sediment-bound Chemicals in Aquatic Systems, (pp. 267-296) New York: Pergamon.

Arukwe, A. & Goksoyr, A. (1997). Changes in three hepatic cytochrome P450 subfamilies during a reproductive cycle in Turbot (*Scophthalmus maximus* L). Journal of Experimental Zoology, 277, 313-325.

Belpaeme, K., Cooreman, K. & Kirsch-Volders, M. (1998). Development and validation of the in vivo alkaline comet assay for detecting genomic damage in marine flatfish. Mutation Research, 415, 167-184.

Belpaeme, K., Delbeke, K., Zhu, L. & KirschVolders, M. (1996). PCBs do not induce DNA breakage in vitro in human lymphocytes. Mutagenesis, 11, 383-389.

Berndtson, A. K. & Chen, T. T. (1994). 2 Unique Cyp1 Genes Are Expressed in Response to 3- Methylcholanthrene Treatment in Rainbow-Trout. Archives of Biochemistry and Biophysics, 310, 187-195.

Besselink, H. T., Flipsen, E., Eggens, M. L., Vethaak, A. D., Koeman, J. H. & Brouwer, A. (1998). Alterations in plasma and hepatic retinoid levels in flounder (*Platichthys flesus*) after chronic exposure to contaminated harbour sludge in a mesocosm study. Aquatic Toxicology, 42, 271-285.

Beyer, J. U., Sandvik, M., Hylland, K., Fjeld, E., Egaas, E., Aas, E. & Skare, J. U. (1996). Contaminant accumulation and biomarker responses in flounder (*Platichthys flesus* L) and Atlantic cod (*Gadus morhua* L) exposed by caging to polluted sediments in Sorfjorden, Norway. Aquatic Toxicology, 36, 75-98.

Beyst, B., Cattrijsse, A. & Mees, J. (1999). Feeding ecology of juvenile flatfishes of the surf zone of a sandy beach. Journal of Fish Biology, 55, 1171-1186.

Boelens, R. G. V., Maloney, D. M., Parsons, A. P. & Walsh, A. R. (1999). Ireland's Marine and Coastal Areas and Adjacent Seas - An Environmental Assessment Marine Institute, Dublin. pp 388.

Boisson, F., Hartl, M. G. J., Fowler, S. W. & Amiard-Triquet, C. (1998). Influence of chronic exposure to silver and mercury in the field on the bioaccumulation potential of the bivalve *Macoma balthica*. Marine Environmental Research, 45, 325-340.

1 Boleas, S., Fernandez, C., Beyer, J., Tarazona, J. V. & Goksoyr, A. (1998).
2 Accumulation and effects of benzo(a)pyrene on cytochrome P450 1A in
3 waterborne exposed and intraperitoneal injected juvenile turbot (*Scophthalmus*
4 *maximus*). Marine Environmental Research, 46, 17-20.
5 Bradford, M. M. (1976). A rapid and sensitive method for quantification of
6 microgram quantities of protein utilizing the principle of protein-dye binding.
7 Analytical Biochemistry, 72, 248-254.
8 Burke, M. D. & Mayer, R. T. (1974). Ethoxyresorufin: direct flourometric assay of a
9 microsomal *O*-dealkylation which is preferentially inducible by 3-
10 methylchloanthrene. Drug Metabolism and Disposal, 2, 583-588.
11 Byrne, P. A. & O'Halloran, J. (1999). Aspects of assaying sediment toxicity in Irish
12 estuarine ecosystems. Marine Pollution Bulletin, 39, 97-105.
13 Camus, L., Aas, E. & Borseth, J. F. (1998). Ethoxyresorufin-O-deethylase activity and
14 fixed wavelength fluorescence detection of PAHs metabolites in bile in turbot
15 (*Scophthalmus maximus* L.) exposed to a dispersed topped crude oil in a
16 continuous flow system. Marine Environmental Research, 46, 29-32.
17 Canova S, Degan P, Peters LD, Livingstone DR, Voltan R, Venier P. (1998). Tissue
18 dose, DNA adducts, oxidative DNA damage and CYP1A-immunopositive
19 proteins in mussels exposed to waterborne benzo[a]pyrene.Mutation Research
20 399, 17-30.
21 Celander, M. & Forlin, L. (1992). Quantification of Cytochrome-P4501a1 and
22 Catalytic Activities in Liver-Microsomes of Isosafrol-Treated and Beta-
23 Naphthoflavone-Treated Rainbow-Trout (Oncorhynchus-Mykiss). Marine
24 Environmental Research, 34, 123-126.
25 Celander, M. & Stegeman, J. J. (1997). Isolation of a cytochrome P450 3A cDNA
26 sequence (CYP3A30) from the marine teleost Fundulus heteroclitus and
27 phylogenetic analyses of CYP3A genes. Biochemical and Biophysical
28 Research Communications, 236, 306-312.
29 Coughlan, B. M., Hartl, M. G. J., O'Reilly, S. J., Sheehan, D., Morthersill, C., van
30 Pelt, F., O'Halloran, J. & O'Brien, N. M. (2002). Detecting genotoxicity using
31 the Comet assay following chronic exposure of Manila clam *Tapes*
32 *semidecussatus* to polluted estuarine sediments. Marine Pollution Bulletin, 44,
33 1359-1365.
34 Courtney, W. A. M. & Langston, W. J. (1980). Accumulation of polychlorinated
35 biphenyls in turbot (*Scrophthalmus maximus*) from sea water sediments and
36 food. Helgoländer Meeresuntersuchungen, 33, 333-339.
37 Devaux, A., Pesonen, M. & Monod, G. (1997). Alkaline comet assay in rainbow trout
38 hepatocytes. Toxicology in Vitro, 11, 71-79.
39 Deventer, K. (1996). Detection of genotoxic effects on cells of liver and gills of *B.*
40 *rerio* by means of single cell gel electrophoresis. Bulletin of Environmental
41 Contamination and Toxicology, 56, 911-918.
42 Eggens, M., Bergman, A., Vethaak, D., Vanderweiden, M., Celander, M. & Boon, J.
43 P. (1995). Cytochrome p4501a indices as biomarkers of contaminant exposure
44 - results of a field study with plaice (*Pleuronectes platessa*) and flounder
45 (*Platichthys flesus*) from the southern North Sea. Aquatic Toxicology, 32,
46 211-225.
47 Eggens, M. L., Vethaak, A. D., Leaver, M. J., Horbach, G. J. M. J., Boon, J. P. &
48 Seinen, W. (1996). Differences in CYP1A response beteween flounder
49 (*Platichthys flesus*) and plaice (*Pleuronectes platesa*) after long-term exposure
50 to harbour dredge spoil in a mesocoms study. Chemosphere, 32, 1357-1380.

Frenzilli, G., Scarcelli, V., Taddei, F. & Nigro, M. (1999). Adaptation of SCGE for monitoring marine ecosystems. Neoplasma, 46, 6.

Fry, J. C. (1993). Biological Data Analysis. A practical Approach. (pp. 439). Oxford: IRL Press - Oxford University Press

Haasch, M. L., Graf, W. K., Quardokus, E. M., Mayer, R. T. & Lech, J. J. (1994). Use of 7-Alkoxyphenoxazones, 7-Alkoxycoumarins and 7- Alkoxyquinolines as Fluorescent Substrates for Rainbow-Trout Hepatic Microsomes after Treatment with Various Inducers. Biochemical Pharmacology, 47, 893-903.

Hartl, M. G. J., Hutchinson, S. & Hawkins, L. E. (2001). Sediment-associated tri-n-butyltin chloride and its effects on osmoregulation of freshwater-adapted 0-group European flounder, Platichthys flesus (L.). Aquatic Toxicology, 55, 125-136.

Hartmann, A., Agurell, E., Beevers, C., Brendler-Schwaab, S., Burlinson, B., Clay, P., Collins, A., Smith, A., Speit, G., Thybaud, V. & Tice, R. R. (2003). Recommendations for conducting the *in vivo* alkaline Comet assay. Mutagenesis, 18, 45-51.

Husoy, A. M., Myers, M. S. & Goksoyr, A. (1996). Cellular localization of cytochrome P450 (CYP1A) induction and histology in Atlantic cod (Gadus morhua L.) and European flounder (Platichthys flesus) after environmental exposure to contaminants by caging in Sorfjorden, Norway. Aquatic Toxicology, 36, 53-74.

Kammann, U., Riggers, J. C., Theobald, N. & Steinhart, H. (2000). Genotoxic potential of marine sediments from the North Sea. Mutation Research-Genetic Toxicology and Environmental Mutagenesis, 467, 161-168.

Kennedy, S. W., Fox, G. A., Jones, S. P. & Trudeau, S. F. (2003). Hepatic EROD activity is not a useful biomarker of polychlorinated biphenyl exposure in the adult herring gull (Larus argentatus). Ecotoxicology, 12, 153-161.

Kilemade, M., Hartl, M. G. J., Sheehan, D., van Pelt, F. N. A. M., O'Brien, N. M. & O'Halloran, J. (2004a). An assessment of the pollutant status of surficial sediment in Cork Harbour in the South East of Ireland with particular reference to polycyclic aromatic hydrocarbons. Marine Pollution Bulletin, 49, 1084-1096.

Kilemade, M. F., Hartl, M. G. J., Sheehan, D., Mothersill, C., van Pelt, F., O'Halloran, J. & O'Brien, N. M. (2004b). Genotoxicity of field-collected inter-tidal sediments from Cork Harbor, Ireland, to juvenile turbot (*Scophthalmus maximus* L.) as measured by the Comet assay. Environmental and Molecular Mutagenesis, 44, 56-64.

Leaver, M. J., Burke, M. D., George, S. G., Davies, J. M. & Raffaelli, D. (1988). Induction of cytochrome-P-450 monooxygenase activities in plaice by model inducers and drilling muds. Marine Environmental Research, 24, 27-30.

Lindstrom-Seppa, P., Huuskonen, S., Pesonen, M., Muona, P. & Hanninen, O. (1992). Unbleached pulp mill effluents affect cytochrome-P450 monooxygenase enzyme activities. Marine Environmental Research, 34, 157-161.

Long, E. R., MacDonald, D. D., Cubbage, J. C. & Ingersoll, C. G. (1998). Predicting the toxicity of sediment-associated trace metals with simultaneously extracted trace metal: Acid-volatile sulfide concentrations and dry weight-normalized concentrations: A critical comparison. Environmental Toxicology and Chemistry, 17, 972-974.

Machala, M., Nezveda, K., Petrivalsky, M., Jarosova, A. B., Piacka, V. & Svobodova, Z. (1997). Monooxygenase activities in carp as biochemical markers of

pollution by polycyclic and polyhalogenated aromatic hydrocarbons: Choice of substrates and effects of temperature, gender and capture stress. Aquatic Toxicology, 37, 113-123.

Mitchelmore, C. L., Birmelin, C., Chipman, J. K. & Livingstone, D. R. (1998). Evidence for cytochrome P-450 catalysis and free radical involvement in the production of DNA strand breaks by benzo[a]pyrene and nitroaromatics in mussel (Mytilus edulis L.) digestive gland cells. Aquatic Toxicology, 41, 193-212.

Mitchelmore, C. L. & Chipman, J. K. (1998a). Detection of DNA strand breaks in brown trout (*Salmo trutta*) hepatocytes and blood cells using the single cell gel electrophoresis (comet) assay. Aquatic Toxicology, 41, 161-182.

Mitchelmore, C. L. & Chipman, J. K. (1998b). DNA strand breakage in aquatic organisms and the potential value of the comet assay in environmental monitoring. Mutation Research, 399, 135-147.

Mizukami, Y., Okauchi, M., Arizono, K., Ariyoshi, T. & Kito, H. (1994). Isolation and sequence of cDNA encoding a 3-methylcholanthrene-iInducible cytochrome-P450 from wild redsea bream, *Pagrus major*. Marine Biology, 120, 343-349.

Mondon, J. A., Duda, S. & Nowak, B. F. (2001). Histological, growth and 7-ethoxyresorufin O-deethylase (EROD) activity responses of greenback flounder *Rhombosolea tapirina* to contaminated marine sediment and diet. Aquatic Toxicology, 54, 231-247.

Myers, M. S., Johnson, L. L. & Collier, T. K. (2003). Establishing the causal relationship between polycyclic aromatic hydrocarbon (PAH) exposure and hepatic neoplasms and neoplasia-related liver lesions in English sole (Pleuronectes vetulus). Human and Ecological Risk Assessment, 9, 67-94.

Nacci, D., Nelson, S., Nelson, W. & Jackim, E. (1992). Application of the DNA alkaline unwinding assay to detect DNA strand breaks in marine bivalves. Marine Environmental Research, 33, 83-100.

Nacci, D. E., Stephanie, C. & Jackim, E. (1996). Detection of DNA damage in individual cells from marine organisms using the single cell gel assay. Aquatic Toxicology, 35, 197-210.

Peters, L. D. & Livingstone, D. R. (1995). Studies on Cytochrome P4501a in Early and Adult Life Stages of Turbot (*Scophthalmus maximus* L). Marine Environmental Research, 39, 5-9.

Peters, L. D., Morse, H. R., Waters, R. & Livingstone, D. R. (1997). Responses of hepatic cytochrome P450 1A and formation of DNA- adducts in juveniles of turbot (*Scophthalmus maximus* L) exposed to water-borne benzo[a]pyrene. Aquatic Toxicology, 38, 67-82.

Pruski, A. M. & Dixon, D. R. (2002). Effects of cadmium on nuclear integrity and DNA repair efficiency in the gill cells of Mytilus edulis L. Aquatic Toxicology, 57, 127-137.

Rice, D. W., Seltenrich, C. P., Keller, M. L., Spies, R. B. & Felton, J. S. (1994). Mixed-function oxidase-specific activity in wild and caged speckled sanddabs *Citharichthys stigmaeus* in Elkhorn Slough, Moss Landing Harbor and Nearshore Monterey Bay, California. Environmental Pollution, 84, 179-188.

Risso-de Faverney, C., Devaux, A., Lafaurie, M., Girard, J. P., Bailly, B. & Rahmani, R. (2001). Cadmium induces apoptosis and genotoxicity in rainbow trout hepatocytes through generation of reactive oxygene species. Aquatic Toxicology, 53, 65-76.

Rotchell, J. M., Steventon, G. B. & Bird, D. J. (2000). Catalytic properties of CYP1A isoforms in the liver of an agnathan (Lampetra fluviatilis) and two species of teleost (Pleuronectes flesus, Anguilla anguilla). Comparative Biochemistry and Physiology C, C125, 203-214.

Schiff, K. & Allen, M. J. (2000). Chlorinated hydrocarbons in flatfishes from the Southern California, USA, Bight. Environmental Toxicology and Chemistry, 19, 1559-1565.

Singh, N. P., McCoy, M. T., Tice, R. R. & Schneider, E. L. (1988). A simple technique for the quantitation of low levels of DNA damage in individual cells. Experimental Cell Research, 175, 184-191.

Sleiderink, H. M., Beyer, J., Scholtens, E., Goksoyr, A., Nieuwenhuize, J., Vanliere, J. M., Everaarts, J. M. & Boon, J. P. (1995). Influence of temperature and polyaromatic contaminants on Cyp1a levels in North Sea dab (*Limanda limanda*). Aquatic Toxicology, 32, 189-209.

Smeets, J. M. W., Wamsteker, J., Roth, B., Everaarts, J. & van den Berg, M. (2002). Cytochrome P4501A induction and testosterone hydroxylation in cultured hepatocytes of four fish species. Chemosphere, 46, 163-172.

Stegeman, J. J. (1981). Polynuclear aromatic hydrocarbons and their metabolisms in the marine environment. In Gelboin, H. V. & Ts'o, P. O. P., Polycyclic Hydrocarbons and Cancer, (pp. 1-60) London: Academic Press.

Stegeman, J. J. (1989). Cytochrome P-450 forms in fish: catalytic, immunological and sequence similarities. Xenobiotica, 19, 1093-1110.

Stegeman, J. J. & Hahn, M. E. (1994). Biochemistry and molecular biology of monooxygenase: Current perspectives on forms, functions, and regulation of cytochrome p450 in aquatic species. In Malins, D. C. & Ostrander, G. K., Aquatic toxicology - Molecular biochemistry and cellular perspectives, (pp. 87- 206) Boca Raton: CRC Press.

Tetreault, G. R., McMaster, M. E., Dixon, D. G. & Parrott, J. L. (2003). Physiological and biochemical responses of Ontario slimy sculpin (*Cottus cognatus*) to sediment from the Athabasca Oil Sands area. Water Quality Research Journal of Canada, 38, 361-377.

Valverde, M., Trejo, C. & Rojas, E. (2001). Is the capacity of lead acetate and cadmium chloride to induce genotoxic damage due to direct DNA-metal interaction? Mutagenesis, 16, 265-270.

Widdows, J., Donkin, P., Staff, F. J., Matthiessen, P., Law, R. J., Allen, Y. T., Thain, J. E., Allchin, C. R. & Jones, B. R. (2002). Measurement of stress effects (scope for growth) and contaminant levels in mussels (Mytilus edulis) collected from the Irish Sea. Marine Environmental Research, 53, 327-356.

Woods, J. A., O'Leary, K. A., McCarthy, R. P. & O'Brien, N. M. (1999). Preservation of comet assay slides: comparison with fresh slides. Mutation Research-Fundamental and Molecular Mechanisms of Mutagenesis, 429, 181-187.

Xue W, Warshawsky D. (2005) Metabolic activation of polycyclic and heterocyclic aromatic hydrocarbons and DNA damage: a review.Toxicology Applied Pharmacolology 206,73-93.

Yawetz, A., Woodin, B. R. & Stegeman, J. J. (1998a). Cytochromes P450 in liver of the turtle Chrysemys picta picta and the induction and partial purification of CYP1A-like proteins. Biochimica Et Biophysica Acta, 1381, 12-26.

Yawetz, A., Zilberman, B., Woodin, B. & Stegeman, J. J. (1998b). Cytochromes P-4501A, P-4503A and P-4502B in liver and heart of Mugil capito treated with CYP1A inducers. Environmental Toxicology and Pharmacology, 6, 13-25.

Table 1
Sediment characterisation and chemical analysis.

	Ballymacoda (Site 1)	Whitegate (Site 2)	Aghada (Site 3)
Dry weight (%)	57	49	46
TOC (%)	3.11	2.77	3.66
< 63 μm fraction (%)	75.15	57.66	76.66
Organics (ng g^{-1})			
Σ PAH	528.30	924.40	1000.7
Σ PCB	2.90	3.20	3.40
Σ OCP	3.31	3.22	3.25
Σ BFR	0.50	0.80	4.70
Σ Organotin	4.90	9.50	12.50
Metals (μg g^{-1})			
Cadmium	<0.1	<0.1	<0.1
Copper	8.1	13.3	27
Lead	15.6	16.4	18
Zinc	78.6	99.1	105.3

TOC: Total Organic Carbon; PAH: Polycyclic Aromatic Hydrocarbons; PCB: Polychlorinated Biphenyls; OCP: Organochlorine pesticide; BFR: Brominated Flame Retardants.

Table 2
Increase of cytochrome P450 enzyme activity above t_0, following exposure to sediment from the reference and contaminated sites (t = days).

		% increase from t_0		
	Site	t_7	t_{14}	t_{21}
EROD	Reference	135	65	59
	Aghada	444	335	393
	Whitegate	427	393	460
BROD	Reference	80	75	68
	Aghada	80	37	65
	Whitegate	n.d*.	67	71
MROD	Reference	41	22	905
	Aghada	1904	1913	1722
	Whitegate	1435	2370	2414
PROD	Reference	82	43	141
	Aghada	1514	1438	2489
	Whitegate	1574	1775	4210

*) no data

Fig. 1. Sediment sampling sites (1) Reference site at Ballymacoda; contaminated sites in Cork Harbour (2) Whitegate and (3) Aghada.

Fig. 2. Western blot analysis of cytochrome P4501A1 expression in liver samples of turbot following a 21 day exposure to contaminated sediments. MW, molecular weight marker; BNF, *β*-naphthoflavone positive control; C, control; t_0, before exposure to contaminated sediments; B, Reference site at Ballymacoda; A, contaminated site at Aghada and W, Whitegate.

Fig. 3. Validation of P450 activity (EROD) assay using BNF (40 mg kg^{-1} *i.p.* in DMSO) as a positive control. * indicating significant difference; n = 5; ($P < 0.05$). Mean ± standard deviation (SD)

Fig. 4. P450 activity before and after exposure to sediments from the Reference site a Ballymacoda and the contaminated sites at Aghada and Whitegate. EROD, 7-ethoxyresorufin-O-deethoxylase; PROD, 7-pentoxyresorufin O-dependentylase; BROD, 7-benzyloxyresorufin O-debenzylase MROD, 7-methoxyresorufin O-methoxyresorufin. ** indicating significant difference between the Reference site at Ballymacoda and t_0 ($P <0.05$); * indicating significantly higher values in the contaminated site at Aghada than Ballymacoda and t_0 ($P< 0.05$); † indicating significantly higher values in the contaminated site at Whitegate than Aghada, Ballymacoda and t_0 ($P <0.05$); n = 8. Mean ± SD

Fig. 5. Validation of the Comet assay performed on liver single-cell suspensions using $CdCl_2$ as a positive control. * indicating significant difference between 0 and 4 µM (P

1 < 0.05); † indicating significantly higher values at 40 than 0 and 4 μM (P <0.05); n =
2 5. Mean ± SD
3
4 Fig. 6. Comet assay showing the percentage DNA in the tail in liver single-cell
5 suspensions from turbot before and after exposure to sediments from the Reference
6 site a Ballymacoda and the contaminated sites at Aghada and Whitegate. * indicating
7 significant difference between the Reference site at Ballymacoda and t_0 (P <0.05); †
8 indicating significantly higher values in the contaminated site at Aghada than
9 Ballymacoda and t_0 (P< 0.05); †† indicating significantly higher values in the
10 contaminated site at Whitegate than Aghada, Ballymacoda and t_0 (P < 0.05); n = 8.
11 Mean ± SD
12

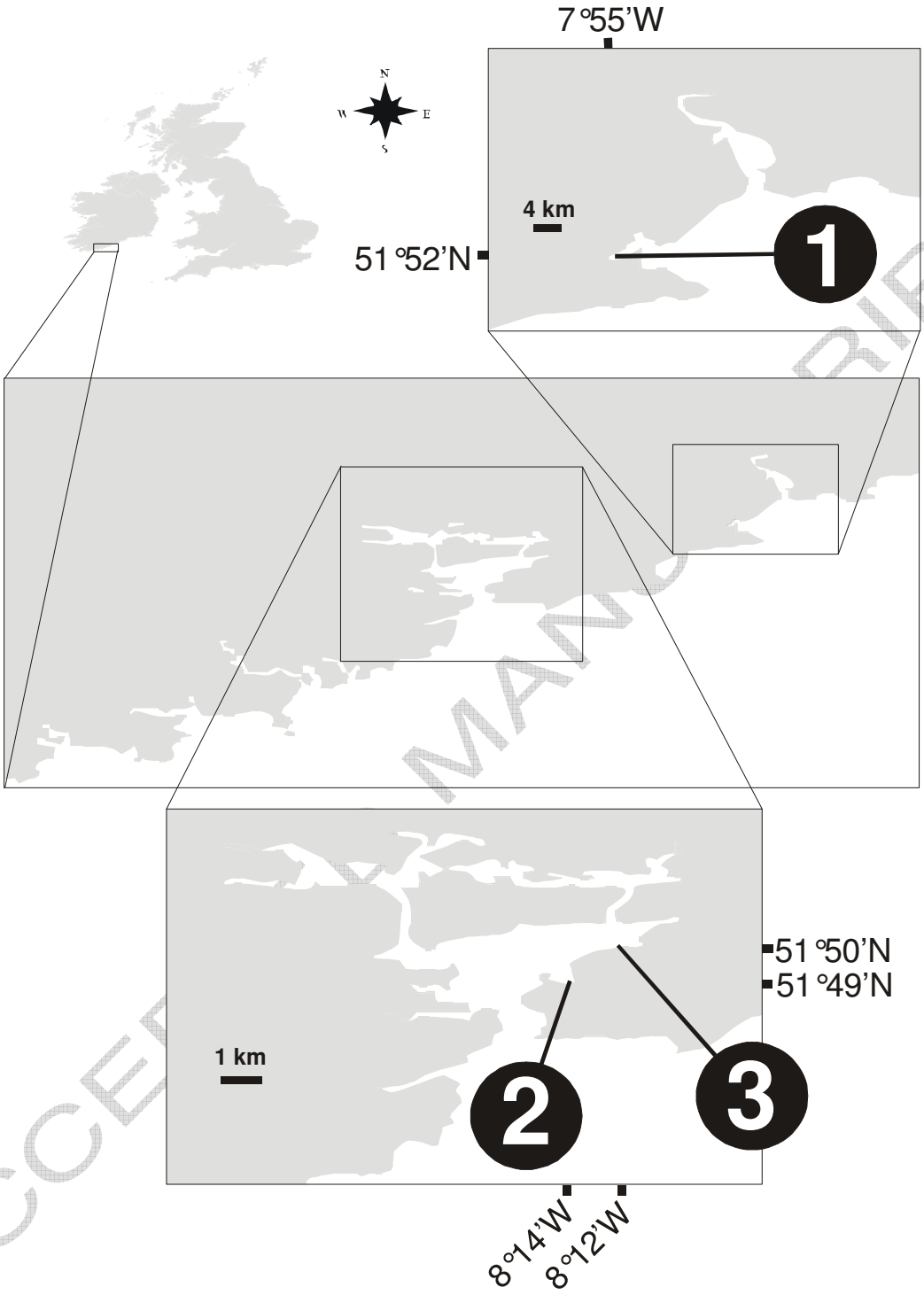


Fig. 1

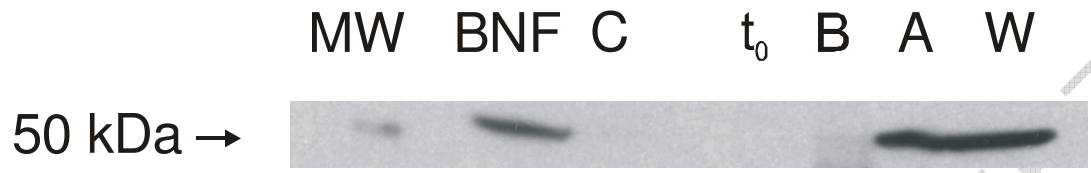


Fig. 2

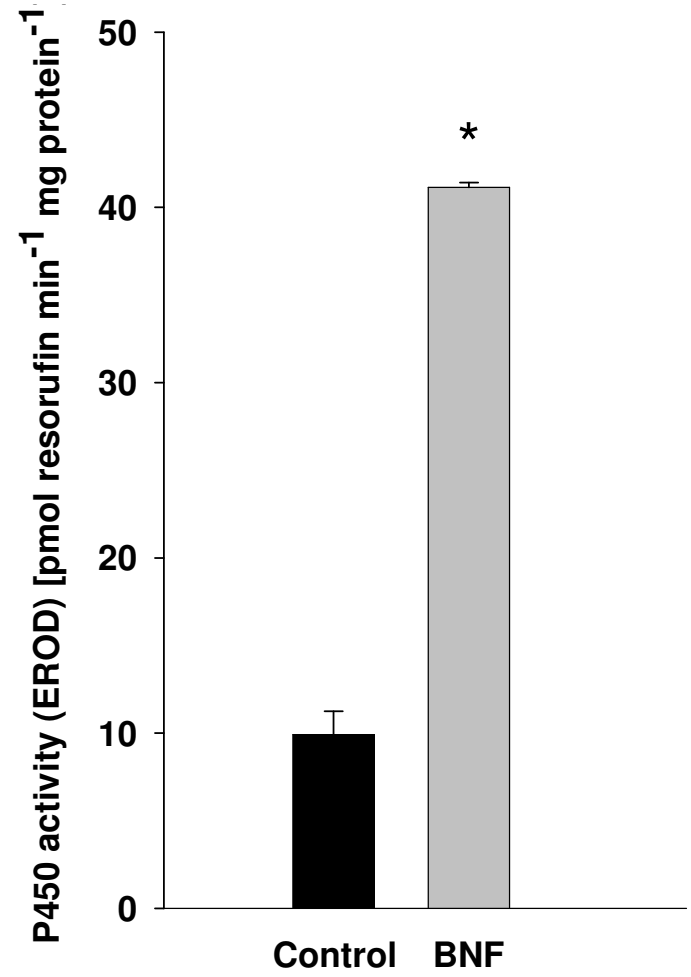

P450 activity (EROD) [pmol resorufin min-1 mg protein-1
50
40
30
20
10
0
*
Control
BNF

Fig.3

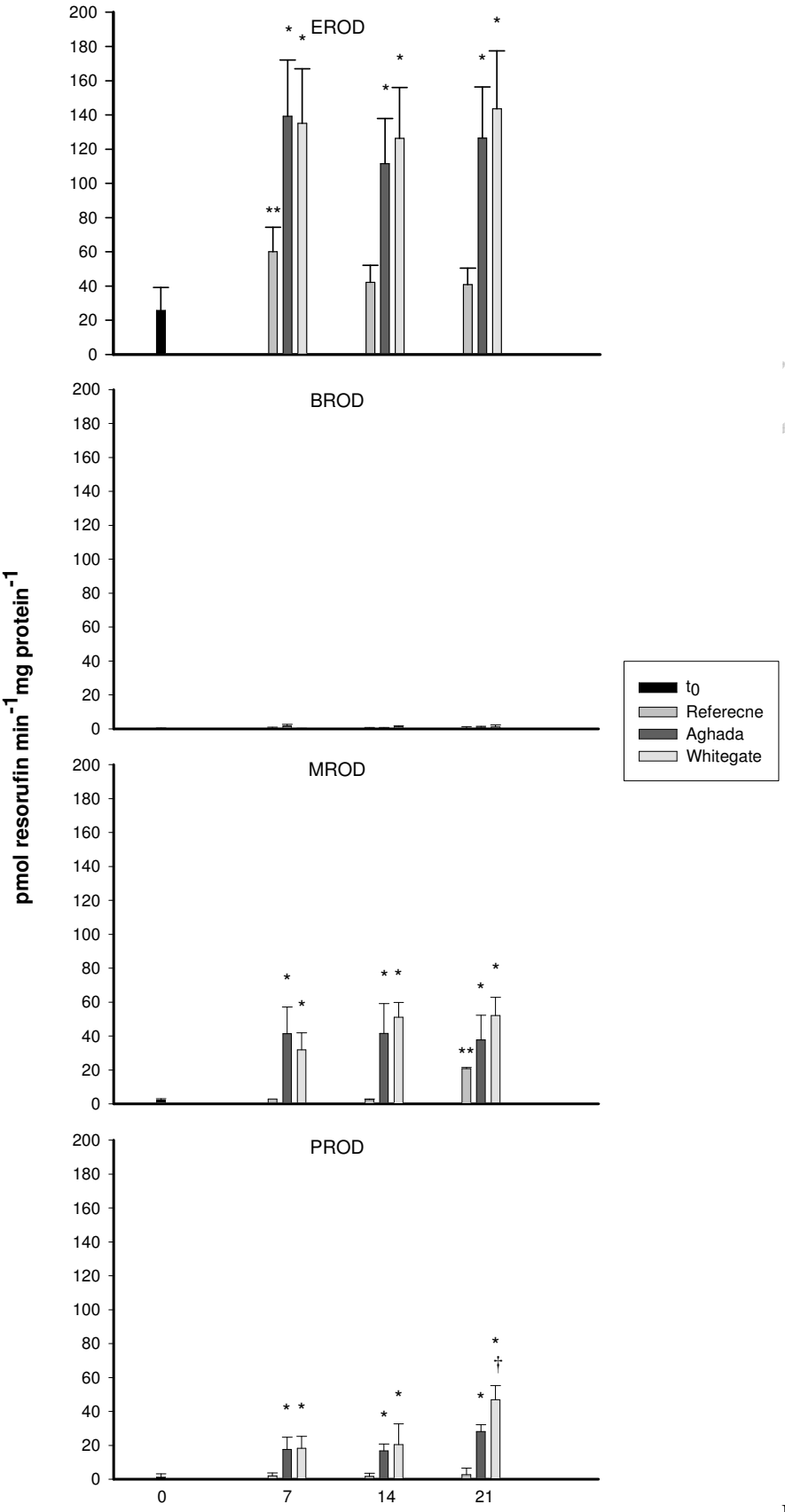

EROD
BROD
MROD
PROD
pmol resorufin min-1 mg protein-1
t0
Referecne
Aghada
Whitegate
0
7
14
21

Fig. 4

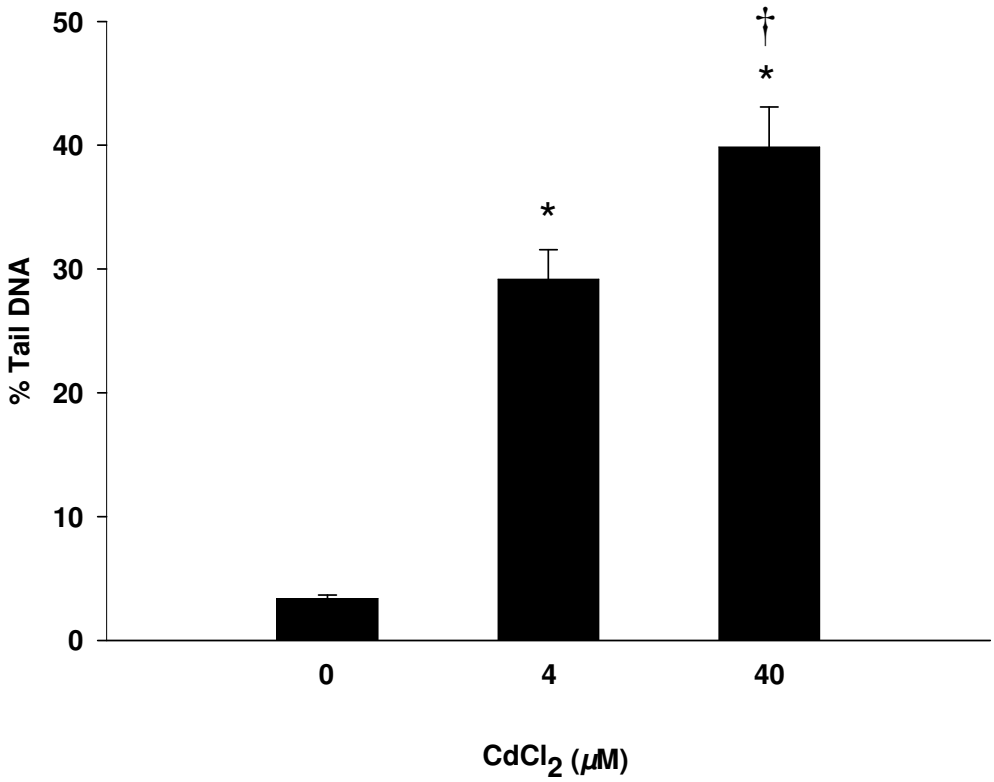


Fig. 5.

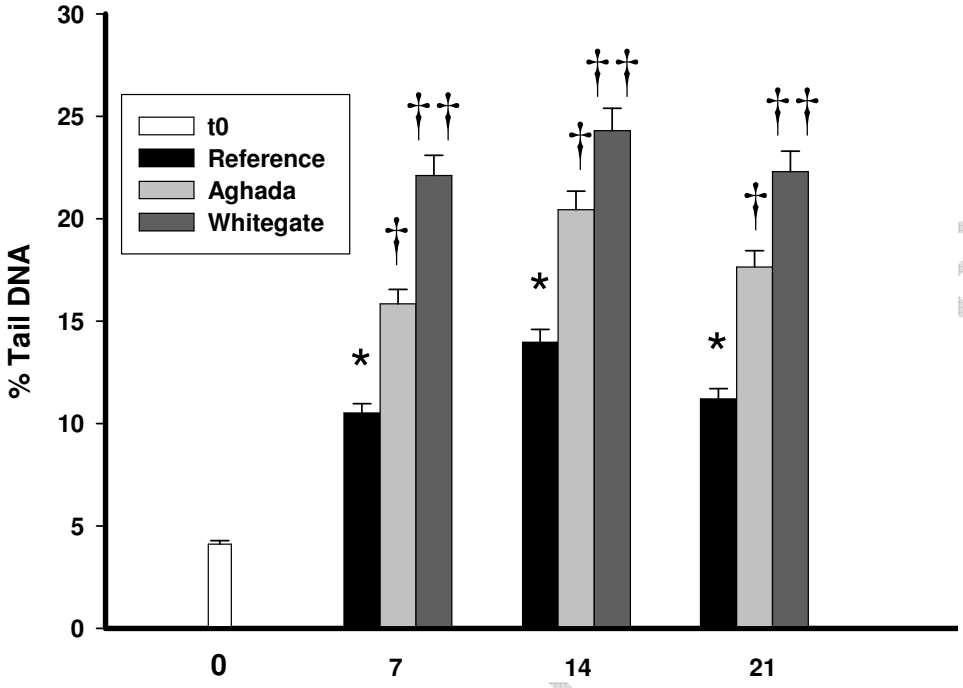


Fig.6