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Note

Use of flathead mullet (*Mugil cephalus*) in coastal biomonitor studies: Review and recommendations for future studies

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ABSTRACT

There has been a widespread world-wide use of flathead mullet, *Mugil cephalus*, in fish biomonitor studies within the coastal zone. This review summarises this research field, focusing on heavy metals, and considers the implications of the accumulated data. Differences in sampling methodology, tissues analysed and units of reported data provide challenges in assessing and benchmarking these biomonitor studies. The benthic feeding strategy of *M. cephalus* invariably increases exposure risk relative to middle or upper water column feeders, nevertheless contaminant accumulation via direct and indirect pathways was regulated sufficiently such that toxicants were below food guidelines in most coastal regions (32 of the 49 examined). Human health issues can arise if fish are consumed from heavily industrialised regions. Recommendations are provided for future biomonitoring studies, based on the results for *M. cephalus* but relevant for fish species more broadly, to provide more comparable data so that managers can benchmark against local conditions.

1. Introduction

The urban coastal landscape consists of a mixture of heavy industry, residential development, port facilities, sewage treatment facilities, golf courses and urban road networks (Pauly et al., 2002; Waltham and Connolly, 2011). This mosaic of anthropogenic activity contributes widespread and varied contamination, often untreated, into the coastal zone, particularly estuaries (Kennish, 2002; Lee et al., 2006). This cocktail of contaminants can persist within the water column or can be stored in sediments for many decades. Consequentially, toxic contaminants are readily available for uptake and accumulation in coastal food webs rendering estuaries, in particular, of interest for biomonitoring studies. Consumption of contaminated local seafood has implications for human health and is of interest to coastal managers challenged with achieving ecosystem conservation while approving further urban development.

Modern monitoring programs frequently integrate fish data and measurements with abiotic parameters (e.g. measures of water and sediment quality) to allow evaluation of the response and resilience of fish species to contaminants in coastal ecosystems (e.g. Water framework directive of the European Union, Allan et al., 2006; Chesapeake Bay Health and Restoration Assessment, www.chesapeakebay.org, Chesapeake Bay Program, 2008). These biomonitoring programs have been particularly successful because of the advantages over targeted water and sediment programs, including direct insight into ecosystem response to contamination, mechanisms of contaminant uptake by fish, and because their loss has societal costs (Whitfield and Elliott, 2002). The uptake of contaminants by fish is generally known to be either via indirect or direct pathways, with distinction between each route assessed by examining specific tissue types that will accumulate higher contaminant concentrations over other tissue types (Dallinger et al., 1987).

Mugil cephalus, (flathead mullet; other common names include striped mullet and sea mullet) inhabit estuarine intertidal, freshwater and coastal marine habitats worldwide from approximately 42°N–42°S latitude (Whitfield et al., 2012). Of all the species within the family Mugilidae, *M. cephalus* is the most abundant and commonly caught in fish investigations. Juvenile fish (<25 mm) are primarily planktonic or carnivorous feeders, whereas beyond that size diet switches to primarily detritivorous, ingesting large amounts of organic matter, sand or mud from the sediment of waterways (Whitfield et al., 2012). *M. cephalus* tends to have a longer gut length compared to other fish species, which would conceivably increase absorption efficiency of contaminants once ingested (Odum, 1970). *M. cephalus* supports commercial and recreational

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fisheries owing to the high protein and vitamins contained in the muscle tissue, the value of roe from females, and are a stable food source in many countries (Whitfield et al., 2012).

This study presents the first review of biomonitor studies, focusing only on heavy metals, published on *M. cephalus*. We examined which coastal regions contained fish unsafe for human consumption, and attempted to evaluate mechanisms of contaminant uptake. We also provide recommendations for future biomonitor studies to facilitate an objective determination of global benchmarking for local fish contamination.

2. Methods

2.1. Search method

In order to capture publications investigating biomonitoring using *M. cephalus*, the literature was examined using the key words: *M. cephalus** AND heavy metals* AND contaminants* AND coastal zone* AND estuaries*. These search terms were applied to three major biological databases: Aquatic Sciences Fisheries Abstracts, Web of Science, and Proquest. In addition, we used Google Scholar search engine to locate government or non-government organisation publications. In all cases, the search approaches were crossed-referenced against one another to produce a final selection of studies ensuring the widest search of the literature. Studies of *M. cephalus* kept in aquaculture facilities, or those specifically examining organophosphates and organochlorines were excluded because of the low number of studies for any meaningful analysis.

2.2. Meta-analysis

Water, sediment and fish tissue data presented in each publication fitting the above criteria were extracted and stored in a database. In situations where the laboratory limit of detection was shown, the result was halved (Quinn and Keough, 2008). In all cases, the average concentration value presented in publications was used. Most publications provided data in tabular format, however, in some cases the data was retrieved directly from graphs, with the primary author contacted on several occasions to clarify interpretation. Several publications included reference sites to benchmark conditions against samples collected from polluted sites; both the reference and impact site data were included in the review in these instances.

Examination of *M. cephalus* contaminant concentrations among studies was problematic given the array of different units used, in particular regarding use of either dry weight or wet weight. To standardise this measurement we adjusted for the water content in tissue following Kirby et al. (2001), where a correction factor of 4.5 was applied to wet tissue results to compensate for the dilution of metal concentrations caused by excess water content. This correction factor is in agreement with Marks et al. (1980), who reported a similar relationship for a range of fish species, including *M. cephalus*, regardless of metal element (nine examined). We also standardised differences in the reported unit mass of metals (e.g. mg/kg and μg/g), such that all values presented are μg/g. Because of the range of guidelines available for comparison to metal concentrations recorded in different studies, we standardised this assessment by using the water (95% ecosystem protection) and sediment guidelines from the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ, 2000) and the Australian and New Zealand Food Safety Standard (ANZFA, 1999) for edible muscle tissue analysis. We have selected these water guidelines for the basis of this assessment as they are very comprehensive and represent a major development in water quality guidelines. The guideline trigger values for toxicants in water are risk-based and derived from appropriate international toxicity data and some Australian and New Zealand data (Warne, 2001); the toxicity data for relevant organisms was arranged in a statistical distribution and the trigger values for the different levels of protection obtained by extrapolation. The sediment guidelines are based on other international guidelines (ANZECC/ARMCANZ, 2000).

3. Results and discussion

3.1. Summary of meta-analysis

This is the first study to complete an evaluation of fish biomonitor studies for a single species with study sites spread across many coastal locations around the globe. A total of 49 *M. cephalus* biomonitor studies were included in our evaluation, encompassing a large proportion of the geographical range of this species (Table 1). All studies had a similar research objective, but there were often major differences in the approach used, including sampling methods and data collection. For example, there were differences from study to study in collection of *M. cephalus*, with just over half (55%) the studies using traditional gill nets and traps, while the remaining 45% purchased specimens directly from professional fishermen. Direct netting allows better quality control over handling and storage than purchasing, but potentially results in unnecessary bycatch of non-target species. While several studies (15%) included spatial (impact and control sites) and temporal (seasonal) sampling components in the design, most studies centred on a single survey event or season. Contaminant concentration data were reported using different units (e.g. μg/g dry weight, μg/g wet weight, mg/kg dry weight, and mg/kg wet weight). In several cases where no units were reported we clarified units of measurement with the primary author. There were also variations in the fish organs tested, and while most studies included gills, liver and muscle, or a combination of these, several studies included brains, kidney, gonads, roe, blood, heart, intestine, stomach, spline, and bone fragments. There were differences in the list of different contaminants examined; a reaction, presumably, to local environmental conditions, land use practices, management priorities, or known contaminant problems. All these variations have posed a challenge for our comparison, and careful assessment and consideration was given.

Biomonitor surveys have an inherent surveillance monitoring focus whereby attention is centred on examining the impact that local land use activities have on water, sediment and fish contaminant levels. All studies had this focus, but also had a secondary objective of determining whether metal concentrations in edible muscle tissue pose a human health risk. Only a small subset of the total studies had a research focus, for example, Maher et al. (1999) examined the distribution of As compounds in a series of fish tissues and was able to construct a conceptual model of total As and As compounds in tissue relative to the circulatory system. Two studies had a strong statistical experimental design; one used a BACI design (Before After Control Impact) to examine the success of land use restoration on decreasing heavy metal concentrations in the tissue of *M. cephalus* (Huang et al., 2008). The second study inspected changes in fish tissue heavy metals collected before and after a tsunami (Prasath and Khan, 2008).

3.2. Consideration of fish biology in biomonitor studies

3.2.1. Fish size, weight and sex

Understanding the biology of the target fish species is important in biomonitoring surveys particularly given a tendency of ontogenetic shift in diet, habitat preference changes, spawning

Table 1
Summary of global mullet (*Mugil cephalus*) biomonitor investigations.

Author	Continent/country/habitat (N)	Sampling design and Collection methods	Length (mm, TL)	Weight (g)	QAQC	Environment				
					Laboratory (% CRM)	Water	Sediment	Fish Tissue		
								Gills	Liver	Muscle
(1) Windom et al. (1973)	Nth America: Gulf Stream, Sargasso Sea (Sth Carolina, Georgia, Florida) (11)	Various locations	X	X	✓ 10–30	X	X	X	X	✓[d]
(2) Eustace (1974)	Australia: Derwent Estuary (38)	Purchased over 8 wks between November 1972 and January 1973	✓[a] 190–27, FL	X	X	X	X	X	X	✓[d]
3) Bebbington et al. (1977)	Australia: Unknown, NSW (30)	Unknown	✓ [a] 280–540, FL	✓ [a] 295–2800	✓ 90–100	X	X	X	X	✓[c]
(4) Plaskett and Potter (1979)	Australia: Cockburn Sound (57)	Seine, trawl or mesh nets, across known metal pollution gradient	✓ [a] 130–330	✓ [a] 13–301	✓ 95–105	X	X	X	X	✓[d]
5) Marks et al. (1980)	Australia: Swan-Avon Estuary (163)	Beach seine & gill net at 10 locations covering 28 km between February 1987 and April 1979	✓ [a] 55–430	✓ [b] 24–832	✓ 95–105	X	X	X	X	✓[d]
(6) Hamza-Chaffai et al. (1996)	Tunisia: Middle Eastern Coast (4)	September and October 1989	X	X	✓ within 4–9	X	X	X	X	✓[d]
(7) Sultana and Rao (1998)	India: (a) Visakhapatnam Harbour (46)	Monthly between March 1986–February 1987 from impacted harbour	X	X	X	✓	X	✓	✓	✓[d]
	(b) Gostani Estuary (10)	Monthly during summer of 1986, as controls	X	X	X	✓	X	✓	✓	✓[d]
(8) Kalay et al. (1999)	Turkey: NE Mediterranean Sea (60)	Purchased November 1996	✓ [a] 269 + 25.3	✓ [a] 182 + 49	✓ within 10	X	X	✓	✓	✓[d]
(9) Maher et al. (1999)	Australia: Lake Macquarie (36)	Gill net in April 1997	X	✓ [b] NP	✓ within 98	X	X	X	X	✓[d]
(10) Colombo et al. (2000)	Sth America: Rio de la Plata Estuary (30)	Collected between October and December 1996	✓ [a] 410 + 20	✓ [a] 1200 ± 200	✓ within 3–11	X	X	X	X	✓[d]
(11) Kirby et al. (2001)	Australia: (a) Lake Macquarie (33)	Sediment cores (surface 50 mm) collected using stainless steel shovel. Fish collection with gill net in May 1993	X	X	✓ 86–112	X	✓	X	✓	✓[d]
	(b) Clyde River Estuary (15)						X	X	X	X
(12) Chen (2002)	Taiwan: Chi-ku Lagoon (3)	Sediment (surface 10 cm,<63um fraction in triplicate) in November 1996 and April 1997, fish caught each season between September 1996 and December 1998	✓ [b] 333–558	X	✓ within 5	X	✓	X	✓	✓[d]
(13) Canli and Atli (2003)	Turkey: NE Mediterranean Sea (20)	Purchased	✓ [b] 150–362	✓ [b] 25–41	✓ within 10	X	X	✓	✓	✓[d]
(14) Yilmaz (2003)	Turkey: Iskenderun Bay (30)	Purchased in September 2001	✓ [a] 218 ± 14	✓ [a] 73.2 g ± 3	X	X	X	X	X	✓[c] and [d]
(15) Chen et al. (2004)	Taiwan: Erren River (6)	Gill net at junction of Saynegong River and Erren River in January 2002	✓ [a] 322 ± 1.8	✓ [a] 273 ± 54	✓±15	X	X	X	✓	✓[d]
(16) Fang et al. (2004)	China: Zhejiang Coast (150)	Collected in May 1998 from stations across study region	X	X	✓ within 5–10	✓	✓	X	X	✓[d]
(17) Khaled (2004)	Egypt: El-Mex Bay (12)	Purchased January 2003	✓ [a] NP	✓ [a] NP	✓	X	X	✓	✓	✓[d]
(18) Yilmaz (2005)	Turkey: Iskenderun Bay (45)	Gill net at three sites during autumn 1999	✓ [a] 179–228	✓ [a] 67–74 ± 4	✓ NP	X	X	X	X	✓[c] and [d]
(19) Cogun et al. (2006)	Turkey: Karatas Coast, NE Mediterranean (40)	Seine net all seasons 2003	✓ [a] 230 ± 3	✓ [a] 120 ± 8	✓ within 10	X	X	✓	✓	✓[d]
(20) Dural et al. (2006)	Turkey: Camlik Lagoon (tidal) (44)	Purchase seasonally in 2005	✓ [a] 135–281	✓ [a] 40–194	✓ NP	X	X	✓	✓	✓[d]
(21) Has-Schon et al. (2006)	Croatia: River Neretva (12)	Collected between July and October 2003	X	X	✓ checked annually	X	X	X	X	✓[d]
(22) Liu et al. (2006)	Taiwan: Anpin harbour (12)	Purchased in December 2001–2002	X	X	✓ CV, 5	X	X	X	X	✓[d]
(23) Turkmen et al. (2006)	Turkey: Iskenderun Bay (45)	Gill net	✓ [a] Ave 256 ± 23.5	✓ [a] Ave 175 ± 36	✓ 93–100	X	X	X	X	✓[d]
(24) Authman and Abbas (2007)	Egypt: Lake Qarun (100)	Purchased March 2004–February 2005	✓ [a] Ave 224	✓ [a] Ave 183	X	✓	X	✓	✓	✓[d]
(25) Chouba et al.	Tunisia: Ghar El Melh Lagoon	Purchased seasonally in 2004	✓ [a] 250–	✓ [a] Ave	✓	X	X	X	✓	✓[d]

(*continued on next page*)

Table 1 (*continued*)

Author	Continent/country/habitat (N)	Sampling design and Collection methods	Length (mm, TL)	Weight (g)	QAQC	Environment				
					Laboratory (% CRM)	Water	Sediment	Fish Tissue		
								Gills	Liver	Muscle
(2007)	(60)		300	14.3 ± 2.2						
(26) Dural et al. (2007)	Turkey: Tuzla Lagoon (76)	Purchased seasonally December 2000–November 2001	✓[a] 145–271	✓[a] 36–186	✓ 96–106	X	X	✓	✓	✓[d]
(27) Ruelas-Inzunza and Paez-Osuna (2007)	Sth America: (a) Guaymas Lagoon, East Gulf of California (3) (b) Altata-Ensendada del Pabellion Lagoon (3)	Purchased April 1998–January 1999	✓[a] 290–410	✓[a] 256–580	✓ 90–115	X	X	✓	✓	✓[d]
(28) Uluozlu et al. (2007)	Turkey: Black Sea (6)[e]	Purchased in 2005	X	X	✓ 95–101	X	X	X	X	✓[d]
(29) Huang et al. (2008)	Taiwan: Reservoir near Tainan City (5)	Collected between September and October 2003	✓[a] 235–40, FL	✓[a] 176–930	✓ within 5	X	X	X	X	✓[d]
(30) Prasath and Khan (2008)	India: Poompuhar Coast (NP) (a) Pre-tsunami (b) Post-tsunami	Water, sediment and fish collected in November 2004 and January 2005 (pre-and post-tsunami)	✓[a] 90–270	X	X	✓	✓	X	X	✓[d]
(31) Uysal et al. (2008)	Turkey: Beymelek Lagoon (3)	Collected November 2006	✓[a] Ave 299 ± 10.9	✓[a] Ave 261 ± 30	✓ NP	X	X	✓	X	✓[d]
(32) Bahnasawy et al. (2009)	Egypt: Lake Manzala (NP)	Collected from study sites	X	X	X	X	X	✓	X	✓[c] and [d]
(33) Boubonari et al. (2009)	Turkey: Monolimni Lagoon (12)	Nets seasonally May 1998–May 1999	X	X	X CV, 1–19	✓	✓	X	X	✓[d]
(34) Padmini et al. (2009)	India: Ennore Estuary (216)[e]	Traps, monthly between October 2004 and September 2006	✓[a] 320–350	X	X	X	X	X	✓	✓[d]
(35) Tapia et al. (2009)	Sth America: (a) Maula River (46) (b) Mataquito River (46)	Purchased winter 2005 and spring 2005, autumn 2006 and summer 2006	X	X	✓ 92–109	X	X	X	X	✓[d]
(36) Tuzen (2009)	Turkey: Black Sea (4)	Purchased during 2008	X	X	✓ NP	X	X	X	X	✓[d]
(37) Yilmaz (2009)	Turkey: Koycegiz Lake-Mugla (44)	Collected June 2005–May 2006	✓[a] Ave 243 ± 27.8	✓ Ave 85.3 ± 12	✓ 92–107	X	X	✓	✓	✓[d]
(38) Frias-Espericueta et al. (2010)		Purchased between August 2004 and June 2005	✓ NP	X	✓ 91–106	X	X	✓	✓	✓[d]
(39) Ruelas-Inzunza et al. (2010)	Sth America: Gulf of California, Mexico (4)	Purchased from June 2003 and March 2004	✓[a] Ave 269 NP	✓ Ave 193±52	✓ NP	X	X	✓	✓	✓[d]
(40) Frias-Espericueta et al. (2011)	Sth America: Coastal lagoons, Mexico (15)	Purchased – March 2006–January 2007	X	X	✓ 91–106	X	X	✓	✓	✓[d]
(41) Karouna-Renier et al. (2011)	Nth America: estuaries in Florida	Collected spring 2004 and 2005 using electrofishing and cast nets	✓ NP	X	✓ 75–120	X	X	X	X	✓[c]
(42) Ambedkar and Muniyan (2011)	India: Vella River, Tamil Nadu (na)	Purchased from January to June 2010	X	X	X	X	X	X	X	✓[d]
(43) Priya et al. (2011)	India: (12) (a) Pulicat Lake (b) Barmouth (opening to sea)	Purchased May 2009	X	X	✓ 95–100	✓	✓	✓	✓	✓[d]
(44) Sacan and Altun (2011)	Turkey: Kuckukekmece Lagoon (24)	Purchased seasonally, winter 2002 to autumn 2003	✓[b] 170–340	✓[b] 263–543	✓ 82–94	X	X	X	X	✓[d]
(45) Waltham et al. (2011)	Australia: Moreton Bay: (a) Marina (20) (b) Estuaries (20) (c) Natural (20)	Collected from each habitat in Winter 2006	✓[b] 170–340	X	✓ DGT 90; sediment 40–100; muscle 85–115	✓	✓	✓	✓	✓[c]

	(d) Canals (20)									
	(e) Lakes (20)									
(46) Djedjibegovic et al. (2012)	Bosnia/Herzegovina: Neretva River (7)	Collected by angling October–November 2003	✓ [a] Ave 289	✓ [a] Ave 192	✓ 90–96	X	X	X	✓	✓[d]
(47) Khoshnood et al. (2012)	Iran: Normoz Strait, Oman Sea and Persian Gulf (24)	Purchased monthly from April 2009–March 2010	✓ [a] Ave 173 ± 22	✓ [a] Ave 480 ± 300	✓ 95–130	X	X	✓	✓	✓[d]
(48) Medeiros et al. (2012)	Sth America: Coast of Rio de Janeiro State (5)	Purchased April–July 2009	X	X	✓ 70–120	X	X	X	X	✓[d]
(49) Mohan et al. (2012)	India: Cochin, Vembanad Lake (na)	Purchased during pre-monsoon season	✓ [a] Ave 234 ± 20	✓ [a] Ave 68–131 ± 2–13	✓ 96–103	X	X	✓	✓	✓[d]
Percentage of investigations reporting details			65	51	65	18	14	44	47	100

N, number of fish examined shown in habitat column.
X, no data.
NP, not presented.
TL, total length of fish.
FL, folk length of fish.
CV, coefficient of variance (ratio of standard deviation to the mean in a string of results).
[a] Relationship between fish tissue metal concentrations and length or weight not examined.
[b] Length and weight relationship with fish tissue metal concentrations examined.
[c] Skin removed.
[d] Skin (presumably) not removed before muscle before analysis.
[e] Approximate calculation of fish number.

patterns and condition tolerances, all of which conceivably confound results. Many studies included fish size and weight measurement (e.g. range or average measurement), but these data were often underutilised and in many cases were presented simply as a range, with no further examination for correlation with metal concentrations. Such morphological measurements have been used successfully to calculate a fish condition factor as a surrogate for fish health. This calculation is based on a length–weight relationship with higher condition factor evidence that fish are healthier compared to fish with a low factor. In a novel application of this calculation, Bervoets and Blust (2003) developed threshold metal concentration values for a gudgeon species above which resulted in a low condition factor, therefore reflecting low fish health, while below the threshold fish were considered healthy. Development of similar metal thresholds for *M. cephalus* would help managers evaluate exposure risks to fish, and indirectly to humans, in response to a proposed urban development and land use change which contributes to increased contaminant runoff.

Biomagnification and bioaccumulation of contaminants in fish continues to be of interest to scientists and managers (Bervoets et al., 2005), and follows the trend of accumulation of contaminants in fish as they become larger/older. In the few studies that examined effect of fish length and weight on fish metal concentrations, no relationships were found and therefore no adjustment for size or weight was made. While this may be the case in the studies examined in this review, examination of fish size and weight should remain in future biomonitor studies in order to decipher implications on tissue metal concentrations more broadly in coastal regions.

The reproductive stage of fish can influence metal concentrations, but is rarely included in biomonitor studies (Yilmaz, 2003). In situations where fish are well advanced through the reproductive cycle, it is expected that metals accumulate in the sexual organs of fish (i.e. gonads and ovaries); a response to higher metabolic activities than during other times of the cycle (Karadede and Unlu, 2000). Accumulation of metals during critical phases of fish embryotic development will not only cause concentrations in the organs of fish to increase, but some metal species are known to have deleterious implications on embryotic development (Jones and Reynolds, 1997). In the few studies to consider reproductive organs, the gonads had higher concentrations than gills and muscle, though not than skin tissue (Yilmaz, 2003).

3.2.2. Implications of including skin in edible tissue analysis

The skin of fish contains lipids that provide an important defence against bacteria, parasites and infection, but the skin can also effectively store contaminants and is often consumed by humans (Lewis, 1970; Yilmaz, 2003). Several studies (10%) specifically examined the skinless edible muscle tissue of fish, while for the remaining studies it was not clearly presented whether edible fillets were analysed with or without the attached skin. Four studies examined metal concentrations in both the skin and skinless edible muscle tissue, and found that the skin contained between 20% and 100% higher metal concentrations than in corresponding skinless edible muscle tissue (Fig. 1). This pattern of higher contaminants in fish skin compared to skinless muscle tissue supports finding in other coastal fish species (e.g. Al-Yousuf et al., 2000; Afonso et al., 2007; Yilmaz and Dogan, 2008). Mindful that fish biomonitor studies typically focus on evaluating safe levels of consumption by humans, examining edible muscle tissue with skin represents the most accurate assessment of human exposure and risk: the skin is more contaminated compared to tissue but small in volume. We are not recommending the analysis of skin-attached over skinless muscle tissue, but recommend that research state whether edible muscle tissue results included skin.

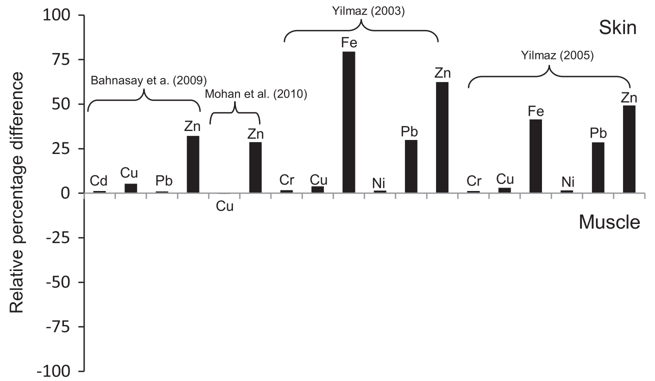

Fig. 1. Relative percentage difference in mean ($N = 3–15$ individual specimens) metal concentrations measured in the skin and skinless edible muscle tissue of the *Mugil cephalus* in published studies. No adjustment was made to standardise tissue weight (e.g. μg/g dry weight, or μg/g wet weight), the focus is on measured differences between tissue types in the same fish.

3.3. Statistical design and quality control

Many studies (65%) included a step to establish analytical precision and repeatability in the laboratory analysis, with the precision achieved seemingly adequate. However, a fundamental drawback in all studies was in the level of attention given to field quality control measures. These were often inadequate for water and sediment sampling, compared with measurement of fish tissue where sub-sample of fish tissues were analysed as part of quality control. Water and sediment contaminant concentrations are known to change over many spatial and temporal dimensions (*sensu* Simpson and Batley, 2007; Mohammad et al., 2012). Without understanding the scale and extent of site heterogeneity, it is difficult and misleading to attempt correlations between water and sediment contaminants measured in fish tissue without additional spatiotemporal data. Any correlation is confounded because of over- or under-estimation in environmental data. The only study to attempt to understand field heterogeneity was Waltham et al. (2011). The authors in that study included a composite sediment sampling technique (three replicate grabs which were homogenised with a subsample taken for analysis) at each site and the use of time-integrated passive samplers (DGT – diffusive gradient in thin film) to evaluate environmental variability. Such a rigorous sampling program requires additional funding for analysis and sample collection, however, the financial imposition is overshadowed by the gains afforded with understanding laboratory and sample variance. An auxiliary approach may include examination of historical datasets, which equally achieve greater insight into spatial and temporal contaminant concentrations, but at a reduced cost.

Table 2
Mean water and sediment contaminant concentrations measured in each study. Concentrations in bold exceed the recommended (usually 95%) trigger value for marine waters and sediment low trigger values (ANZECC/ARMCANZ, 2000): (a) is the impact site and (b) is the reference site for a study. (ND) not detected. In study 45: (a) marina, (b) estuary, (c) open bay natural wetland, (d) artificial residential canal, and (e) artificial residential lake.

Study	As	Cd	Cu	Hg	Pb	Zn
Water (μg/L)						
7a	–	–	**17.5**	–	–	**1300**
7b	–	–	ND	–	–	**17.6**
16	6.8	0.05	**1.5**	0.02	0.7	**16.8**
24	–	–	**2.1**	–	–	0.1
33	–	0.1	**10**	–	**25.1**	**48.8**
43a	–	**0.36**	**9.2**	–	**5.8**	**32.5**
43b	–	0.18	**5.4**	–	2.2	**25.1**
45a	–	0.02	**2.3**	–	0.02	**23.7**
45b	–	0.01	**1.4**	–	0.02	**20.1**
45c	–	0.01	**1.4**	–	0.02	10.9
45d	–	0.02	0.8	–	0.02	**27**
45e	–	0.02	0.9	–	0.04	**22.3**
Trigger value	–	0.2	1.3	0.1	4.4	15
Sediment (μg/g dry weight)						
11	–	–	26.7	–	–	126.7
12[a]	13	–	14.8	–	20	103
16	8.6	–	29.4	0.06	23.9	80.6
30a	–	–	31.4	–	–	5.9
30b	–	–	47.9	–	–	7.3
33	–	0.2	60	–	**90**	70
43a	–	1.2	31.7	–	10.3	76.1
43b	–	0.9	26.4	–	8.3	63.4
45a	2.1	0.1	31.5	–	14.9	71.5
45b	1.9	0.1	7.1	–	8.6	47.5
45c	2.4	0.1	5.8	–	5	32.2
45d	3.5	0.1	12.2	–	48.4	105.1
45e	2.6	0.1	10.6	–	39.8	76.9
Trigger value	20	1.5	65	–	50	200

(–) No analysis.
[a] Results extracted from a graph, compared to all other studies which presented data in tables.

3.4. M. cephalus contaminant concentrations – global comparison

3.4.1. Environmental variables

Very few studies inspected metal concentrations in the water and sediment at the same locations where fish were collected (Table 2). Of the 49 studies, 18% and 14% examined either water or sediment contaminants at sites respectively, while only 10% of studies examined both water and sediment. This low number of studies is probably a consequence of the purchase of fish from commercial fisherman. A secondary reason is probably a reflection of a greater focus on examining human health exposure risks, rather than understanding the mechanisms of contaminant uptake.

Some water and sediment anomalies are presented, in particular, Cu, Pb and Zn which exceed the ANZECC/ARMCANZ (2000) 95% level of aquatic ecosystem protection guidelines (Table 2). Sources of Cu, Pb and Zn in coastal waters are many and varied, such as vehicle road runoff, discharge from smelting and industrial facilities, sewage treatment plants and antifouling paints on vessels, and determining a single source among many would be difficult (Lee et al., 2006; Waltham et al., 2011), unless a comprehensive monitoring program was undertaken (Dunn et al., 2007; Chen et al., 2012; Reese et al., 2012).

Sultana and Rao (1998) recorded the highest water metal concentrations in the impact location. For sediments, the only exceedance was Pb recorded in Monolimni Lagoon, Turkey, by Boubonari et al. (2009). Curiously, only a single study focused on the <63 μm fine sediment fraction, with the remaining studies not offering the size faction examined. The <63 μm sediment fraction is known to yield higher contaminants, because of the greater total surface area of smaller sediment particles, compared to much larger sediment size fractions which have a smaller relative surface area, and is therefore representative of the labile potential within sediments (ANZECC/ARMCANZ, 2000).

3.4.2. Mechanisms of contaminant uptake

In studies that examine the pathway of uptake of contaminants, accumulation of contaminants in the gills of fish is believed to be associated with available concentrations in the water column. Conversely, uptake of contaminants in food or ingested material, including sediment in the case of *M. cephalus*, is reflected by high metal concentrations in the liver (Heath, 1995). This distribution of metals is a response to primary source of contaminants; water or food. This pattern of contaminant distribution in fish has been widely demonstrated (e.g. Al-Yousuf et al., 2000; Alquezar et al.,

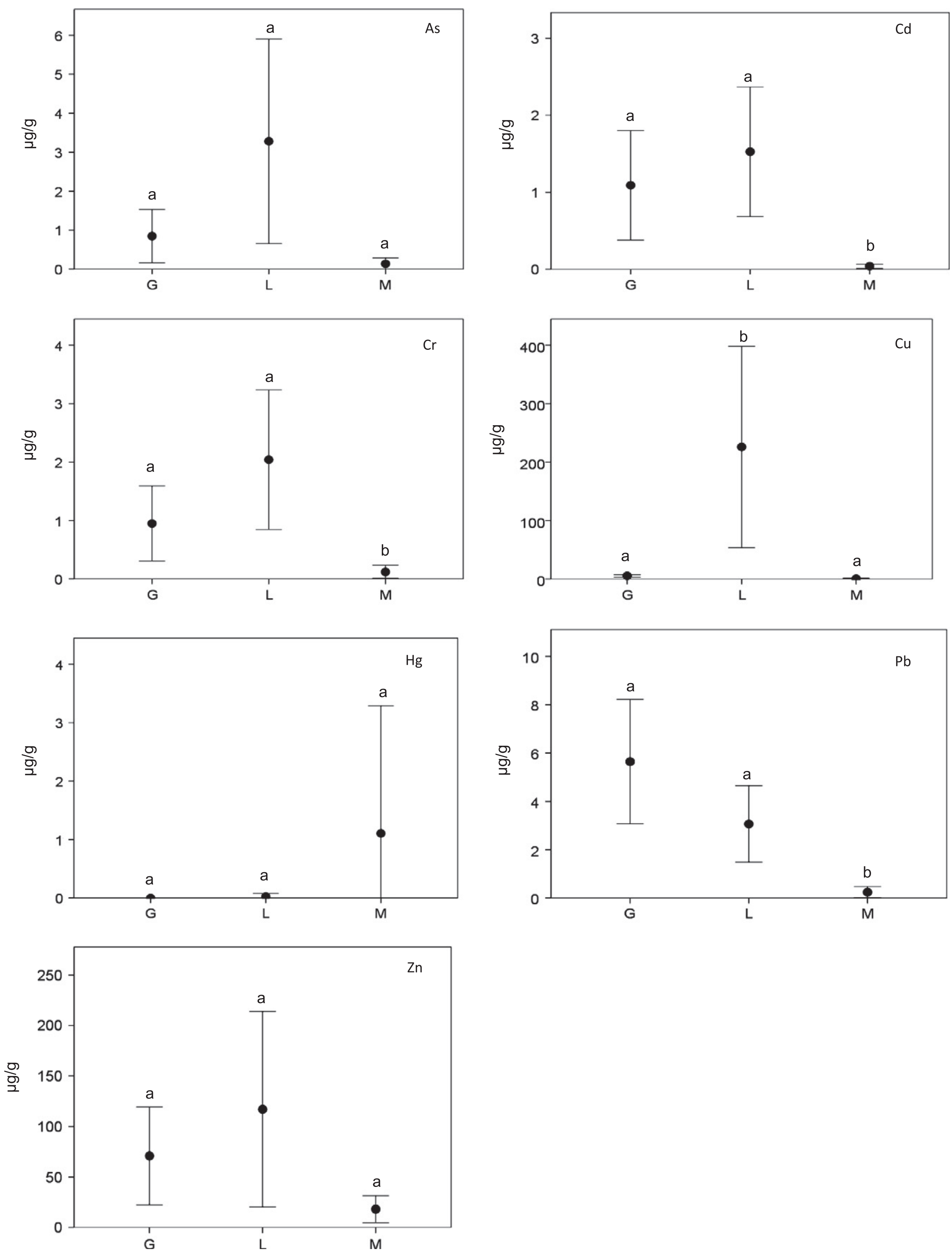

Fig. 2. Mean (μg/g) and standard error (±) for measured metal concentrations in (G) gills ($N = 27$), (L) liver (36); and (M) muscle tissue (61) for all studies pooled. Values for each tissue with different lower case letters differ significantly between according to Tukey's post hoc test ($p < 0.01$).

Table 3
Mean contaminant concentrations in edible muscle tissue for *Mugil cephalus*. Reported for those elements where a guideline is available (ANZFA, 1999). (a) is the impact site, while (b) is the reference site. For study 45 same as Table 2. Numbers in bold exceed the ANZFA (1999) guideline.

Study	As	Cd	Cu	Hg	Pb	Zn
1	0.5	0.5	1.9	0.1	–	17
2	–	0.1	3.2	–	–	53.2
3	0.8	0.04	0.59	0.33	**0.65**	5.3
4	–	0.22	1.71	–	**1.92**	22.3
5	–	0.24	1.46	**1.91**	–	17.4
6	–	0.28	**19.12**	–	–	180
7a	–	0.36	1.82	–	**3.24**	23.19
7b	–	0.16	0.5	–	**1.22**	6.56
8	–	0.96	4.48	–	**6.25**	26.1
9	**4.7**	–	–	–	–	–
10	–	–	8.5	–	–	78.9
11a	–	0.05	**20**	–	–	30
11b	–	0.01	5	–	–	10
12	0.011	–	0.001	0.00005	–	0.01
13	–	0.66	4.41	–	**5.32**	37.39
14	–	–	0.006	–	0.03	0.16
15	0.002	0.001	0.002	0.001	–	0.002
16	2.8	0.26	3.12	0.07	0.4	35.8
17	–	1.88	4.88	–	**2.92**	27
18	–	–	3.48	–	**26.72**	132.4
19	–	1.5	9.8	–	**6.3**	24.2
20	–	0.07	–	–	–	88.9
21	0.0001	0.0001	–	0.0002	0.0001	–
22	**4.18**	–	–	–	–	–
23	–	0.0003	0.001	–	0.002	0.005
24a	–	–	–	–	–	0.002
24b	–	–	–	–	–	0.002
25	–	0.02	–	0.34	0.2	–
26	–	0.09	0.53	–	**0.92**	39.6
27a	–	0.17	0.98	–	**0.5**	22
27a	–	0.3	3.4	–	**1.0**	18.4
28	–	0.45	1.26	–	**0.61**	40.2
29	–	–	–	0.4	–	–
30a	–	–	**20.5**	–	–	156.8
30b	–	–	14.4	–	–	142.1
31	–	–	0.001	–	–	0.006
32	–	1.74	5.75	–	**2.75**	29.62
33	–	0.6	**50.0**	–	**3.0**	**220**
34	–	0.006	0.005	–	0.001	0.0001
35a	–	–	0.006	–	0.004	–
35b	–	–	0.006	–	0.003	–
36	0.23	0.35	2.14	**70**	**0.68**	86.2
37	–	0.48	**25.4**	–	**1.72**	**394.4**
38	–	0.31	1.3	–	**2.62**	10.78
39	–	0.1	1.0	–	**1.1**	–
40	–	0.17	1.06	–	**1.63**	21.77
41	–	–	–	0.01	–	–
42	–	0.0006	0.0005	–	0.0004	0.0003
43a	–	0.55	1.73	–	**10.75**	8.25
43b	–	1.74	5.75	–	**2.75**	29.62
44	–	0.02	1.11	–	0.43	21.7
45a	0.004	0.0001	0.005	0.0001	0.0001	0.03
45b	0.003	0.0001	0.006	0.0001	0.0001	0.02
45c	0.003	0.0001	0.005	0.0001	0.0001	0.02
45d	0.003	0.0001	0.002	0.0001	0.0001	0.03
45e	0.002	0.0001	0.007	0.0001	0.0001	0.02
46	–	0.0001	0.001	0.002	0.0001	–
47	0.0001	0.0001	0.001	0.0001	0.0001	0.003
48	0.002	0.0001	0.005	–	0.0008	0.02
49	–	–	–	–	–	41.07
Food standard	2.0#	2.0	10.0	0.5	0.5	200

(ND) not detected.
(–) No analysis.
Inorganic As.

2006; Ambedkar and Muniyan, 2011), but most elegantly in manipulated experiments where fish were exposed to a combination of contaminated water and clean food (i.e. gills > liver), and clean water and spiked food (i.e. gills < liver; Kraal et al., 1995). The use of in situ passive samplers (DGT) for a field calibration of oyster biomonitors has also been described (Jordan et al., 2008), and this approach may be useful with fish biomonitors too.

We pooled the data in all studies to examine the metal content in gill, liver and muscle tissue (standardised to μg/g dry weight) for those routinely investigated metal elements (Fig. 2). The results show an interesting distribution, where As, Hg and Zn had no measurable difference in concentrations among the tissues (one-way ANOVA, $df_{2,125}$ $F = 0.5–18.1$, $P = 0.1–0.59$), while Co, Cr and Pb had significantly higher concentrations in gills and liver, compared to muscle tissue ($df_{2,125}$ $F = 9.5–28.1$, $P < 0.01$). In contrast, Cu had a different metal distribution to the other elements, with significantly higher concentrations in liver compared to gills and muscle tissue ($df_{2,125}$ $F = 44.6$, $P < 0.01$). Accumulation of metals in the gills and liver tissue at concentrations higher than muscle tissue seems typical for fish within this family. Several studies examined the pathway uptake of contaminants in *M. cephalus* in comparison to carnivorous and herbivorous fish species, that typically occupy the middle to upper water column, and concluded that the significantly higher concentrations in *M. cephalus* reflects the benthic feeding strategy; it is continually sifting through benthic sediments for food which not only increases exposure to contaminated sediments, but also liberates metals from sediments into the water column after disturbance (Yilmaz, 2005; Dural et al., 2006; Waltham et al., 2011). For this species, there is strong evidence that it can regulate and reduce metals after uptake, and this is demonstrated with higher concentrations in liver over gills for some elements, and vice versa for other metals. The tendency for *M. cephalus* to accumulate highest contaminants in the gills and liver, however, does not conclusively demonstrate whether direct or indirect uptake pathways are operating. These accumulation mechanisms underpinning uptake of contaminants could be investigated with the use of chemical tracers (Xu and Wang, 2002).

3.4.3. Human health risks

M. cephalus was reported to be safe for human consumption in 65% or 32 coastal regions studied, with no elevated metal concentrations above the presented safe seafood standard (Table 3). However, this also means that the remaining coastal regions had at least one metal element that exceeded guidelines, and therefore the fish were potentially unsafe for human consumption. Of all the elements, Cd concentrations always complied with the food guidelines and posed no risk to human health. For Hg and Zn, most studies presented concentrations that complied with food guidelines, with elevated concentrations found in a few studies, from different regions. In the case of Cu, tissue concentrations exceeded the guidelines in several locations, highlighting potential health concerns and also the need to further investigate sources of Cu in those waterways. Arsenic concentrations were detected above maximum permitted concentrations for human health at several localities, however, the maximum permissible concentration in this standard is based on inorganic As, which typically represents only a small proportion of the total As measured in marine fauna (Edmonds and Francesconi, 1993). It is likely that the method of analysis in each study did not discriminate between inorganic and organic forms of As, therefore we cannot confirm whether these exceeded concentrations in fact satisfy the standard. Lead was the element that most commonly exceeded the guideline, with tissue concentrations 50 times above the safe limit observed. Sources of Pb in the coastal zone are widespread, including historic or current use of leaded fuel, waste refuge areas, or even lead from recreational fishing sinkers; Pb is a very ubiquitous contaminant that is known to associate strongly with particulate matter (Luoma and Rainbow, 2008 and the references therein). The dominant route of uptake is therefore likely to be through consumption of particles or sediment, rather than through uptake from water. Trophic transfer of Pb is considered to be low except in highly contaminated sites (Luoma and Rainbow, 2008 and the references therein).

Overall, *M. cephalus* collected in Turkey (see Boubonari et al., 2009; Yilmaz, 2009) contained the highest muscle concentrations of Cu, Pb and Zn, which also exceeded the food guidelines. Tuzen (2009) also completed a study in Turkey and recorded the highest Hg concentrations of any study using *M. cephalus*, demonstrating the likelihood of human health risks if consumed: note that Boubonari et al. (2009) and Yilmaz (2009) did not include Hg in the suite of testing, so it is unknown whether Hg is an environmental problem more broadly, supporting the underlying theme of this review that inconsistencies exist among studies, especially at a local scale. For managers in Turkey, there is an apparent need to firstly identify bulk sources of contaminants, and to then implement an appropriate program of source control and on-going maintenance.

4. Conclusions and recommendations

This review demonstrates a very widespread world-wide use of *M. cephalus* in biomonitoring studies in the coastal zone and considers the potential to develop upon this further for biomonitor studies with fish. These studies all have an underlying objective of determining fish health and potential exposure risk to humans consuming *M. cephalus*, which is particularly important given the high rates of consumption of this species. There were subtle differences in the methodology and data collected which highlights the challenge in achieving a completely balanced assessment and benchmarking exercise among studies. This review has demonstrated that *M. cephalus* in many study regions have low uptake of contaminants and at this stage, *M. cephalus* in many coastal locations comply with the Australian and New Zealand Food Safety Standard. The results, however, provide evidence that some coastal regions support fish that are unsafe to eat, a response presumably to uncontrolled and unregulated waste disposal into coastal wetlands.

The wide geographical distribution of this fish species assists with benchmarking contaminant concentrations in local populations compared to populations elsewhere, but to achieve this requires comparable data which, on the basis of this review, could be done better. To achieve this for *M. cephalus*, the following recommendations are necessary in future studies:

(1) Sample and laboratory methodology, and data must be clear with no ambiguity reported leading to confusion and criticism concerning quality of the data.
(2) Laboratory and field quality assurance and quality control testing should be sufficient to improve confidence in the data presented and underlying study conclusions.
(3) Assessment of morphological measurements, including fish condition and health assessment, and implication of skin in the assessment of edible fish muscle analysis.
(4) Examination of water and sediment samples at sites to provide a clear understanding of the extent of contamination, and to assist with examination of uptake; and
(5) Inclusion of gill and liver, along with muscle, tissue analysis to examine the underlying mechanisms of contaminant uptake and accumulation in fish species.

Examination of contaminant accumulation in marine organisms is becoming increasingly important in coastal monitoring programs in response to assessment of bioaccumulation of anthropogenic waste discharged and examination of the exposure risks to humans consuming seafood sourced from coastal waters (McKinley and Johnston, 2010; Gall et al., 2012). The development of comparable datasets is critical to the success of biomonitoring programs, so that meaningful comparisons can be made, particularly in response to restoration efforts implemented by managers. Our recommendations are important for future biomonitor studies involving *M. cephalus*, and extend to other coastal fish species more broadly.

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