Original Article

Species- and size-related patterns in stable isotopes and mercury concentrations in fish help refine marine ecosystem indicators and provide evidence for distinct management units for hake in the Northeast Atlantic

T. Chouvelon[1]*, F. Caurant[1], Y. Cherel[2], B. Simon-Bouhet[1], J. Spitz[1,3], and P. Bustamante[1]

[1]*Littoral Environnement et Sociétés, UMR 7266 CNRS – Université de La Rochelle, 2 rue Olympe de Gouges, 17000 La Rochelle, France*
[2]*Centre d'Etudes Biologiques de Chizé, UPR 1934 du CNRS, BP 14, 79360 Villiers-en-Bois, France*
[3]*Marine Mammal Research Unit, Fisheries Centre, University of British Columbia, 2202 Main Mall, Vancouver, British Columbia V6T 1Z4, Canada*

Corresponding author: tel: +33 546507625; fax: +33 546456284; e-mail: tiphaine.chouvelon@hotmail.fr

Recent European environmental policies argue for the development of indicators of the ecological status of ecosystems that are easy to implement and powerful enough to detect changes quickly. For instance, some indicators that are currently proposed for monitoring foodweb structure and functioning are based on the size of organisms, using size as a proxy for trophic level. However, these indicators do not necessarily accurately reflect the underlying trophic structure and dynamics to follow. Ecological tracers (i.e. chemical parameters measured in consumer tissues to infer the trophic ecology of organisms) may serve as complementary indicators of trophic level, and may also help distinguish different populations of a species when more commonly used methods (e.g. genetic, age determination) present their own limitations. Here, we analysed the potential of muscle $\delta^{13}C$ and $\delta^{15}N$ values and of muscle mercury (Hg) concentrations to depict size-related trophic habits of different fish species. We expected that intra- and interspecific variation in these ecological tracers could be helpful in refining currently proposed indicators of marine ecosystems, and also help in discriminating management units for some species. Four fish species were selected for their economical and/or ecological importance in the Bay of Biscay (Northeast Atlantic): whiting *Merlangius merlangus*, European hake *Merluccius merluccius*, Atlantic horse mackerel *Trachurus trachurus* and anglerfish *Lophius piscatorius*. Muscle $\delta^{13}C$ and $\delta^{15}N$ values segregated the species and enabled us to discriminate species-specific feeding strategies with increasing size of individuals. Fish body size was not always linearly correlated with $\delta^{15}N$ or trophic level. In contrast, Hg concentrations and size-related Hg patterns were more similar from species to species. Interestingly, muscle $\delta^{15}N$ values together with Hg concentrations segregated the two putative stocks of European hake within the Bay of Biscay. Hence, we propose the combined use of ecological tracers as a promising method for accurately assigning a consumer to a trophic guild, investigating Hg trophodynamics in foodwebs, and helping discriminate distinct ecological populations within a species when defining management units.

Introduction

Trophic linkages between organisms take a central place in the general structure and functioning of ecosystems (e.g. Lindeman, 1942; Odum, 1969; Cury *et al.*, 2003). Recently, the European Union has adopted the Marine Strategy Framework Directive (MSFD) that aims at implementing an ecosystem-based management of European marine ecosystems (European Commission, 2008). The descriptor "Foodwebs" is one of the 11 qualitative descriptors proposed to define and to monitor the "good environmental status" of these ecosystems. Within this descriptor, the

three following indicators are proposed: (i) productivity/performance of key predator species using their productivity per unit biomass; (ii) proportion of large fish (by mass) or Large Fish Indicator (LFI); (iii) abundance trends of functionally important selected groups/species (European Commission, 2010; Rombouts et al., 2013). However, the relevance of these indicators is discussed for some geographical areas and countries affected by the MSFD (e.g. for the Bay of Biscay area in the Northeast Atlantic, see Borja et al., 2011; more generally, see Rombouts et al., 2013). They effectively present significant general limitations to their applicability for depicting the ecological status of foodwebs. For instance, the LFI [indicator (ii)] is based on the common assumption that fish body size is related to fish trophic level, so that changes in the size-spectrum of a fish community should reflect changes in its trophic structure. The LFI was primarily developed to assess the effect of fishing on the fish community (e.g. Shin et al., 2005; Greenstreet et al., 2011), as many fisheries target the largest fish considered as *a priori* highest trophic level organisms within a given ecosystem. However, fish body size is not always well correlated with trophic level (e.g. Jennings et al., 2001, 2002). This limits the use of fish size as a proxy for fish trophic level and consequently the use of the LFI as a reliable indicator to assess foodweb status over (Rombouts et al., 2013).

In recent decades, stable isotope analysis (SIA) of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) in consumers' tissues has proven to be a powerful tool for describing the trophic ecology of organisms (e.g. Kelly, 2000; Bearhop et al., 2004; Michener and Kaufman, 2007). Because the difference in $\delta^{13}C$ between a source and its consumer (i.e. between two trophic levels) is generally low (≤ 1‰; De Niro and Epstein, 1978; Post 2002), $\delta^{13}C$ values have often been considered to be a powerful tracer of the primary producer at the base of the foodweb supporting consumers, and consequently as a tracer of their foraging habitat (e.g. France, 1995; Hobson, 1999; Cherel and Hobson, 2007). Alternatively, consumers generally present 2.5–5‰ higher $\delta^{15}N$ values relative to their sources (De Niro and Epstein, 1981; Vanderklift and Ponsard, 2003), so $\delta^{15}N$ values have been widely used as a proxy for their trophic position or to calculate their trophic level (e.g. Vander Zanden et al., 1997; Post, 2002; Cherel et al., 2010; Navarro et al., 2011). Temporal changes in $\delta^{15}N$ values (or derived-trophic levels) have then already been used as indicators of the responses of biological communities to environmental changes (e.g. for the North Sea fish community under fishing pressure, based on the same assumption as for the LFI indicator commented above; Jennings et al., 2002; Nicholson and Jennings, 2004; Greenstreet and Rogers, 2006). The isotopic compositions of organisms (notably $\delta^{15}N$ values) have also proved to be powerful tools for understanding pollutants bioaccumulation and pathways throughout foodwebs (e.g. Cabana and Rasmussen, 1994). Naturally occurring stable isotopes have even been able to disentangle individuals' trophic positions, feeding habits and foraging habitats exploited for explaining trace metal concentrations such as mercury (Hg) in top predators (e.g. Cossa et al., 2012; Ramos et al., 2013).

Hg is a non-essential metal released into the environment from both natural and anthropogenic sources, reaching the ocean through river inputs and atmospheric depositions (Fitzgerald et al., 2007). Trophic transfer is then the main pathway for the intake of Hg by marine organisms, bioaccumulating in higher trophic level consumers (Eisler, 1987; Boening, 2000; Fitzgerald et al., 2007). As for $\delta^{15}N$ values, Hg may potentially be another good tracer of the trophic level of aquatic organisms (e.g. Cabana

and Rasmussen, 1994), given its proven biomagnification within foodwebs (Gray, 2002). Due to methylation processes by microorganisms in marine sediments on the shelf and in the low oxygen zone of oceanic waters (Bacci, 1989; Fitzgerald et al., 2007), enhanced Hg concentrations in biota from meso-pelagic and deep-water environments have been already highlighted in several ecosystems (Monteiro et al., 1996; Thompson et al., 1998; Choy et al., 2009; Chouvelon et al., 2012a). However, the age of organisms (or their size, as a proxy of age) remains a major factor driving Hg bioaccumulation, principally in its methylated form (Fitzgerald et al., 2007) in the muscle tissue of numerous marine organisms (e.g. Monteiro and Lopes, 1990; Cronin et al., 1998; Bustamante et al., 2006).

Finally, approaches using ecological tracers (e.g. stable isotopes, trace metals, or fatty acid analysis) can provide information on the diet of individuals on different time-scales, depending on the tracer and its turnover in the analysed tissue. Therefore, in addition to segregating species of different trophic ecology, the combination of these approaches may be used to distinguish a different use of resources and/or habitats by the different populations of the same species, and help then in defining management units. Indeed, management units defined by the ecological tracers or other tools may be more appropriate for response to actual management issues rather than those defined by genetic differences alone, which often provide a picture on a broader spatio-temporal scale (e.g. Taylor and Dizon, 1999; Cope and Punt, 2009). When compared with genetic data, other tools providing biological or ecological information at the individual scale (e.g. tagging/mark–recapture data) can effectively lead to questioning of the validity of the stock assessment units used by the ICES study groups concerning exploited fish species (e.g. Fritsch et al., 2007; Pawson et al., 2007 for the European sea bass *Dicentrarchus labrax* in the Northeast Atlantic). For the European hake *Merluccius merluccius* in the Northeast Atlantic in particular, the ICES recognizes the existence of two putative stocks since the 1970s: a northern stock extending from the northern Bay of Biscay to the west of Norway, and a southern stock from the southern Bay of Biscay to the Iberian Peninsula (ICES, 2008; Murua, 2010). However, there is still no clear biological (e.g. Piñeiro and Saínza, 2003) or genetic evidence (Roldan et al., 1998; Castillo et al., 2004; Cimmaruta et al., 2005) that the two are distinct populations, and more generally the knowledge of the biology and ecology of the European hake in the Northeast Atlantic is still quite limited (Murua, 2010).

The main objectives of this study were thus threefold: (i) to assess size-related feeding strategies through SIA in four different fish species of economical and ecological importance in the Northeast Atlantic (whiting *Merlangius merlangus*, European hake, Atlantic horse mackerel *Trachurus trachurus* and anglerfish *Lophius piscatorius*); (ii) to evaluate whether body size, trophic position or diet and foraging habitat can explain size-related variations in $\delta^{15}N$ values and total-Hg (T-Hg) concentrations in fish muscle tissue; and (iii) to evaluate the potential of the combined used of SIA and Hg concentrations to help refine current indicators of marine foodwebs and ecosystems and their contribution to the definition of management units for hake in the Northeast Atlantic. Consequently, our three hypothesis were primarily that: (i) fish body size should be related to fish trophic level, although the relationship between both variables may be not always strictly linear; (ii) bioaccumulation of Hg in fish should be driven by size (as a proxy for age), the trophic level of individuals and their foraging habitat; and (iii) the combined analysis of stable isotopes and Hg concentrations can

contribute to elucidating the existence of two putative stocks of hake in the Bay of Biscay.

Material and methods
Selected species
Four fish species were selected according to their phylogeny (i.e. different families), habitat (i.e. coastal vs. offshore area) and vertical distribution in the water column (i.e. pelagic vs. bentho-pelagic and benthic species): whiting and European hake (both belonging to the Gadiform order), Atlantic horse mackerel, and anglerfish. (See species' characteristics summarized in Table 1.) Commercial fisheries target these species in the Northeast Atlantic, notably in the Bay of Biscay (Palomares and Pauly, 2010; Lassalle et al., 2011). They also present different diets, with food items ranging from small planktonic crustaceans to small pelagic or bentho-pelagic fish, and have dietary shifts associated with growth (i.e. juvenile vs. adult fish). Information on the main prey species consumed during ontogenesis by the four selected fishes is detailed in Table 1.

Sampling
A total of 255 individuals covering a large range in size for each of the four species were collected during the annual EVHOE groundfish survey, conducted by the Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER), from the continental shelf to the shelf edge of the Bay of Biscay in the autumns of 2008 and 2009 (Figure 1, Tables 2 and 3). On board, the total length (TL) of each organism was measured, and a piece of dorsal white muscle (without skin) was removed for stable isotope and Hg analyses. Samples were immediately placed in individual plastic bags and frozen at $-20°$C. They were then freeze-dried, ground into a fine powder and stored in individual plastic vials until analysis.

Isotopic and mercury analyses
Because lipids are depleted in ^{13}C relative to other tissue components (De Niro and Epstein, 1977), they were extracted from muscle samples using cyclohexane. Most solvents (e.g. a chloroform–methanol mixture) that are currently used for lipid extraction generally affect δ^{15}N values in an unpredictable way (e.g. Post et al., 2007 and references therein). As an alternative, cyclohexane is a non-chlorinated solvent with low toxicity that can rapidly extract lipids from the tissues of marine organisms (Smedes, 1999). Moreover, internal laboratory experiments investigating a range of marine taxa (including fish) have proven that cyclohexane significantly corrects δ^{13}C values for the lipid effect in originally fatty muscle samples only (i.e. samples with bulk C:N ratio > 3.5; Post et al., 2007), without affecting δ^{15}N values of any samples (Chouvelon et al., unpublished data). For each sample, an aliquot of $\sim$ 100 mg of fine powder was thus agitated with 4 ml of cyclohexane for 1 h. The samples were then centrifuged for 5 min at 4000g, and the supernatant containing lipids was discarded. The samples were finally dried in an oven at 45°C for 48 h, and subsamples of lipid-free powder were weighed in tin cups (0.40 $\pm$ 0.05 mg). SIAs were performed with a Thermo Scientific Flash EA1112 elemental analyser coupled to a Thermo Scientific Delta V Advantage mass spectrometer (CF IR-MS). The results are presented in the usual δ notation relative to the deviation from standards (Vienna Pee Dee Belemnite for δ^{13}C, and atmospheric nitrogen for δ^{15}N), in parts per thousand (‰). Replicate measurements of internal laboratory standards (acetanilide) indicate measurement errors of <0.15‰ for both δ^{13}C and δ^{15}N values.

T-Hg analyses were carried out with an Advanced Mercury Analyser (ALTEC AMA 254) on at least two homogenized dry muscle tissue subsamples (untreated/non-lipid-extracted powder) for each individual. For Hg determination, the metal was evaporated by progressive heating up to 800°C, then held under an oxygen atmosphere for 3 min, and finally amalgamated on a gold net. Afterwards, the net was heated to liberate the collected Hg, which was finally measured by atomic absorption spectrophotometry. Hg analyses were run according to a thorough quality control program including the analysis of a certified reference material (CRM) TORT-2 (lobster hepatopancreas; National Research Council, Canada). CRM aliquots were treated and analysed under the same conditions as the samples. CRM results were in good agreement with the certified values, with an average recovery rate of 95%. The detection limit was 5 ng·g^{-1} dry mass (dm). Hg concentrations in tissues are expressed in ng·g^{-1} dm.

Data treatment
The variability of δ^{13}C and δ^{15}N values measured within each species (and consequently, the isotopic niche width of each species investigated) (Newsome et al., 2007) was first described in terms of the species-specific areas on a δ^{13}C–δ^{15}N biplot that includes all individual values (Figure 2). The species trophic niche width was estimated by: (i) the total area (TA), which is a measure of the area of the polygon delineated by the line drawn through the most extreme datapoints of a species in the δ^{15}N–δ^{13}C space (Layman et al., 2007); and (ii) a sample size-corrected version of the Bayesian estimate of the standard ellipse area (SEAc) proposed by Jackson et al. (2011), as an alternative to the TA metric that tends to increase with sample size. The variability of Hg concentrations measured within each species was represented on a boxplot (Figure 3). Since data did not satisfy conditions for the use of parametric statistics (i.e. normality checked by Shapiro–Wilks tests, and homogeneity of variances checked by Bartlett tests), Kruskal–Wallis (KW) tests followed by post hoc multiple comparison tests with Holm's adjustment method were performed to test general interspecific differences in muscle δ^{13}C and δ^{15}N values and Hg concentrations.

To identify size-related, spatial and/or year-related trends for explaining δ^{15}N variability, Gaussian Generalized Additive Models (GGAMs) were fitted to individual δ^{15}N values (Zuur et al., 2007) using the mgcv package in R (R Development Core Team, 2011). GAMs are non-parametric generalizations of multiple linear regressions; they are particularly helpful in capturing and modelling non-linear relationships (Zuur, 2012). TL of individuals was considered as a continuous explanatory variable in the models, as well as δ^{13}C values. In the Bay of Biscay, δ^{13}C values (and δ^{15}N values) integrate information about the habitat of individuals (i.e. neritic vs. oceanic domain) as they decrease significantly in organisms found further offshore (Chouvelon et al., 2012b; Nerot et al., 2012). Major river plumes of the Bay of Biscay also influence isotopic signatures of the more coastal species (Chouvelon et al., 2012b). Sampling locations of organisms were thus split into two latitudinal zones, the northern area (a priori influence of the Loire river plume) vs. southern area (a priori influence of the Gironde river plume), as generally defined during EVHOE surveys. The latitudinal zone (LZ) of sampling of individuals was then treated as a possible categorical explanatory variable and added as an interaction term when appropriate (i.e. for the species whose sampling was balanced enough for

Table 1. Characteristics of the four fish species from the Bay of Biscay selected for this study (phylogeny, distribution and main dietary items during ontogenesis from literature data—mainly stomach contents data).

Order and family	Species	Common name	Usual depth range[a]	Habitat[a]	General distribution in the water column[a]	Main prey species and feeding habits in the Northeast Atlantic[b]	
						Juveniles	Adults
Gadiform							
Gadidae	*Merlangius merlangus*	Whiting	0–100 m	Coastal/Shelf	Bentho-pelagic	Small crustaceans such as mysids or the shrimps *Crangon* sp.	Bentho-pelagic fish such as the Norway pout *Trisopterus esmarkii*, or pelagic fish such as the sandeel *Ammodytes marinus*, the sprat *Sprattus sprattus* and the herring *Clupea harengus*. Prey size and the proportion of fish in the diet increase with increasing whiting size.
Merlucciidae	*Merluccius merluccius*	European hake	0–400 m	Shelf/Upper slope	Bentho-pelagic	Crustaceans (mainly euphausiids)	Small pelagic fish such as the horse mackerel *Trachurus trachurus*, the anchovy *Engraulis encrasicolus* and the sardine *Sardina pilchardus*, or the herring *C. harengus*; also the bentho-pelagic blue whiting *Micromesistius poutassou* in larger individuals. Fully piscivorous once adult, with prey size increasing with increasing hake size.
Perciform							
Carangidae	*Trachurus trachurus*	Atlantic horse mackerel	0–200 m	Coastal/Shelf	Pelagic	Planktonic crustaceans such as copepods and euphausiids	Planktonic crustaceans such as copepods and euphausiids, also small fish (<10 cm) in larger individuals (>30 cm), such as gobiids or small individuals of *E. encrasicolus*, *Gadiculus argenteus* and *M. poutassou* from the shelf edge. Prey size increasing with increasing horse mackerel size.
Lophiiform							
Lophiidae	*Lophius piscatorius*	Anglerfish	0–1000 m	Shelf/Upper slope	Bentho-pelagic/Benthic	(not documented)	Variety of prey, ranging from bentho-pelagic fish such as *Trisopterus* sp., or even pelagic fish such as the sandeel *A. marinus*, but also cephalopods and large crustaceans such as the lobster *Nephrops norvegicus*.

[a]References: Quéro, 2003; Trenkel *et al.*, 2009; Palomares and Pauly, 2010.
[b]References: whiting (Moore and Moore, 1976; Hislop *et al.*, 1991; Greenstreet *et al.*, 1998; Pedersen, 1999), European hake (Guichet 1995; Velasco and Olaso, 1998; Mahe *et al.*, 2007), Atlantic horse mackerel (Macer, 1977) and anglerfish (Crozier, 1985; Olaso *et al.*, 1999; Velasco *et al.*, 2001; Cabral and Murta, 2002; Le Loc'h, 2004; Laurenson and Priede, 2005).
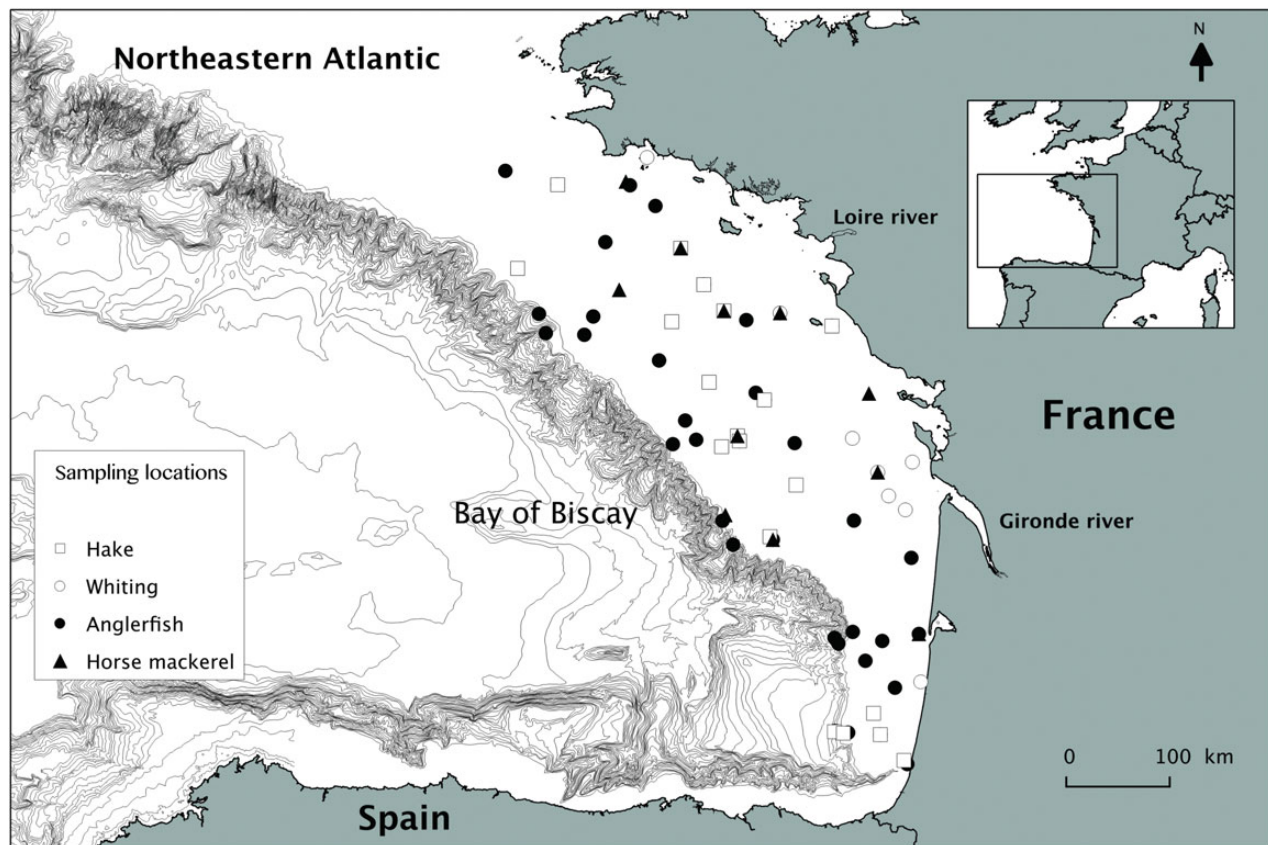
Figure 1. Map of sampling locations within the Bay of Biscay for whiting *Merlangius merlangus*, European hake *Merluccius merluccius*, Atlantic horse mackerel *Trachurus trachurus* and anglerfish *Lophius piscatorius*. The grey lines are isobaths corresponding to the shelf break (200 m depth and beyond), separating therefore the continental shelf area (where most of the sampling occurred) and deep-water areas. The Loire and Gironde rivers are also indicated.

Table 2. Characteristics of the species analysed (n = sample size, C:N mass ratios after delipidation, muscle $\delta^{13}C$ and $\delta^{15}N$ values, muscle Hg concentrations) for four fish species from the Bay of Biscay.

Species	n	Total length (mm) Min, max	C:N mass ratio Mean ± SD (min, max)	$\delta^{13}C$ (‰) Mean ± SD (min, max)	$\delta^{15}N$ (‰) Mean ± SD (min, max)	[Hg] (ng·g^{-1} dm) Mean ± SD (min, max)
Whiting (*Merlangius merlangus*)	47	80, 480	3.1 ± 0.0 (3.1, 3.2)	−16.6 ± 0.4 (−17.6, −15.9)	14.5 ± 0.5 (13.2, 15.3)	342 ± 278 (41, 1065)
Hake (*Merluccius merluccius*)	90	65, 720	3.1 ± 0.0 (3.1, 3.2)	−17.8 ± 0.5 (−19.0, −16.7)	12.8 ± 1.0 (10.7, 14.9)	267 ± 375 (26, 1954)
Horse mackerel (*Trachurus trachurus*)	57	80, 410	3.2 ± 0.1 (3.1, 3.4)	−18.2 ± 0.9 (−20.2, −17.1)	12.0 ± 1.2 (8.9, 14.0)	307 ± 317 (17, 1112)
Anglerfish (*Lophius piscatorius*)	61	80, 1020	3.1 ± 0.0 (3.0, 3.2)	−17.1 ± 0.5 (−18.3, −15.7)	13.3 ± 1.2 (9.1, 15.9)	716 ± 489 (43, 2460)

Values are mean ± Standard Deviation (SD), and/or minimum and maximum.

considering this latitudinal effect). As prey assemblages and therefore prey availability may vary from one year to another, and because interannual variations in the isotopic signatures of some prey species from the Bay of Biscay have been shown to occur (Chouvelon *et al.*, 2012b), the year (Y) of sampling of individuals was also treated as a possible categorical explanatory variable and added as an interaction term when appropriate. Details of the original models tested for each species are provided in Table 3.

To identify size-related, diet-related, spatial and/or year-related trends for explaining Hg concentration variability, GGAMs were fitted to individual log-transformed Hg concentrations. Size and $\delta^{15}N$ values of individuals covaried within each species (Figure 2 and results of the previous GGAMS), so these two parameters could not be included in the same model (Zuur *et al.*, 2007). Consequently, TL and $\delta^{13}C$ values of individuals were first considered as the continuous explanatory variables in all models, while

Table 3. Details of the sampling performed for each species and of the original models that could be tested for each species. The number of individuals sampled per latitudinal zone (LZ) and per year (Y) is given, as well as the size range in mm (in brackets).

Species	Sampling details					Original models tested
Whiting			**Year (Y) ***			
(*Merlangius*			2008	2009	**Total**	$\delta^{15}N = s(TL, by = Y) + Y + s(\delta^{13}C)/$
merlangus)	**Latitudinal zone (LZ)**	North	25 (80, 470)	16 (210, 480)	**41** (80, 480)	$\log[Hg] = s(TL, by = Y) + Y + s(\delta^{13}C)/\log[Hg] = s(\delta^{15}N, by = Y) + Y$
		South	1 (450)	5 (200, 230)	**6** (200, 450)	
		Total	**26** (80, 470)	**21** (200, 480)	**47** (80, 480)	
Hake			**Year (Y) ***			First: $\delta^{15}N = s(TL) + LZ + Y + s(\delta^{13}C)/$
(*Merluccius*			2008	2009	**Total**	$\log[Hg] = s(TL) + LZ + Y + s(\delta^{13}C)/\log[Hg] = s(\delta^{15}N) + LZ + Y$
merluccius)	**Latitudinal zone (LZ)***	North	27 (130, 630)	36 (65, 720)	**63** (65, 720)	Then: $\delta^{15}N = s(TL, by = LZ) + LZ + Y + s(\delta^{13}C)/$
		South	5 (470, 680)	22 (90, 720)	**27** (90, 720)	$\log[Hg] = s(TL, by = LZ) + LZ + Y + s(\delta^{13}C)/\log[Hg] = s(\delta^{15}N, by = LZ) + LZ + Y$
		Total	**32** (130, 680)	**58** (65, 720)	**90** (65, 720)	
Horse mackerel			**Year (Y) ***			
(*Trachurus*			2008	2009	**Total**	$\delta^{15}N = s(TL, by = Y) + Y + s(\delta^{13}C)/$
trachurus)	**Latitudinal zone (LZ)**	North	31 (80, 410)	21 (80, 320)	**52** (80, 410)	$\log[Hg] = s(TL, by = Y) + Y + s(\delta^{13}C)/\log[Hg] = s(\delta^{15}N, by = Y) + Y$
		South	0	5 (80 – 100)	**5** (80, 100)	
		Total	**31** (80, 410)	**26** (80, 320)	**57** (80, 410)	
Anglerfish			**Year (Y)**			
(*Lophius*			2008	2009	**Total**	$\delta^{15}N = s(TL, by = LZ) + LZ + s(\delta^{13}C)/$
piscatorius)	**Latitudinal zone (LZ)***	North	9 (530, 970)	35 (80, 940)	**44** (80, 970)	$\log[Hg] = s(TL, by = LZ) + LZ + s(\delta^{13}C)/\log[Hg] = s(\delta^{15}N, by = LZ) + LZ$
		South	4 (520, 910)	13 (100, 1020)	**17** (100, 1020)	
		Total	**13** (520, 970)	**48** (80, 1020)	**61** (80, 1020)	

*Attributed to the factor(s) that could be tested in the GGAM models, depending on the number of individuals and the size range within each category (i.e. sampling balanced enough for considering the latitudinal and/or the year effect).
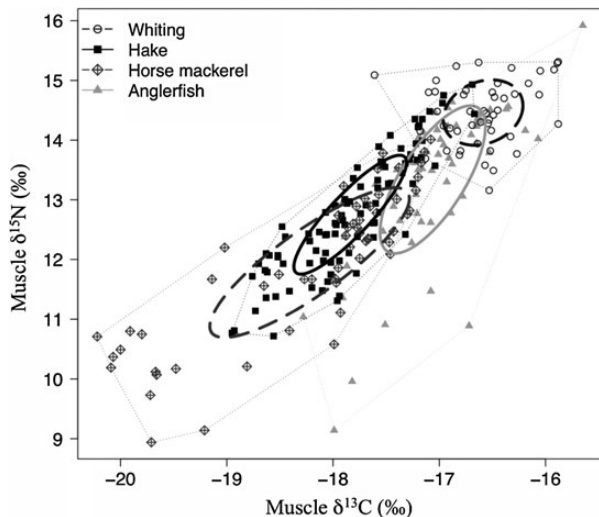
Figure 2. Individual muscle $\delta^{13}C$ and $\delta^{15}N$ values for the four fish species analysed, namely whiting *Merlangius merlangus*, European hake *Merluccius merluccius*, Atlantic horse mackerel *Trachurus trachurus* and anglerfish *Lophius piscatorius*. The total area (TA; clear dotted lines) and the standard ellipse area (SEAc; bold lines) derived from individual $\delta^{13}C$ and $\delta^{15}N$ values are also represented for each species (see sections "Data treatment" and "Species-specific differences and variability in stable isotope ratios and Hg concentrations").
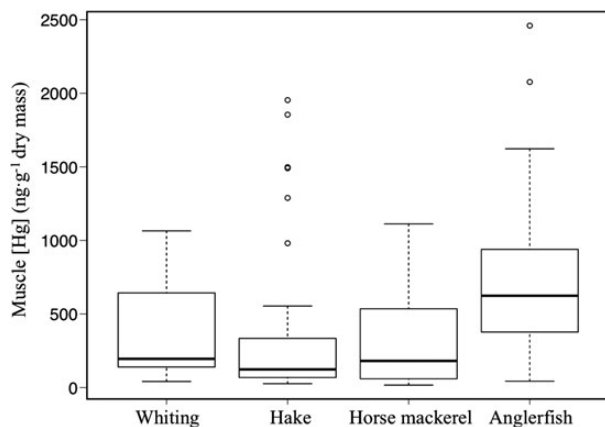

Figure 3. Boxplots of muscle Hg concentrations measured in the four fish species considered, namely whiting *Merlangius merlangus*, European hake *Merluccius merluccius*, Atlantic horse mackerel *Trachurus trachurus* and anglerfish *Lophius piscatorius*. The box length represents the interquartile, the bar length represents the range, and the horizontal lines in bold are median values.

LZ and/or Y were treated as categorical explanatory variables and added as interaction terms when appropriate. In a second step, GGAMS included individual $\delta^{15}N$ values (instead of TL) as the continuous explanatory variable while, as previously, LZ and/or Y were treated as categorical explanatory variables and added as interaction terms when appropriate. $\delta^{13}C$ values were not included because of their covariation with $\delta^{15}N$ values (following Zuur *et al.*, 2007). Details of the original models tested for each species are provided in Table 3.

Finally, a selection of the "best" GGAM was based upon the Akaike Information Criterion (AIC), as the smaller the AIC, the simpler the

model and the more it explains data variability. Also, when the AIC was not significantly different between the two last nested models, the simplest model was preferred (Zuur *et al.*, 2007). A model validation was finally systematically applied by checking normality and homogeneity in models' residuals with no violation of independence (Zuur *et al.*, 2007; Zuur, 2012). Models not complying with all these assumptions were not presented.

Results

Analysis of species-specific differences in feeding strategies or Hg bioaccumulation patterns with size would have been more powerful if "species" could have been included as a fixed factor in a single modelling for each parameter (i.e. $\delta^{15}N$ values on the one hand and Hg concentrations on the other hand). Such models were tested but not kept because model validation was not good (due to violation of independence, notably because size range is different among the four species).

Species-specific differences and variability in stable isotope ratios and Hg concentrations

Stable isotope ratios and Hg concentrations varied greatly between and within the four species (Figure 2, Table 2). The species range of $\delta^{13}C$ and $\delta^{15}N$ values was the largest in anglerfish (3.1‰ and 6.8‰, respectively), and the narrowest in whiting (1.7‰ and 2.1‰, respectively) (Table 2). However, TA and SEAc calculated were the highest for horse mackerel (7.6 and 1.8, respectively), followed by anglerfish (7.5 and 1.4) and hake (3.5 and 0.8). Whiting presented the lowest values of TA and SEAc (2.5 and 0.6, respectively) (Figure 2). Overall, Hg ranged from $17-2460$ ng·g^{-1} dm, with the largest range of values in anglerfish and the narrowest in whiting (Table 2, Figure 3).

On average, the four fish species segregated by both their $\delta^{13}C$ and $\delta^{15}N$ values (p_{KW} and $p_{post\ hoc}$ tests all < 0.001). The coastal and bentho-pelagic whiting presented the highest average $\delta^{13}C$ and $\delta^{15}N$ values, while the pelagic horse mackerel presented the lowest values. The bentho-pelagic hake displayed significantly lower $\delta^{13}C$ and $\delta^{15}N$ values than whiting, and anglerfish showed intermediate isotopic signatures relative to both species of Gadiforms (Table 2, Figure 2). On the other hand, only Hg concentrations in anglerfish differed significantly from those of the three other species (p_{KW} < 0.001 and $p_{post\ hoc}$ tests all < 0.001 for anglerfish, $p_{post\ hoc}$ tests all > 0.05 between the three other species) (Table 2, Figure 3).

Intraspecific variability of muscle $\delta^{15}N$ values

Taking into account the model with the best statistical value for each species (herein the "best model"), the total explained deviance in $\delta^{15}N$ values variability was high (52%, 86%, 90% and 85% for whiting, hake, horse mackerel and anglerfish, respectively) (Table 4). The effect of TL on $\delta^{15}N$ values was highly significant for all species (all p < 0.0001) and, overall, the relationships between muscle $\delta^{15}N$ values and TL showed an increase in $\delta^{15}N$ values with increasing size of individuals (Figure 4). The effect of $\delta^{13}C$ values was also significant for all species except for whiting. Finally, a significant effect of LZ and of the interaction term was found for hake, but not for anglerfish (Table 4). Individuals of hake sampled in the south of the Bay of Biscay presented significantly lower $\delta^{15}N$ values than those from the north, whatever their size (Figure 4).

Table 4. Detailed results for "best models" (i.e. significantly smallest Akaike Information Criterion AIC) selected to explain $\delta^{15}N$ values or (log-transformed) Hg concentrations variability in the muscle of the four fish species from the Bay of Biscay.

Species	$\delta^{15}N$ (‰)		log [Hg] (ng·g^{-1} dm)			
	$\delta^{15}N$ as a function of size (TL)		log [Hg] as a function of size (TL)		log [Hg] as a function of $\delta^{15}N$	
	Variables	p-value	Variables	p-value	Variables	p-value
Whiting (*Merlangius merlangus*)	$\delta^{15}N = s(TL)$ AIC = 49 Explained deviance = 52%		log [Hg] = s(TL, by = Y) + Y AIC = −67 Explained deviance = 93%		—	
	s(TL)	<0.0001***	s(TL)*2008	<0.0001***		
			s(TL)*2009	<0.0001***		
			Y	0.576		
Hake (*Merluccius merluccius*)	$\delta^{15}N = s(TL,$ by = LZ) + LZ + s($\delta^{13}C$) AIC = 89 Explained deviance = 86%		log [Hg] = s(TL, by = LZ) + LZ + s($\delta^{13}C$) AIC = −107 Explained deviance = 93%		log [Hg] = s($\delta^{15}N$) + LZ AIC = 2 Explained deviance = 72%	
	s(TL)*North	0.0021**	s(TL)*North	<0.0001***	s($\delta^{15}N$)	<0.0001***
	s(TL)*South	<0.0001***	s(TL)*South	<0.0001***	LZ	<0.0001***
	LZ	<0.0001***	LZ	<0.0001***		
	s($\delta^{13}C$)	<0.0001***	s($\delta^{13}C$)	0.0002***		
Horse mackerel (*Trachurus trachurus*)	$\delta^{15}N = s(TL) + s(\delta^{13}C)$ AIC = 83 Explained deviance = 90%		log [Hg] = s(TL, by = Y) + Y + s($\delta^{13}C$) AIC = −83 Explained deviance = 97%		—	
	s(TL)	<0.0001***	s(TL)	<0.0001***		
	s($\delta^{13}C$)	<0.0001***	s(TL)*2008	<0.0001***		
			s(TL)*2009	<0.0001***		
			Y	0.178		
			s($\delta^{13}C$)	0.0491*		
Anglerfish (*Lophius piscatorius*)	$\delta^{15}N = s(TL) + s(\delta^{13}C)$ AIC = 108 Explained deviance = 85%		log [Hg] = s(TL) + LZ + s($\delta^{13}C$) AIC = −58 Explained deviance = 88 %		—	
	s(TL)	<0.0001***	s(TL)	<0.0001***		
	s($\delta^{13}C$)	<0.0001***	LZ	0.0289*		
			s($\delta^{13}C$)	0.0023**		

Explanatory variables included in the models are specified [i.e. total length (TL), and/or latitudinal zone (LZ) of sampling, and/or year of sampling (Y), and/or $\delta^{13}C$ values, and/or $\delta^{15}N$ values when appropriate; see section "Data treatment" and Table 2]. AIC values and the total explained deviance by each best model performed are also indicated. Finally, the *p*-value associated with each term of the model applied is given, indicating the degree of significance for the effect of an explanatory variable within each best model. *$p < 0.05$; **$p < 0.01$; ***$p < 0.001$.

Intraspecific variability of muscle Hg concentrations

All the models run with TL were valid (i.e. assumptions of normality and homogeneity in residuals were respected, with no violation of independence), while most models run with $\delta^{15}N$ values were not. Only those for hake was valid (Table 4).

For all species, the total explained deviance in muscle Hg concentration variability was high in the best models run with TL, ranging between 88 and 97% (Table 4). For hake, the total explained deviance was higher when log-transformed Hg concentrations were modelled as a function of TL (93%) than when modelled as a function of $\delta^{15}N$ values (72%), and the AIC was accordingly considerably lower (−107 vs. 2; Table 4).

Considering the best models with TL as the continuous explanatory variable, the effect of size was highly significant for all species (all $p < 0.0001$). Overall, GGAM smoothers revealed a continuous Hg bioaccumulation with increasing individual size (Figure 5). A significant effect of the interaction term associated with Y was found in whiting and horse mackerel, highlighting a slightly different form of the relationship between log-transformed Hg concentrations and TL over the two years of sampling (Figure 5).

A significant effect of LZ was found for hake and anglerfish (Table 4), with individuals sampled in the south of the Bay of Biscay presenting significantly higher Hg concentrations than those sampled in the northern part of the area for similar size ($p < 0.0001$ and $p = 0.0289$ for hake and anglerfish, respectively). However, the interaction term associated with LZ was only significant for hake (Table 4), highlighting some difference in the shape of the relationship between log-transformed Hg concentrations and TL between individuals from the two zones (especially in larger individuals; Figure 5).

Considering the best model using $\delta^{15}N$ values as the continuous explanatory variable (instead of TL) for explaining Hg concentration variability in hake, the effects of $\delta^{15}N$ values and of LZ were also highly significant ($p < 0.0001$ for both variables; Table 3). For similar $\delta^{15}N$ value, individuals sampled in the south of the Bay of Biscay presented with significantly higher Hg concentrations than those sampled in the northern part of the area. However, the interaction term was not significant, suggesting an identical relationship of Hg concentrations with $\delta^{15}N$ values for individuals from the two zones (Figure 6).

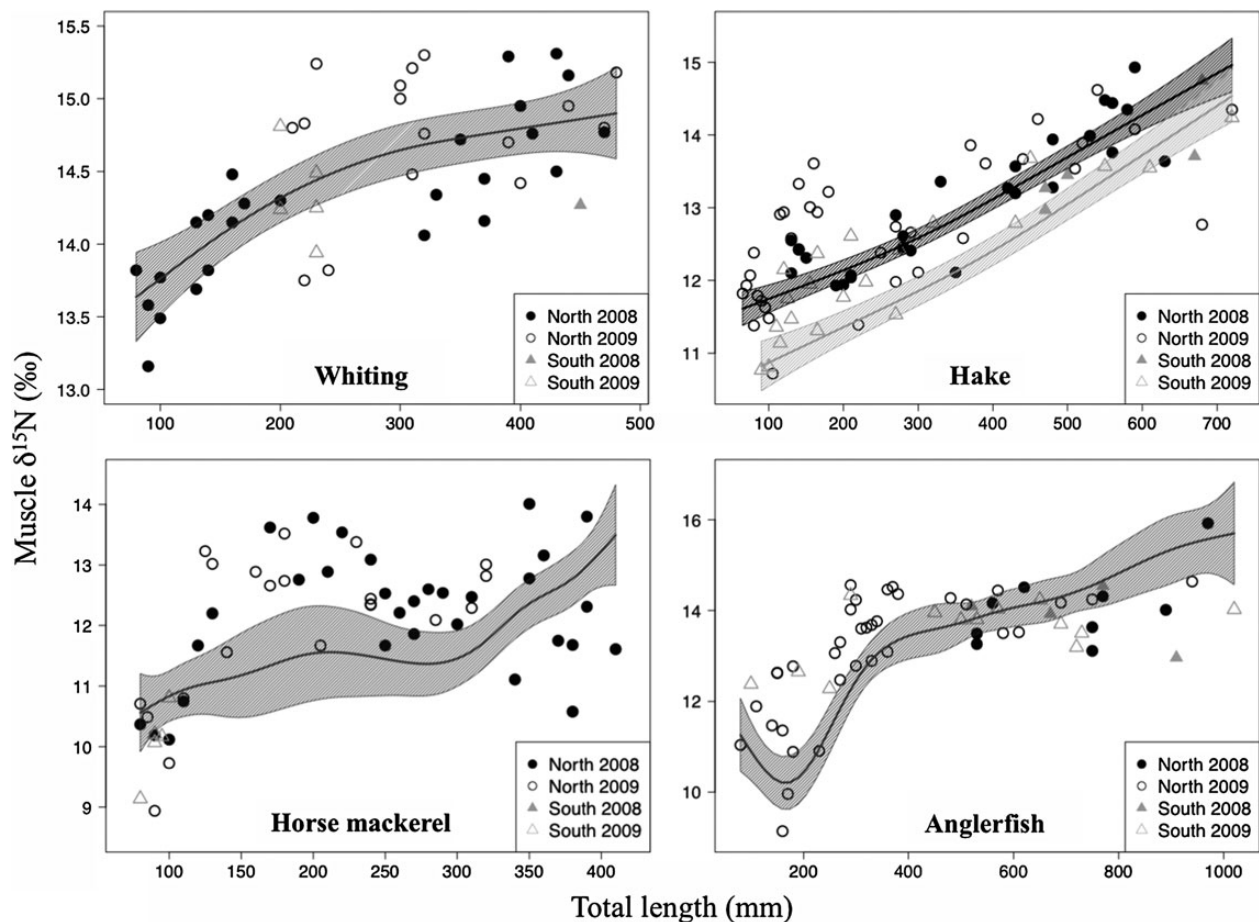**Figure 4.** Relationships between muscle $\delta^{15}N$ values (‰) and total length (TL, mm) for the four fish species from the Bay of Biscay. Results from the GGAM models (fitted to individual $\delta^{15}N$ values to identify size-related and spatial and/or interannual trends explaining the variability of values within each species, see text) are plotted on raw $\delta^{15}N$ and TL data in the form of a smoother (i.e. polygon including 95% confidence interval), illustrating the modelled relationship between both parameters. For the species for which the latitudinal zone (LZ) and the interaction term were significant in the GGAM (i.e. hake; see Table 3), two smoothers are then presented (one in black for northern individuals, one in grey for southern individuals).

Discussion

Muscle $\delta^{13}C$ and $\delta^{15}N$ values were good proxies for the different feeding ecology of fishes, both between species (segregation) and within species (size-related changes). In contrast, muscle Hg concentrations alone did not segregate the four fish species. As expected, Hg concentrations increased with size (i.e. age) and trophic level ($\delta^{15}N$ values) within each species. Finally, hakes sampled in the southern part of the Bay of Biscay presented significantly lower muscle $\delta^{15}N$ values and higher Hg concentrations than those sampled in the northern part, whatever their size. This study therefore provides crucial information for the combined use of these ecological tracers (stable isotope ratios and Hg concentrations in marine organisms) to refine indicators for monitoring the ecological status of marine foodwebs and ecosystems. It also highlights their potential usefulness for segregating populations, notably those of hake in European waters, by working at a more relevant time-scale than the genetic to define management units.

Stable isotopes document species- and size-related feeding strategies

Average isotope values followed the decreasing order: whiting > anglerfish > hake > horse mackerel. Therefore, compared with

the habitat and general distribution assigned to each species in the literature, the isotopic segregation of the four species agreed relatively well with the isoscapes previously described for the Bay of Biscay (Chouvelon *et al.*, 2012b; Nerot *et al.*, 2012). Higher $\delta^{13}C$ and $\delta^{15}N$ values were effectively recorded by these authors for coastal and/or benthic to bentho-pelagic organisms (e.g. whiting), when compared with more offshore and/or pelagic organisms (e.g. horse mackerel) in this ecosystem. Such isotopic trends were related to a variety of potentially co-occurring factors: (i) river plumes may be a vector of ^{15}N-enriched organic matter into coastal waters (McClelland *et al.*, 1997; Vizzini and Mazzola, 2006), which is then diluted in offshore areas; (ii) benthic algae were reported to be enriched by 5‰ on average relative to phytoplankton in marine coastal environments, due to the differential carbon fixation and greater diffusion resistance by benthic algae (France, 1995); (iii) fractionation processes occurring at the dissolved inorganic nitrogen level, and differential nutrient availability during phytoplankton blooms in coastal waters or in upwelling areas may enrich or deplete in ^{15}N the nitrogen sources available for primary producers in different environments (see detailed review by Montoya, 2007); the associated "isotopic perturbation" may then propagate in the different food chains; etc. As such, the
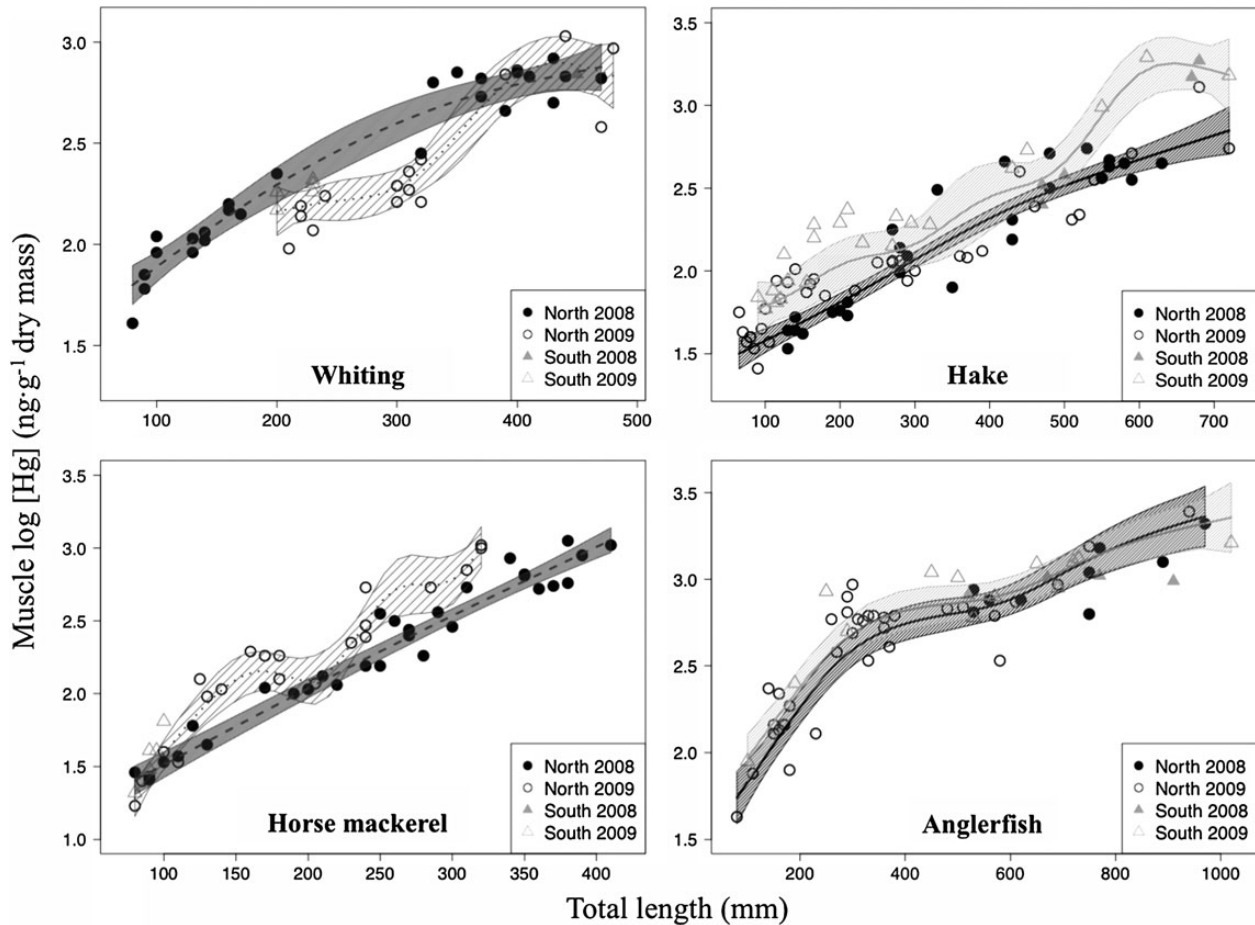
Figure 5. Relationships between muscle log-transformed Hg concentrations (ng·g^{-1} dry mass) and total length (TL, mm) for the four fish species from the Bay of Biscay. Results from the GGAM models (fitted to individual log-transformed Hg concentrations to identify size-related and spatial and/or interannual trends explaining the variability of concentrations within each species, see text) are plotted on raw log-transformed Hg concentrations and TL data in the form of a smoother (i.e. polygon including 95% confidence interval), illustrating the modelled relationship between both parameters. For the species for which the latitudinal zone (LZ) and/or the interaction term were significant in the GGAM (i.e. hake and anglerfish; see Table 3), two smoothers are then presented (one in black for northern individuals, one in grey for southern individuals). For the species for which the interaction term and/or the year of sampling (Y) was significant in the GGAM (i.e. whiting and horse mackerel; see Table 3), two smoothers are then presented (one with dense dark grey lines for 2008 individuals, one with lighter grey lines for 2009 individuals).

anglerfish (*a priori* a deeper species occurring more offshore than the hake) should have presented lower average δ^{13}C and δ^{15}N values, which is not the case. A higher consumption of benthic to bentho-pelagic prey by anglerfish may thus explain higher average isotope values by the anglerfish in comparison with the hake (which feeds mostly on small pelagic fish when adult). Moreover, adult hakes are known to inhabit shelf-edge areas as well (Quéro, 2003; Trenkel *et al.*, 2009; Palomares and Pauly, 2010), where they would prey upon blue whiting *Micromesistius poutassou* (Guichet, 1995; Mahe *et al.*, 2007). Finally, several biotic and abiotic parameters such as species, age, type and quality of the consumer's food, temperature, etc. may also impact the trophic enrichment factors between prey and consumers (e.g. Vanderklift and Ponsard, 2003; Caut *et al.*, 2009; Martínez del Rio *et al.*, 2009). This may in turn influence the isotopic differences measured between individuals and consumer species analysed, calling for caution in the interpretation of isotopic data.

The positive and significant within-species relationships between δ^{15}N values and TL agree with the common expectation that the trophic level of organisms increases with growth, because

body size generally drives prey sizes (Cohen *et al.*, 1993). However, the shape of the relationship was species-specific, confirming the potential of the isotopic tool to depict different size-related feeding patterns. This has already been shown in other taxa, such as crustaceans, cephalopods, marine mammals (Le Loc'h and Hily, 2005; Chouvelon *et al.*, 2011; Meissner *et al.*, 2012) and fish (Jennings *et al.*, 2001, 2002). For whiting, the progressive transition to piscivory deduced from stomach contents data was well illustrated by the changing isotopic signatures from smaller to larger individuals. For hake, δ^{15}N values increased linearly with size, and this pattern was also consistent with that described for individuals of this species from the Great Mud Bank area in the Bay of Biscay (Le Loc'h and Hily, 2005). The continuous increase in δ^{15}N values with predator size therefore suggests progressive dietary shifts with size for both species of Gadiforms, namely an increase in the trophic level of both species throughout growth/mass increase of the individuals. The diets of horse mackerel and anglerfish, and their potential changes with growth are poorly documented in the Northeast Atlantic. Hence, stable isotopes offer an opportunity to obtain new information on their size-related feeding strategies.
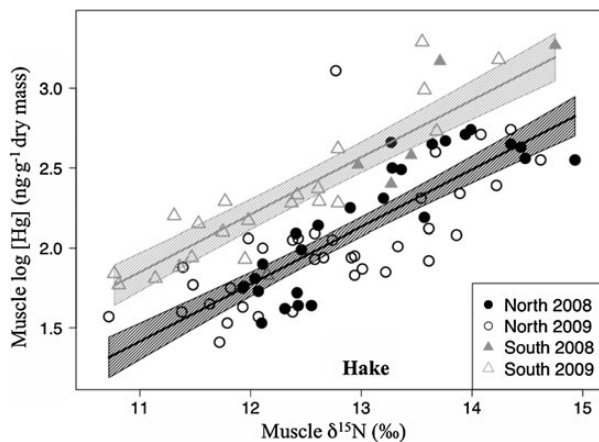
Figure 6. Relationship between muscle log-transformed Hg concentrations (ng·g^{-1} dry mass) and $\delta^{15}N$ values (in ‰) for hake from the Bay of Biscay. Results from the GGAM model (fitted to individual log-transformed Hg concentrations to identify diet-related and spatial trends explaining the variability of concentrations within the species, see text) are plotted on raw log-transformed Hg concentrations and TL data in the form of a smoother (i.e. polygon including 95% confidence interval), illustrating the modelled relationship between both parameters. As the latitudinal zone (LZ) (but not the interaction) was significant in the GGAM (see Table 3), two smoothers are presented (one in black for northern individuals, one in grey for southern individuals).

Neither species showed the clear increase in $\delta^{15}N$ values with predator size, as did whiting and hake. In horse mackerel, only the largest individuals (>30 cm) are reported to possibly prey on small fish (<10 cm). Before reaching such sizes, and along a large part of their life cycle, planktonic crustaceans effectively remain the main prey items for this species. This could explain the very slight slope observed between individuals of ~ 150–300 mm TL in horse mackerel. In anglerfish, a very small increase in $\delta^{15}N$ values with predator size was also observed for individuals from ~ 350–1000 mm TL. This is further evidence that body size is not always well correlated with trophic level (e.g. Jennings et al., 2002, for fish) and, hence, a change in the size structure of a community does not necessarily mean that there is a change in its trophic level structure.

With regard to the significant role of $\delta^{13}C$ value (and not only TL) in explaining $\delta^{15}N$ value variability in three out of the four species analysed (i.e. hake, horse mackerel, anglerfish), it is undoubtedly inherent that there is both ^{13}C and ^{15}N enrichment between sources and consumers, even if this is less important for ^{13}C than for ^{15}N (De Niro and Epstein 1978, 1981; Post, 2002). On the other hand, we cannot exclude a possible effect of age-dependent or length-based migrations that can occur in pelagic fish (e.g. Nøttestad et al., 1999) such as horse mackerel. The feeding of some individuals and/or part of the populations in different areas [presenting different baseline signatures in $\delta^{13}C$ (and $\delta^{15}N$) in the Bay of Biscay (Chouvelon et al., 2012b)] may then also explain part of the interindividual variation in $\delta^{15}N$ values. Furthermore, $\delta^{13}C$ values were not significant in explaining $\delta^{15}N$ values variability in whiting, probably because of the relatively low range of spatial distribution of this species (and consequently the narrowest range in $\delta^{13}C$ values) in comparison with the others. Indeed, the range of isotope values, TA and SEAc values all followed the decreasing order: anglerfish or horse mackerel > hake > whiting. This is quite consistent with the usual depth range described, the range of sizes sampled, and the reported

variety of the types of prey captured at the different stages within the four species analysed. However, parts of the whole isotopic niche width of each species are probably missing here because (i) some size classes could not be sampled (especially the smallest ones), and (ii) sampling occurred in autumn, thus missing a possible seasonality of the diet within species.

Finally, the spatial parameter LZ was only significant in hake for explaining intraspecific $\delta^{15}N$ values variability. Southern individuals presented lower $\delta^{15}N$ values than northern individuals. However, the shape of the relationship between $\delta^{15}N$ values and TL was quite similar for the two groups, despite the significance of the interaction term in the model. This phenomenon of distinct $\delta^{15}N$ values between southern and northern individuals has previously been reported for coastal species of fish and cephalopods from the Bay of Biscay (Chouvelon et al., 2012b). This result confirms that hakes from the southern part of the area do not feed on the same food sources as those from the northern part (e.g. different prey composition, or the same prey with different isotopic signatures in the different environments). This also suggests that there is no significant feeding migration of individuals between the areas during growth.

Size-related, diet-related and spatial patterns of Hg bioaccumulation in muscle

As previously found in cephalopods from the Bay of Biscay (Chouvelon et al., 2011), muscle Hg concentrations poorly discriminated fish species. Size-related patterns for muscular Hg concentrations were quite uniform between species, with Hg concentrations increasing continuously with increasing size. This confirms that age is a key factor determining Hg bioaccumulation in fish muscle (e.g. Monteiro and Lopes, 1990; Cronin et al., 1998; Cossa et al., 2012). Indeed, Hg mainly accumulates as methyl-Hg, and this organic form likely binds with sulfhydryl groups of muscular proteins (Bloom, 1992; Konovalov, 1994; Harris et al., 2003). Methyl-Hg then shows very slow elimination rates (Amlund et al., 2007; Dang and Wang, 2012), leading to Hg bioaccumulation with age in the muscle. On the other hand, muscle Hg concentrations were linked to the trophic level ($\delta^{15}N$) of organisms as well. The second model kept for hake for explaining variability in hake Hg concentrations (using $\delta^{15}N$ values instead of TL as the main explanatory covariable) effectively highlighted a positive and significant relationship between Hg concentrations and $\delta^{15}N$ values. However, the AIC of this second model was higher and the total explained deviance lower than for the first model with TL.

Body size was not the only covariable explaining Hg concentration variability; spatial parameters in particular (i.e. $\delta^{13}C$ values and/or LZ) were also highly significant in most of the models performed. Muscle $\delta^{13}C$ values were significantly and positively related to Hg concentrations in three out of the four species (i.e. whiting, hake, anglerfish; relationships not shown). This is likely to have resulted from the enrichment in ^{13}C with trophic level rather than potentially enhanced bioaccumulation of Hg in more coastal organisms (Chouvelon et al., 2012b). Indeed, Hg rather tends to bioaccumulate more in organisms living in meso-pelagic and deep-water environments than in those living in shallower waters (Monteiro et al., 1996; Thompson et al., 1998; Choy et al., 2009; Chouvelon et al., 2012a). This has been related to an enhanced production of methyl-Hg and of monomethyl-Hg in particular in subthermocline, low-oxygen waters; this organic form of Hg is very stable and the most bioavailable form for marine organisms (Bacci, 1989; Fitzgerald et al., 2007). Moreover, as mentioned

above, Hg accumulates with age, and deep-sea fauna are thought to include very long-lived species; even though uncertainties still exist in the age determination of these organisms (e.g. Allain and Lorance, 2000; Cailliet *et al.*, 2001).

The effect of Y was minimal (i.e. significant interaction only) or negligible (i.e. not significant) in all models where this factor could be tested. As such, the slightly different form of the relationship between log-transformed Hg concentrations and size between the two years of sampling for whiting and horse mackerel may be due to different prey availability and thus variability in the diet of these species between years. On the other hand, a significant effect of LZ was detected in hake and anglerfish. Southern individuals presented higher muscle Hg concentrations than northern individuals for similar size, and this effect was particularly pronounced in hake. Such differences between southern and northern individuals can be explained by both biotic (i.e. growth rate) and abiotic factors (i.e. methyl-Hg exposure through food), as recently described for hake from the Gulf of Lion (northwestern Mediterranean) vs. hake from the northern part of the Bay of Biscay (Cossa *et al.*, 2012). If individuals from the southern stock would grow at higher rates and mature earlier than previously considered (Piñeiro and Saínza, 2003), the possible slower growth rate of southern hakes could then explain, at least in part, the higher Hg bioaccumulation by this population. Indeed, a hake from the northern part of the Bay of Biscay and with a TL of ca. 60 cm would be ∼ 3 years old (De Pontual *et al.*, 2006) compared with ∼ 7 years old for a similar-sized individual from the southern stock (Piñeiro and Saínza, 2003). Alternatively, the continental shelf in the southern Bay of Biscay is significantly narrower than in the northern part, so individuals feeding in the southern area may have a greater access to methyl-Hg-enhanced meso-pelagic prey (Chouvelon *et al.*, 2012a for the Bay of Biscay case study). This hypothesis is also supported by the fact that the two groups display different $\delta^{15}N$ values, with southern hake always presenting lower $\delta^{15}N$ values characteristic of more oceanic feeding in the Bay of Biscay (Chouvelon *et al.*, 2012a). Besides, lower $\delta^{15}N$ values of southern individuals relative to northern individuals may be also due to different nitrogen inputs (with different isotopic composition) from both major river plumes in the Bay of Biscay that are then reflected in local food-webs. For some coastal species from this ecosystem, lower $\delta^{15}N$ values have effectively been reported in individuals collected in the southern Bay of Biscay in comparison with those collected in the northern part (Chouvelon *et al.*, 2012b).

Stable isotopes and Hg concentrations as tools for refining ecological indicators of marine ecosystems and defining managements units for the European hake

Our isotopic results highlighted species-specific feeding strategies during ontogenesis. In particular, in some species (i.e. horse mackerel and anglerfish), individuals could show similar $\delta^{15}N$ values (commonly used as a proxy for the trophic level) while they were of very different sizes. This could lessen the accuracy of using size-based indicators to evaluate the ecological state of foodwebs, because it implies that the size structure of a community does not necessarily reflect its trophic level structure, and consequently its underlying trophic linkages and dynamics (for a complete review see Rombouts *et al.*, 2013). Other potential limitations of size-based indicators include that they generally only take into account demersal fish species because they are monitored for fisheries purposes. Also, short-term variations in recruitment processes may cause a

bias in interpretation by directly affecting the size spectrum (e.g. Nicholson and Jennings, 2004; Greenstreet and Rogers, 2006; Greenstreet *et al.*, 2011). All these issues strongly suggest that using size-based metrics casually is not ideal, despite apparent advantages such as the routine collection of these data (i.e. size spectrum of the species caught) during scientific surveys (Greenstreet and Rogers, 2006). On the other hand, Hg concentrations increased continuously with increasing size of individuals within all species due to bioaccumulation with both age and/or with trophic level. In fact, it appears that the coupling of both profiles (i.e. $\delta^{15}N$ values not transformed in trophic levels and Hg concentrations as a function of size of individuals, taking into account their sampling zone) is likely to give accurate information on the transition of individuals to another trophic level from a certain size. From its size and from these isotopic and mercury profiles, a consumer could therefore be assigned quite accurately to a trophic guild. Moreover, using both these metrics in significant species of an ecosystem (e.g. those contributing to 90% of the biomass, excluding plankton) could enable a regular assessment of Hg contamination of the food-webs studied. As the Bay of Biscay continental shelf foodweb is mainly controlled by lower trophic levels and detrital biomass (Lassalle *et al.*, 2011), this would mainly include benthic invertebrate species and fish (demersal but also small pelagic fish) that could be sampled for SIA and Hg analyses in this specific area.

In the case of hake frequenting the Bay of Biscay, this study showed that both parameters (isotopic composition and Hg concentrations) measured in different-sized individuals can help the discrimination of distinct hake populations and management units in the Northeast Atlantic. Indeed, difficulties in age estimation for this species have hampered the assessment of stocks for a long time (Piñeiro and Saínza, 2003). Here, we assume the definition of management units as "demographically independent populations whose population dynamics depend largely on local birth and death rates rather than on immigration" (Palsbøll *et al.*, 2007), or again as "geographical areas with restricted interchange of the individuals of interest with adjacent areas" (Taylor and Dizon, 1999). A recent pilot study has already explored the potential of combining different tools and notably ecological tracers for discriminating common dolphin populations and defining such management units in the Northeast Atlantic (Caurant *et al.*, 2009). In the area, the common dolphin is a protected species subjected to bycatch due to interactions with fisheries (Morizur *et al.*, 1999; Lassalle *et al.*, 2012; Spitz *et al.*, 2013). By confronting at a interregional scale the results inferred from stomach contents analysis, ecological tracers analysis (i.e. trace metals, stable isotopes, fatty acids) and genetic analysis, three possible management units have been proposed for this small cetacean species in European Atlantic waters (Caurant *et al.*, 2009). Our study confirms that ecological tracers (i.e. chemical parameters such as stable isotopes or trace metal concentrations used to infer the trophic ecology of individuals) can help in discriminating different populations/stocks and defining (short-term) management units of a species, when other methods (e.g. genetic and age determination) present their own limitations (e.g. Taylor and Dizon, 1999). One must keep in mind that when using ecological tracers, only the differences are effectively informative, as different prey compositions can lead to similar isotopic signatures or trace metal concentrations in a predator's tissues (e.g. Bearhop *et al.*, 2004). Overall, migrations of hake are estimated to be low, even within a stock (e.g. between the Bay of Biscay and the Celtic Sea; Drouineau *et al.*, 2010). Here, the clear segregation for both isotopic signatures (notably $\delta^{15}N$

values) and Hg concentration between southern and northern hakes sampled in the Bay of Biscay, whatever the size of individuals, confirms the existence of two stocks exploiting different trophic resources in these waters. The interchange of individuals between northern and southern areas of the Bay of Biscay cannot be excluded, however it seems clearly restricted.

Acknowledgements

Funding

References

Allain, V., and Lorance, P., 2000. Age estimation and growth of some deep-sea fish from the Northeast *Atlantic Ocean. Cybium*, 24: 7–16.

Amlund, H., Lundebye, A. K., and Berntssen, M. H. G. 2007. Accumulation and elimination of methylmercury in Atlantic cod (*Gadus morhua* L.) following dietary exposure. Aquatic Toxicology, 83: 323–330.

Bacci, E. 1989. Mercury in the Mediterranean. Marine Pollution Bulletin, 20: 59–63.

Bearhop, S., Adams, C. E., Waldron, S., Fuller, R. A., and MacLeod, H. 2004. Determining trophic niche width: a novel approach using stable isotope analysis. Journal of Animal Ecology, 73: 1007–1012.

Bloom, N. S. 1992. On the chemical form of mercury in edible fish and marine invertebrate tissue. Canadian Journal of Fisheries and Aquatic Sciences, 49: 1010–1017.

Boening, D. 2000. Ecological effects, transport, and fate of mercury: a general review. Chemosphere, 40: 1335–1351.

Borja, A., Galparsoro, I., Irigoien, X., Iriondo, A., Menchaca, I., Muxika, I., Pascual, M., et al. 2011. Implementation of the European Marine Strategy Framework Directive: a methodological approach for the assessment of environmental status, from the Basque Country (Bay of Biscay). Marine Pollution Bulletin, 62: 889–904.

Bustamante, P., Lahaye, V., Durnez, C., Churlaud, C., and Caurant, F. 2006. Total and organic Hg concentrations in cephalopods from the North East Atlantic waters: influence of geographical origin and feeding ecology. Science of the Total Environment, 368: 585–596.

Cabana, G., and Rasmussen, J. B. 1994. Modelling food chain structure and contaminant bioaccumulation using stable nitrogen isotopes. Nature, 372: 255–257.

Cabral, H. N., and Murta, A. G. 2002. The diet of blue whiting, hake, horse mackerel and mackerel of Portugal. Journal of Applied Ichthyology, 18: 14–23.

Cailliet, G. M., Andrews, A. H., Burton, E. J., Watters, D. L., Kline, D. E., and Ferry-Graham, L. A. 2001. Age determination and validation studies of marine fishes: do deep-dwellers live longer? Experimental Gerontology, 36: 739–764.

Castillo, A. G. F., Martinez, J. L., and Garcia-Vasquez, E. 2004. Fine spatial structure of Atlantic hake (*Merluccius merluccius*) stocks revealed by variation at microsatellite loci. Marine Biotechnology, 6: 299–306.

Caurant, F., Chouvelon, T., Lahaye, V., Mendez-Fernandez, P., Rogan, E., Spitz, J., and Ridoux, V. 2009. The use of ecological tracers for discriminating populations: the case of the short-beaked common dolphin *Delphinus delphis* in the European Atlantic waters. International Whaling Commission, SC61/SM/34.

Caut, S., Angulo, E., and Courchamp, F. 2009. Variation in discrimination factors ($\Delta^{15}N$ and $\Delta^{13}C$): the effect of diet isotopic values and applications for diet reconstruction. Journal of Applied Ecology, 46: 443–453.

Cherel, Y., Fontaine, C., Richard, P., and Labat, J. P. 2010. Isotopic niches and trophic levels of myctophid fishes and their predators in the Southern Ocean. Limnology and Oceanography, 55: 324–332.

Cherel, Y., and Hobson, K. A. 2007. Geographical variation in carbon stable isotope signatures of marine predators: a tool to investigate their foraging areas in the Southern Ocean. Marine Ecology Progress Series, 329: 281–287.

Chouvelon, T., Spitz, J., Caurant, F., Mèndez-Fernandez, P., Autier, J., Lassus-Débat, A., Chappuis, A., et al. 2012a. Enhanced bioaccumulation of mercury in deep-sea fauna from the Bay of Biscay (northeast Atlantic) in relation to trophic positions identified by analysis of carbon and nitrogen stable isotopes. Deep-Sea Research Part I, 65: 113–124.

Chouvelon, T., Spitz, J., Caurant, F., Mèndez-Fernandez, P., Chappuis, A., Laugier, F., Le Goff, E., et al. 2012b. Revisiting the use of $\delta^{15}N$ in meso-scale studies of marine food webs by considering spatio-temporal variations in stable isotopic signatures—the case of an open ecosystem: the Bay of Biscay (North-East Atlantic). Progress in Oceanography, 101: 92–105.

Chouvelon, T., Spitz, J., Cherel, Y., Caurant, F., Sirmel, R., Mèndez-Fernandez, P., and Bustamante, P. 2011. Inter-specific and ontogenic differences in $\delta^{13}C$ and $\delta^{15}N$ values and Hg and Cd concentrations in cephalopods. Marine Ecology Progress Series, 433: 107–120.

Choy, C. A., Popp, B. N., Kaneko, J. J., and Drazen, J. C. 2009. The influence of depth on mercury levels in pelagic fishes and their prey. Proceedings of the National Academy of Sciences of the United States of America, 106: 13865–13869.

Cimmaruta, R., Bondanelli, P., and Nascetti, G. 2005. Genetic structure and environmental heterogeneity in the European hake (*Merluccius merluccius*). Molecular Ecology, 14: 2577–2591.

Cohen, J. E., Pimm, S. L., Yodzis, P., and Saldaña, J. 1993. Body sizes of animal predators and animal prey in food webs. Journal of Animal Ecology, 62: 67–78.

Cope, J. M, and Punt, A. E. 2009. Drawing the lines: resolving fishery management units with simple fisheries data. Canadian Journal of Fisheries and Aquatic Sciences, 66: 1256–1273.

Cossa, D., Harmelin-Vivien, M., Mellon-Duval, C., Loizeau, V., Averty, B., Crochet, S., Chou, L., et al. 2012. Influences of bioavailability, trophic position, and growth on methylmercury in hakes (*Merluccius merluccius*) from Northwestern Mediterranean and Northeastern Atlantic. Environmental *Science and Technology*, 46: 4885–4893.

Cronin, M., Davies, I. M., Newton, A., Pirie, J. M., Topping, G., and Swan, S. 1998. Trace metal concentrations in deep sea fish from the North Atlantic. Marine Environmental Research, 45: 225–238.

Crozier, W. W. 1985. Observations on the food and feeding of the angler-fish, *Lophius piscatorius* L., in the northern Irish Sea. Journal of Fish Biology, 27: 655–665.

Cury, P., Shannon, L., and Shin, Y. J. 2003. The functioning of marine ecosystems: a fisheries perspective. *In* Responsible Fisheries in the Marine Ecosystem, pp. 103–123. Ed. by M. Sinclair, and G. Valdimarsson. CAB International, Walingford.

Dang, F., and Wang, W. X. 2012. Why mercury concentration increases with fish size? Biokinetic explanation. Environmental Pollution, 163: 192–198.

De Niro, M. J., and Epstein, S. 1977. Mechanism of carbon fractionation associated with lipid synthesis. Science, 197: 261–263.

De Niro, M. J., and Epstein, S. 1978. Influence of diet on the distribution of carbon isotopes in animals. Geochimica et Cosmochimica Acta, 42: 495–506.

De Niro, M. J., and Epstein, S. 1981. Influence of diet on the distribution of nitrogen isotopes in animals. Geochimica et Cosmochimica Acta, 45: 341–351.

De Pontual, H., Groison, A. L., Pineiro, C., and Bertignac, M. 2006. Evidence of underestimation of European hake growth in the Bay of Biscay, and its relationship with bias in the agreed method of age estimation. ICES Journal of Marine Science, 63: 1674–1681.

Drouineau, H., Mahévas, S., Bertignac, M., and Duplisea, D. 2010. A length-structured spatially explicit model for estimating hake growth and migration rates. ICES Journal of Marine Science, 67: 1697–1709.

Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. US Fish and Wildlife Service Biological Report 85, (1.10).

European Commission. 2008. Directive 2008/56/EC of the European Parliament and of the Council of 17 June 2008, establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive). Official Journal of the European Union, L 164, pp. 19–40.

European Commission. 2010. Commission Decision of 1 September 2010 on criteria and methodological standards on good environmental status of marine waters. Official Journal of the European Union, L 232, pp. 14–24.

Fitzgerald, W. F., Lamborg, C. H., and Hammerschmidt, C. R. 2007. Marine biogeochemical cycling of mercury. Chemical Reviews, 107: 641–662.

France, R. L. 1995. Carbon-13 enrichment in benthic compared to planktonic algae: food web implications. Marine Ecology Progress Series, 124: 307–312.

Fritsch, M., Morizur, Y., Lambert, E., Bonhomme, F., and Guinand, B. 2007. Assessment of sea bass (*Dicentrarchus labrax*, L.) stock delimitation in the Bay of Biscay and the English Channel based on mark–recapture and genetic data. Fisheries Research, 83: 123–132.

Gray, J. S. 2002. Biomagnification in marine systems: the perspective of an ecologist. Marine Pollution Bulletin, 45: 46–52.

Greenstreet, S. P. R., McMillan, J. A., and Armstrong, E. 1998. Seasonal variation in the importance of pelagic fish in the diet of piscivorous fish in the Moray Firth, NE Scotland: a response to variation in prey abundance? ICES Journal of Marine Science, 55: 121–133.

Greenstreet, S. P. R., and Rogers, S. I. 2006. Indicators of the health of the North Sea fish community: identifying reference levels for an ecosystem approach to management. ICES Journal of Marine Science, 63: 573–593.

Greenstreet, S. P. R., Rogers, S. I., Rice, J. C., Piet, G. J., Guirey, E. J., Fraser, H. M., and Fryer, R. J. 2011. Development of the EcoQO for the North Sea fish community. ICES Journal of Marine Science, 68: 1–11.

Guichet, R. 1995. The diet of European hake (*Merluccius merluccius*) in the northern part of the Bay of Biscay. ICES Journal of Marine Science, 52: 21–31.

Harris, H. H., Pickering, I. J., and George, G. N, 2003. The chemical forms of mercury in fish. Science, 301: 1203.

Hislop, J. R. G., Robb, A. P., Bell, M. A., and Armstrong, D. W. 1991. The diet and food consumption of whiting (*Merlangius merlangus*) in the North Sea. ICES Journal of Marine Science, 48: 139–156.

Hobson, K. A. 1999. Tracing origins and migration of wildlife using stable isotopes: a review. Oecologia, 120: 314–326.

ICES. 2008. Report of the Working Group on the Assessment of Southern Shelf Stocks of Hake, Monk and Megrim (WGHMM). ICES Document CM 2008/ACOM: 07, 613 pp.

Jackson, A. L., Inger, R., Parnell, A. C., and Bearhop, S. 2011. Comparing isotopic niche widths among and within communities: SIBER – Stable Isotope Bayesian Ellipses in R. Journal of Animal Ecology, 80: 595–602.

Jennings, S., Greenstreet, S. P. R., Hill, L., Piet, G. J., Pinnegar, J. K., and Warr, K. J. 2002. Long-term trends in the trophic structure of the North Sea fish community: evidence from stable isotope analysis, size-spectra and community metrics. Marine Biology, 141: 1085–1097.

Jennings, S., Pinnegar, J. K., Polunin, N. V. C., and Boon, T. V. 2001. Weak cross-species relationships between body size and trophic level belie powerful size-based trophic structuring in fish communities. Journal of Animal Ecology, 70: 934–944.

Kelly, J. F. 2000. Stable isotopes of carbon and nitrogen in the study of avian and mammalian trophic ecology. Canadian Journal of Zoology, 78: 1–27.

Konovalov, Y. D. 1994. A review of the binding of cadmium and mercury in fish by proteins and low molecular thiols. Hydrobiological Journal, 30: 42–51.

Lassalle, G., Gascuel, D., Le Loc'h, F., Lobry, J., Pierce, G. J., Ridoux, V., Santos, M. B., *et al.* 2012. An ecosystem approach for the assessment of fisheries impacts on marine top predators: the Bay of Biscay case study. ICES Journal of Marine Science, 69: 925–938.

Lassalle, G., Lobry, J., Le Loc'h, F., Bustamante, P., Certain, G., Delmas, D., Dupuy, C., *et al.* 2011. Lower trophic levels and detrital biomass control the Bay of Biscay continental shelf food web: implications for ecosystem management. Progress in Oceanography, 91: 561–575.

Laurenson, C. H., and Priede, I. G. 2005. The diet and trophic ecology of anglerfish *Lophius piscatorius* at the Shetland Islands, UK. Journal of the Marine Biological Association UK, 85: 419–424.

Layman, C. A., Arrington, D. A., Montana, C. G., and Post, D. M. 2007. Can stable isotope ratios provide for community-wide measures of trophic structure? Ecology, 88: 42–48.

Le Loc'h, F. 2004. Structure, fonctionnement, évolution des communautés benthiques des fonds meubles exploités du plateau continental Nord Gascogne. PhD Thesis, Université de Bretagne Occidentale, Brest, France.

Le Loc'h, F., and Hily, C. 2005. Stable carbon and nitrogen isotope analysis of *Nephrops norvegicus*/*Merluccius merluccius* fishing grounds in the Bay of Biscay (Northeast Atlantic). Canadian Journal of Fisheries and Aquatic Sciences, 62: 123–132.

Lindeman, R. L. 1942. The trophic-dynamic aspect of ecology. Ecology, 23: 399–418.

Macer, C. T. 1977. Some aspects of the biology of the horse mackerel *Trachurus trachurus* (L.) in waters around Britain. Journal of Fish Biology, 10: 51–62.

Mahe, K., Amara, R., Bryckaert, T., Kacher, M., and Brylinski, J. M. 2007. Ontogenetic and spatial variation in the diet of hake (*Merluccius merluccius*) in the Bay of Biscay and the Celtic Sea. ICES Journal of Marine Science, 64: 1210–1219.

Martínez del Rio, C., Wolf, N., Carleton, S. A., and Gannes, L. Z. 2009. Isotopic ecology ten years after a call for more laboratory experiments. Biological Reviews of the Cambridge Philosophical Society, 84: 91–111.

McClelland, J. M., Valiela, I., and Michener, R. H., 1997. Nitrogen-stable isotope signatures in estuarine food webs: a record of increasing urbanization in coastal watersheds. Limnology and Oceanography, 42: 930–937.

Meissner, A. M., MacLeod, C. D., Richard, P., Ridoux, V., and Pierce, G. 2012. Feeding ecology of striped dolphins, *Stenella coeruleoalba*, in the north-western Mediterranean Sea based on stable isotope analyses. Journal of the Marine Biological Association UK, 92: 1677–1687.

Michener, R. H., and Kaufman, L. 2007. Stable isotope ratios as tracers in marine food webs: an update. *In* Stable Isotopes in Ecology and Environmental Science, pp. 238–282. Ed. by R. Michener, and K. Lajtha. Blackwell Publishing Ltd, 566 pp.

Monteiro, L. R., Costa, V., Furness, R. W., and Santos, R. S. 1996. Mercury concentrations in prey fish indicate enhanced

bioaccumulation in mesopelagic environments. Marine Ecology Progress Series, 141: 21–25.

Monteiro, L. R., and Lopes, H. D. 1990. Mercury content of swordfish, Xiphias gladius, in relation to length, weight, age, and sex. Marine Pollution Bulletin, 21: 293–296.

Montoya, J. P., 2007. Natural abundance of ^{15}N in marine planktonic ecosystems. *In* Stable Isotopes in Ecology and Environmental Science, pp. 76–201. Ed. by R. Michener, and K. Lajtha. Blackwell Publishing Ltd, 566 pp.

Moore, J. W., and Moore, I. A. 1976. The basis of food selection in some estuarine fishes: eels, *Anguilla anguilla* (L.), whiting, *Merlangius merlangus* (L.), sprat, *Sprattus sprattus* (L.) and stickleback, *Gasterosteus aculeatus* (L.). Journal of Fish Biology, 9: 375–390.

Morizur, Y., Berrow, S. D., Tregenza, N. J. S., Couperus, A. S., and Pouvraud, S. 1999. Incidental catches of marine mammals in pelagic trawl fisheries of the northeast Atlantic. Fisheries Research, 41: 297–307.

Murua, H. 2010. The biology and fisheries of European hake, *Merluccius merluccius*, in the north-east Atlantic. Advances in Marine Biology, 58: 97–154.

Navarro, J., Coll, M., Louzao, M., Palomera, I., Delgado, A., and Forero, M. G. 2011. Comparison of ecosystem modelling and isotopic approach as ecological tools to investigate food webs in the NW Mediterranean Sea. Journal of Experimental Marine Biology and Ecology, 401: 97–104.

Nerot, C., Lorrain, A., Grall, J., Gillikin, D. P., Munaron, J. M., Le Bris, H., and Paulet, Y. M. 2012. Stable isotope variations in benthic suspension feeders across a large depth gradient on the continental shelf. Estuarine, Coastal and Shelf Science, 96: 228–235.

Newsome, S. D., Martínez del Rio, C., Bearhop, S., and Phillips, D. L. 2007. A niche for isotopic ecology. Frontiers in Ecology and Environment, 5: 429–436.

Nicholson, M. D., and Jennings, S. 2004. Testing candidate indicators to support ecosystem-based management: the power of monitoring surveys to detect temporal trends in fish community metrics. ICES Journal of Marine Science, 61: 35–42.

Nøttestad, L., Giske, J., Holst, J. C., and Huse, G. 1999. A length-based hypothesis for feeding migrations in pelagic fish. Canadian Journal of Fisheries and Aquatic Sciences, 56: 26–34.

Odum, E. P. 1969. The strategy of ecosystem development. Science, 164: 262–270.

Olaso, I., Cenderero, O., and Abaunza, P. 1999. The diet of horse mackerel, *Trachurus trachurus* (Linneaus, 1758), in the Cantabrian Sea (north Spain). Journal of Applied Ichthyology, 15: 193–198.

Palomares, M. D., and Pauly, D. (Eds) 2010. SeaLifeBase, World Wide Web electronic publication, Version (12/2010). www.sealifebase.org (last accessed 25 November 2013).

Palsbøll, P. J., Bérubé, M., and Allendorf, F. W. 2007. Identification of management units using population genetic data. Trends in Ecology and Evolution, 22: 11–16.

Pawson, M. G., Pickett, G. D., Leballeur, J., Brown, M., and Fritsch, M. 2007. Migrations, fishery interactions, and management units of sea bass (*Dicentrarchus labrax*) in Northwest Europe. ICES Journal of Marine Science, 64, 332–345.

Pedersen, J. 1999. Diet comparison between pelagic and demersal whiting in the North Sea. Journal of Fish Biology, 55: 1096 –1113.

Peterson, B. J., and Fry, B. 1987. Stable isotopes in ecosystem studies. Annual Review of Ecology and Systematics, 18: 293–320.

Piñeiro, C., and Saínza, M. 2003. Age estimation, growth and maturity of the European hake (*Merluccius merluccius* (Linnaeus, 1758)) from Iberian Atlantic waters. ICES Journal of Marine Science, 60: 1086–1102.

Pope, J. G., Shepherd, J. G., Webb, J., Stebbing, A. R. D., and Mangel, M. 1994. Successful surf-riding on size spectra: the secret of survival in the sea. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences, 343: 41–49.

Post, D. M. 2002. Using stable isotopes to estimate trophic position: models, methods and assumptions. Ecology, 83: 703–718.

Post, D. M., Layman, C. A., Arrington, D. A., Takimoto, G., Quattrochi, J., and Montaña, C. G. 2007. Getting to the fat of the matter: models, methods, and assumptions for dealing with lipids in stable isotope analyses. Oecologia, 152: 179–189.

Quéro, J.C. 2003. Guide des Poissons de l'Atlantique Européen. Les guides du naturaliste. Ed. by Delachaux and Niestlé, Paris, France.

Ramos, R., Ramírez, F., and Jover, L. 2013. Trophodynamics of inorganic pollutants in a wide-range feeder: the relevance of dietary inputs and biomagnification in the Yellow-legged gull (*Larus michahellis*). Environmental Pollution, 172: 235–242.

R Development Core Team. 2011. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, www.R-project.org (last accessed 25 November 2013).

Roldan, M. I., Marin, J. L. G., Utter, F. M., and Pla, C. 1998. Population genetic structure of European hake, *Merluccius merluccius*. Heredity, 81: 327–334.

Rombouts, I., Beaugrand, G., Fizzala, X., Gaill, F., Greenstreet, S. P. R., Lamare, S., Le Loc'h, F., *et al.* 2013. Food web indicators under the Marine Strategy Framework Directive: from complexity to simplicity? Ecological Indicators, 29: 246–254.

Shin, Y. J., Rochet, M. J., Jennings, S., Field, J. G., and Gislason, H. 2005. Using size-based indicators to evaluate the ecosystem effects of fishing. ICES Journal of Marine Science, 62: 384–396.

Smedes, F. 1999. Determination of total lipid using non-chlorinated solvents. The Analyst, 124: 1711–1718.

Spitz, J., Chouvelon, T., Cardinaud, M., Kostecki, C., and Lorance, P. 2013. Prey preferences of adult sea bass *Dicentrarchus labrax* in the northeastern Atlantic: implications for bycatch of common dolphin *Delphinus delphis*. ICES Journal of Marine Science, 70: 452–461.

Taylor, B. L., and Dizon, A. E. 1999. First policy then science: why a management unit based solely on genetic criteria cannot work. Molecular Ecology, 8: S11–S16.

Thompson, D. R., Furness, R. W., and Monteiro, L. R. 1998. Seabirds as biomonitors of mercury inputs to epipelagic and mesopelagic marine food chains. Science of the Total Environment, 213: 299–305.

Trenkel, V. M., Berthelé, O., Lorance, P., Bertrand, J., Brind'Amour, A., Cochard, M. L., Coppin, F., *et al.* 2009. Atlas des grands invertébrés et poissons observés par les campagnes scientifiques. Bilan 2008. Ifremer, Nantes, EMH: 09–003, 100 pp.

Vanderklift, M. A., and Ponsard, S., 2003. Source of variation in consumer-diet $\delta^{15}N$ enrichment: a meta-analysis. Oecologia, 136: 169–182.

Vander Zanden, M. J., Cabana, G., and Rasmussen, J. B. 1997. Comparing trophic position of freshwater fish calculated using stable nitrogen isotope ratios ($\delta^{15}N$) and literature dietary data. Canadian Journal of Fisheries and Aquatic Sciences, 54: 1142–1158.

Velasco, F., and Olaso, I. 1998. European hake *Merluccius merluccius* (L., 1758) feeding in the Catabrian Sea: seasonal, bathymetric and length variations. Fisheries Research, 38: 33–44.

Velasco, F., Olaso, I., and Sánchez, F. 2001. The role of cephalopods as forage for the demersal fish community in the southern Bay of Biscay. Fisheries Research, 52: 65–77.

Vizzini, S., and Mazzola, A., 2006. The effects of anthropogenic organic matter inputs on stable carbon and nitrogen isotopes in organisms from different trophic levels in a southern Mediterranean coastal area. Science of the Total Environment, 368: 723–731.

Zuur, A. F., Ieno, E. N., and Smith, G. M. 2007. Analysing Ecological Data. Springer, New York (USA).

Zuur, A.F. 2012. A beginner's guide to Generalized Additive Models with R. Highland Statistics Ltd, Newburgh (UK).

Handling editor: Marta Coll