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Evidence for thyroid endocrine disruption in wild fish in San Francisco Bay, California, USA. Relationships to contaminant exposures

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ABSTRACT

It is well documented that many coastal and estuarine environments adjacent to developed and industrialized urban centers, such as the San Francisco Bay Area, are significantly contaminated by anthropogenic chemicals. However, it is not well understood to what extent existing contaminants, many with continuing inflows into the environment, may impact exposed wildlife. This study provided an initial characterization of thyroid endocrine-related effects and their relationship to accumulated contaminants in two indigenous fish species sampled from different San Franicsco Bay Area study sites. Plasma concentrations of thyroxine (T4) were significantly reduced in fish sampled from highly impacted locations such as Oakland Inner Harbor and San Leandro Bay as compared with fish from other locations representing relatively lower human impact, including Bodega Bay, Redwood City and a remote site on Santa Catalina Island. Triiodothyronine (T3) levels also varied significantly by location, with differing T3/T4 ratios in fish from some locations suggestive of altered peripheral deiodinase activity. The changes in thyroid endocrine parameters were significantly correlated with hepatic concentrations of certain environmental contaminants. A large number of polychlorinated biphenyl (PCB) congeners, both co-planar (dioxin-like) and non-co-planar, exhibited significant inverse correlations with T4 levels in the fish, while in contrast, T3 and T3/T4 ratio were positively correlated with PCB exposures. The positive correlation between T3/T4 ratio and PCBs supports the hypothesis that environmental PCBs may alter T4 deiodination or turnover, actions of PCBs reported in laboratory experiments. Some relationships between chlorinated pesticides including DDT and chlordanes, but fewer relationships with PAHs, were also observed. Together, these findings indicate that the thyroid endocrine system is exhibiting alterations associated with different aquatic environments in the San Francisco Bay Area, which are significantly related to current-day exposures of the fish to contaminant chemicals such as PCBs.

1. Introduction

In aquatic environments adjacent to large urban centers, such as in the San Francisco Bay Area, a wide variety of chemical contaminants are typically present, yet their phenotypic effects in resident fish and other wildlife are not well understood. Selected chemicals which can be present in such environments may have thyroid-disrupting effects in fish, as indicated from laboratory experiments. Since thyroid hormones have critical roles in development and physiological homeostasis, disruption of the thyroid system can present a threat to health in wild fish populations. However, studies are lacking that evaluate, within a real ecosystem, the connections between altered endocrine phenotype and environmental contaminant exposures.

Thyroid hormones are produced upon activation of the neuroendocrine hypothalamo-pituitary-thyroid (HPT) axis (reviews by Eales, 2006; Blanton and Specker, 2007; Zoeller et al., 2007). Under hypothalamic control, the pituitary secretes thyroid-stimulating hormone (TSH) which proceeds to the thyroid gland to activate synthesis of thyroxine (T4; 3,5,3′,5′-tetraiodo-L-thyronine) and triiodothyronine (T3; 3,5,3′-triiodo-L-thyroinine). T4 generally represents >95% of the thyroid hormone output and it is typically present in higher quantities than T3 in the blood circulation, with the higher T4 concentrations serving as a pool of prohormone that can be converted into T3 by 5′-iodothyronine deiodinases in target tissues (Eales, 2006; Zoeller et al., 2007). Thyroid hormones are essential for early development in fishes, including larval–juvenile transitions and induction of metamorphosis in flatfish (Inui et al., 1995; Power et al., 2001; Yamano, 2005; Shiao and Wang, 2006;

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Blanton and Specker, 2007; Klaren et al., 2008). Thyroid hormones are also deposited into the yolk of fish eggs, and used during subsequent embryonic development (Kobuke et al., 1987; Leatherland, 1989). In juvenile and adult life, they are necessary (permissive) for normal somatic growth and have a number of effects on growth endocrine genes, such as pituitary growth hormone (GH), GH receptors, and insulin-like growth factor-I (IGF-I; Power et al., 2001; Plohman et al., 2002; Schmid et al., 2003).

Given the multi-component neuroendocrine HPT axis, there are a large number of potential target genes or proteins that can be altered by endocrine-disrupting chemicals (EDCs). Such effects may range from agonistic to antagonistic actions of EDCs on target tissue receptors, to alterations in thyroid hormone synthetic pathways, deiodinase functions in peripheral tissues, or carrying proteins in the blood (Brown et al., 2004a; Boas et al., 2006; LeRoy et al., 2006; Zoeller, 2007; Crofton, 2008; Soldin et al., 2008). Polychlorinated biphenyls (PCBs), dioxin (2,3,7,8-tetrachlorodibenzo-*p*-dioxin; TCDD), chlorinated pesticides, polybrominated diphenyl ethers (PBDEs), and metals such as mercury, have all been implicated as thyroid-disruptive agents when tested experimentally in fish (Adams et al., 2000; Brown et al., 2004a,b; Boas et al., 2006; Kahn et al., 2002; LeRoy et al., 2006; Soldin et al., 2008). However, current knowledge is limited mostly to laboratory testing of a small subset of chemicals that are found within human-impacted environments.

The San Francisco Bay appears on the "Section 303(d) list" of impaired water bodies, as required by the federal Clean Water Act. The California State Water Board compiled the most recent 303(d) list in 2006, approved by the US EPA in 2007 (http://www.waterboards.ca.gov/sanfranciscobay/TMDL/303dlist.htm). Contaminants in San Francisco Bay and its major tributaries on this 303(d) list include polyaromatic hydrocarbons (PAHs), pesticides (dieldrin, chlordanes, and DDTs), additional chlorinated compounds such as PCBs, TCDD, furan compounds, and trace elements (Hg and Se). Total maximum daily loads (TMDLs), required of water bodies on the 303(d) list, have been developed for PCBs, Hg, Cu and Ni (adopted in 2008) or are currently in development (for pesticides, TCDD, others).

Contaminants such as PCBs and pesticides accumulate to high concentrations at the top of the food web and therefore may pose a health risk to piscivorous and other wildlife. The Regional Monitoring Program for Water Quality in the San Francisco Estuary (RMP; http://www.sfei.org/rmp) has monitored contaminant concentrations in sport fish every three years since 1997. Data collected by the RMP series and in other studies indicate that contaminant concentrations in the San Francisco Bay ecosystem are sufficiently elevated to cause concern for adverse effects on biota (Fairey et al., 1997; Davis et al., 2004, 2007; Greenfield et al., 2005; Oros et al., 2005, 2007; San Francisco Estuary Institute, 2008). In the 2003 and 2006 RMP fish contamination surveys, several fish species were demonstrated to have tissue concentrations of total PCBs (sum of congeners) of 100–200 ng/g wet weight (ww) or higher, as compared with a screening value of 10 ng/g ww (PCB TMDL of the San Francisco Bay Regional Water Quality Control Board). Given sediment PCB concentrations between 5.7 and 8.7 ppb in recent years (San Francisco Estuary Institute, 2008), substantial bioaccumulation into fish tissues is indicated. In addition to PCBs, several other contaminants are present in fish tissues at concentrations above thresholds of concern, including pesticides, Hg, dioxin, and PBDEs (Fairey et al., 1997; Davis et al., 2004; Greenfield et al., 2005; Oros et al., 2005, 2007). The potential for endocrine disruptive effects and other phenotypic/physiological effects due to exposure to these contaminants are not well understood.

In California and elsewhere, water quality objectives and management of coastal and estuarine environments have been based largely upon contaminant concentrations, with little additional information on biological effects in resident wildlife. This study therefore pursued establishing linkages between contaminant exposures and effects in a critical endocrine system, as a first step toward understanding phenotypic impacts of existing contaminants. We evaluated the hypothesis that current-day exposures to environmental contaminants in San Francisco Bay Area aquatic habitats can be related to indices of thyroid endocrine status in resident wild fish. In addition, it was of interest to determine whether differences in thyroid status could be differentiated among study sites that represented differing signatures of contaminant chemicals.

2. Materials and methods

2.1. Study fish

The target fish species included shiner surfperch (*Cymatogaster aggregata*) and Pacific staghorn sculpin (*Leptocottus armatus*), both indigenous to San Francisco Bay. The shiner surfperch has received continuing monitoring for tissue contaminant levels by the RMP since 1997 (http://www.SFEI.org/rmp). Pacific staghorn sculpin is typically captured at the same locations as the surfperch and exhibits strong site fidelity during its first year, remaining mostly along the Bay margins and in association with bottom sediments (Jones, 1962). Sculpin feed on sediment invertebrates including polychaetes, mollusks, and amphipods (Jones, 1962; Fitch and Lavenberg, 1975) and accumulate environmental contaminants such as PCBs, DDTs and Hg (Davis et al., 2004). Because of these characteristics, young-of-the-year sculpin were chosen as an additional potential biosentinel species.

2.2. Fish sampling locations and sample collection

In conjunction with field efforts in August 2006 for the RMP fish contamination studies (http://www.sfei.org/rmp; Hunt et al., 2008), a subset of the captured surfperch and sculpin were made available for tissue sampling for use in a pilot study. Within San Francisco Bay, fish were collected from the following locations (latitude and longitude values in parentheses): Redwood City (RED; 37.51, −122.20); San Pablo Bay (SPB; 37.96, −122.47), Berkeley waterfront (BER; 37.89, −122.32), and Oakland Inner Harbor (OAK; 37.78, −122.25) (Fig. 1). Tomales Bay (TOB; 38.14, −122.87) was chosen as a potential far-field reference location.

Based upon the results of the 2006 pilot, in which significant differences in endocrine status in both species were observed between study sites (described in Section 3), a second, expanded study was carried out in August 2007 using the same and additional study locations (Fig. 1). In addition to RED, SPB, OAK, and TOB, fish were also caught and sampled from Richmond Harbor (RCH; 37.91, −122.35), Richmond Lauritzen Channel (RLC; 37.92, −122.36), San Leandro Bay (SLB; 37.74, −122.21), San Francisco waterfront (SFW; 37.77, −122.38), and San Francisco's Hunter's Point (HPT; 37.72, −122.36). These additional sites were chosen on the basis of their expected different signatures of environmental chemicals. RLC is an EPA superfund site characterized by DDT contamination derived from former pesticide formulation businesses (http://www.epa.gov/region09/superfund/index.html); RCH serves as a nearby site but which is more proximal to the central SF Bay waters and also has a large urban marina. SLB (Daum et al., 2000) and HPT (EPA superfund site; http://www.epa.gov/region09/superfund/index.html) exhibit elevated environmental PCB levels. In addition to the above sites within San Francisco Bay, fish were also collected at far-field sites

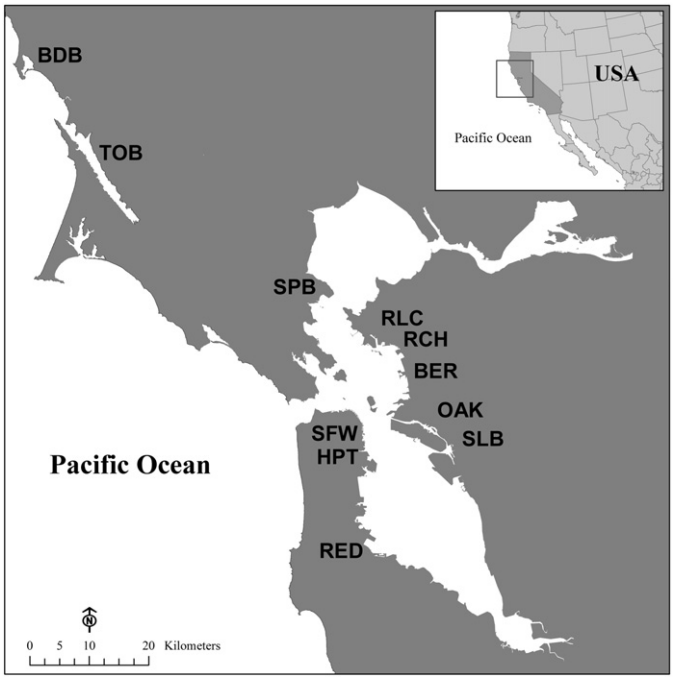

Fig. 1. San Francisco Bay Area study locations. RED = Redwood City; HPT = Hunter's Point in San Francisco; SFW = San Francisco waterfront; SPB = San Pablo Bay; RCH = Richmond Harbor; RLC = Richmond Lauritzen Channel; BER = Berkeley waterfront; OAK = Oakland Inner Harbor; SLB = San Leandro Bay; BDB = Bodega Bay; TOB = Tomales Bay. In addition, a far-field site on the Pacific Ocean side of Santa Catalina Island (not in map) was used to sample a population of shiner surfperch. Latitudes and longitudes for each site are provided (Section 2).

including TOB (as above), Bodega Bay (BDB; 38.32, −123.05), and at a remote location on the Pacific Ocean side of Santa Catalina Island (CAT; 33.38, −118.47).

2.3. Field methods

Fish were captured using an otter trawl pulled behind a small craft. Blood was collected from the caudal vein using a heparinized syringe with a 22 g needle, centrifuged for 3 min at 3000 rpm to pellet the cells, and plasma was removed. Plasma samples were frozen on dry ice until transport to the laboratory and then stored at −80 °C until hormone assays (described below). Standard body lengths (to 0.1 cm) were measured for each individual; body weights could not be measured due to equipment failure in the field. Livers were removed whole, placed into clean foil envelopes, frozen on dry ice until transport to the laboratory, and then stored at −80 °C until chemical analysis (described below). Sex of individual fish was assigned based upon gross anatomy of the internal gonads. In both years, 9–25 surfperch and 9–23 sculpin were captured and sampled per study location, with the following exceptions: TOB (0 surfperch and 5 sculpin in 2006; 6 sculpin in 2007), BDB (2 surfperch in 2007), HPT (0 surfperch in 2007), and CAT (0 sculpin in 2007).

2.4. Plasma hormone analyses

T4 and T3 concentrations in plasma were measured by enzyme immunoassay (EIA) using reagents from Immunometrics, Ltd. (London, UK) and Fisher Scientific (Tustin, CA). Standard curves for T4 or T3 (concentrations between 5 and 250 ng/ml) were used to calculate ng/ml concentrations from the unknown samples using SigmaPlot 8.0 software (SYSTAT Software Inc.). EIAs were previously validated for the fish plasma and exhibit intra-assay and inter-assay coefficients of variation of 2.9% and 4.1%, respectively (Brar, 2009). In addition to individual ng/ml values for T4 and T3, ratios of T3/T4 concentrations were calculated for each individual as an indicator of peripheral conversion of T4 to T3 (5′-deiodination).

2.5. Hepatic tissue extractions and chemical analyses

Livers were measured for PCBs, pesticides, and PAHs using EPA method 8270Cm (http://www.crgmarinelabs.com, http://www.IIRMES.org). Liver samples were weighed to the nearest 0.1 g wet weight, and extracted for organic constituents using a pressurized fluid microwave extraction method. Samples were added to 25 ml methylene chloride:acetone (3:1) at 25 °C in a solvent-rinsed Teflon pressurized extraction vessel, placed into a microwave carousel apparatus (CEM Corporation; Matthews, NC), heated to 100 °C for 15 min, and then held at 100 °C for an additional 15 min during carousel rotation. After cooling to 25 °C, the solvent was decanted into collection flasks over sodium sulfate to separate any water from the extract. The extraction process was repeated for a total of 3 extractions. Extracts were then evaporated to less than 1 ml using a Rotavapor® apparatus (Büchi Laboratory Equipment, Inc., Flawil, Switzerland), 4 ml n-hexane added, and evaporated again to 500 μl to eliminate any residual methylene chloride and acetone. This was followed by column cleaning of the extract, with the eluate subsequently evaporated to 1 ml and transferred into a 2 ml GC/MS vial.

Contaminant concentrations in each sample were then measured by gas chromatography/mass spectrometry (GC/MS) using an Agilent 6890N GC system with a DB-5 60 m gas chromatography column coupled to an Agilent 5973 Mass Selective Detector (Agilent Technologies, Santa Clara, CA). Liver tissue extractions and GC/MS methods were subjected to QA/QC procedures according to EPA guidelines. Accuracy of GC/MS data was assessed by additional analyses of matrix spikes, surrogate spikes, certified reference materials, positive controls, and/or laboratory control materials on a minimum frequency of 1 per batch, with the typical batch size consisting of 10–15 samples. Recovery surrogates were spiked into every extraction vessel prior to solvent addition. We required that 95% of the target compounds greater than 10 times the minimum detectable limit (MDL) be within the specified acceptance limits (relative percent difference of 0–30%). The acceptance ranges were determined from a minimum detection limit (MDL) study. Concentrations for each compound were calculated as ng compound per gram wet weight (ng/g ww). Values for total PCBs, total DDTs, total chlordanes, and total PAHs were also calculated, by adding together the concentrations of all individual detected congeners. A total of fifty fish were selected for hepatic contaminant analyses, as described in Section 3.

2.6. Statistical analyses

Hormone parameters included ng/ml T4 and T3 concentrations and T3/T4 ratio; hepatic chemical concentrations were expressed in ng/g ww. All statistical tests and graphical representations were carried out using SigmaStat version 3.5 and SigmaPlot 8.0 software (SYSTAT, Inc., Chicago, IL). Mean ± SE values for the different parameters were calculated for each study location and analyzed using one-way analysis of variance (ANOVA) followed by ad hoc multiple comparisons tests. For the hormone parameters, values were first log transformed, followed by ANOVA and the Holm–Sidek ad hoc test. Relationships between parameters (e.g., hormone concentration vs. hepatic contaminant concentration) were evaluated using Pearson's correlation analysis to obtain correlation coefficients (R). In all statistical analyses, differences

between groups or correlations were considered significant when $p < 0.05$.

3. Results

3.1. Animals

For surfperch, overall mean body lengths were 9.5 ± 0.4 cm in 2006 and 9.1 ± 0.1 cm in 2007, which were not significantly different between years. In addition, there were no significant differences in mean body length among groups (study locations) in 2006. In 2007, most groups were also not significantly different in length, except that the OAK (9.6 ± 0.2 cm) and CAT (9.9 ± 0.2 cm) groups had slightly higher means than the RCH (8.5 ± 0.2 cm) and TOB (8.5 ± 0.15 cm) groups ($p < 0.05$). Based upon available length and age data, all surfperch were estimated to be of 1–2 years and mature (http://hmsc.oregonstate.edu/projects/msap/); inspection of the gonads confirmed that all surfperch were gonadally mature, with varying reproductive status. Three of the females were pregnant, with 7, 4, and 6 offspring (from SFW, BDB and TOB, respectively). There was an overall sex ratio of 1:1.4 male:female. Neither size (length) nor sex had any significant interaction with any of the endocrine or hepatic parameters in this study. The three pregnant surfperch also did not exhibit any detectable differences in any measure, as compared with non-pregnant female or male surfperch.

In sculpin, mean body lengths were 9.3 ± 0.5 cm in 2006 and 9.0 ± 0.3 cm in 2007, and were not significantly different between years. There were no location-associated differences in mean body length in 2006. In 2007, most groups were also not significantly different in size, except that the SFW (11.4 ± 0.5 cm) and RLC (10.8 ± 0.3 cm) groups had significantly higher means ($p < 0.05$) as compared with the other groups. All sculpin were young-of-the-year and immature, and it was not possible to determine sex accurately. There were no significant interactions between length of sculpin and any of the other parameters in the study.

3.2. Location-associated differences in thyroid endocrine status in fish

In the 2006 pilot study, a principal aim was to determine whether differences in fish thyroid status could be identified among the different study locations. Surfperch sampled at the RED, BER, and SPB locations exhibited T4 concentrations between 35 and 45 ng/ml (Fig. 2a), levels comparable to control T4 plasma concentrations published in other fish studies (e.g., LeRoy et al., 2006; Yang et al., 2007). In contrast, surfperch sampled at OAK had significantly reduced T4 concentrations (by ≥60%, to 12.3 ± 4.4 ng/ml, $p < 0.001$). Sculpin also exhibited significantly reduced T4 concentrations at the OAK location (by >50%) as compared with fish from RED ($p < 0.01$; Fig. 2b); however, sculpin from BER and SPB also had significantly reduced T4 concentrations ($p < 0.05$). Therefore, the pilot data indicated that both species exhibited significant differences in T4 levels that occurred in a spatially distinct manner, with the OAK location associated with significant effects in both species.

During the following August (2007), surfperch showed the same spatial patterns in plasma T4 as in the pilot study, with significantly lower concentrations in fish from OAK as compared with the groups from RED and SPB ($p < 0.05$; Fig. 3a). In addition, fish from the RCH location exhibited virtually identical T4 concentrations to the 2006 group from the nearby BER site. Sculpin also exhibited the same spatial pattern in T4 concentrations (Fig. 4a) as compared with that observed in the pilot study (Fig. 3b).

The expanded number of sites in the 2007 study indicated additional spatial differences in plasma T4 concentrations in both species. In surfperch, T4 concentrations were 30–40 ng/ml in fish from BDB, RCH, and RLC, similar to the levels in fish from RED and SPB, (Fig. 3a). In addition, small quantities of plasma available from the surfperch sampled at Catalina Island (CAT) were pooled and measured for T4: the mean T4 concentration from 4 plasma pools was 34.6 ± 1.8 (4). In contrast to the above groups, surfperch sampled from other locations exhibited lower T4 concentrations: fish from OAK had lower T4 levels than the RED and RLC groups ($p < 0.05$), while fish from SLB and TOB had particularly depressed T4 levels (significantly lower than in all other groups, $p < 0.01$). In

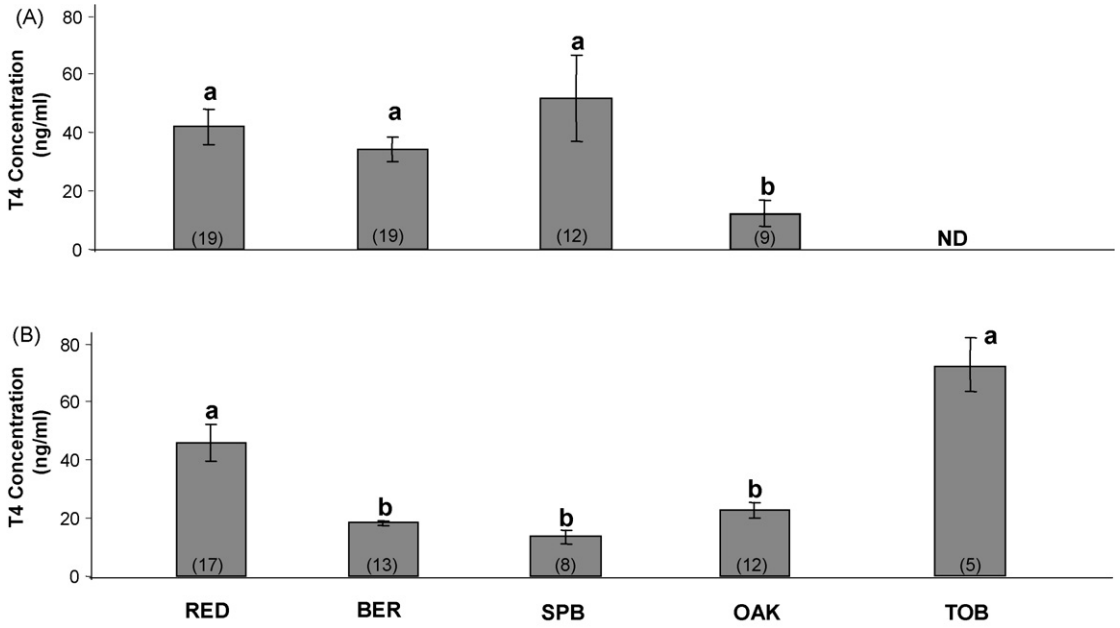

Fig. 2. Plasma thyroxine (T4) concentrations in shiner surfperch (panel A) and Pacific staghorn sculpin (panel B) sampled from different locations in San Francisco Bay in 2006, including Redwood City (RED), Berkeley waterfront (BER), San Pablo Bay (SPB), and Oakland Inner Harbor (OAK). Sculpin were also sampled from Tomales Bay (TOB). Study site locations are shown in map in Fig. 1. Bars represent mean ng/ml values ± SE, with the number of samples given in parentheses. Different (a and b) superscripts indicate significantly different values between sites.

the sculpin, spatial patterns in T4 levels (Fig. 4a) exhibited some similarities to those observed in the surfperch. Sculpin from RED and BDB exhibited relatively higher plasma T4 as compared with levels in several other groups, including OAK, SLB and TOB ($p < 0.05$). However, in contrast to surfperch, sculpin from RCH, RLC, and SPB also exhibited relatively reduced plasma T4 concentrations ($p < 0.05$ vs. RED, BDB and SFW).

Plasma T3 concentrations exhibited a spatial pattern that differed from that of T4 (Fig. 3b). Surfperch sampled from BDB, RCH, RLC, and SLB all had T3 concentrations around 12–15 ng/ml, which were significantly higher than in fish from RED, SFW, and TOB ($p < 0.05$). Calculated ratios of T3/T4 were around 0.5 in a majority of groups, and showed an overall spatial pattern comparable to that of plasma T3 concentrations (Fig. 3c). Surfperch from RED had the lowest T3/T4 ratio (0.15 ± 0.03; $p < 0.05$ vs. all other groups), while the group from SLB exhibited the opposite pattern (highest T3/T4 ratio; 1.2 ± 0.14; $p < 0.05$ vs. all other groups). It was also noted that fish from TOB exhibited relatively reduced levels of both T3 and T4, while three groups (BDB, RCH, and RLC) had relatively increased levels of both hormones (Fig. 3). As in surfperch, T3 levels in sculpin were relatively lower in the RED and TOB groups as compared with fish from BDB, RCH, and RLC ($p < 0.05$; Fig. 4b). However, the sculpin also exhibited relatively lower T3 concentrations in SPB, OAK, SLB and HPT, as compared with several other groups ($p < 0.05$). As in surfperch, T3/T4 ratios in sculpin largely followed the pattern of T3 levels (Fig. 4c) and were significantly higher in most groups as compared with the RED group ($p < 0.05$). Noted differences with the surfperch included relatively lower T3/T4 ratio in sculpin from TOB, and relatively higher T3/T4 ratio in sculpin from RLC (the latter group also exhibited relatively higher PCB exposure, as described in Section 3.3).

Therefore, over two years of study, the thyroid endocrine system in two different fish species exhibited several location-associated differences in San Francisco Bay Area aquatic habitats, a number of which were consistent between the two species. Since a key objective in the study was to determine whether alterations in thyroid endocrine status could be related to exposures to environmental contaminants in the same fish, liver concentrations of PCBs, PAHs, and pesticides were measured in selected sub-groups. These groups were then used to evaluate potential relationships using correlation analysis.

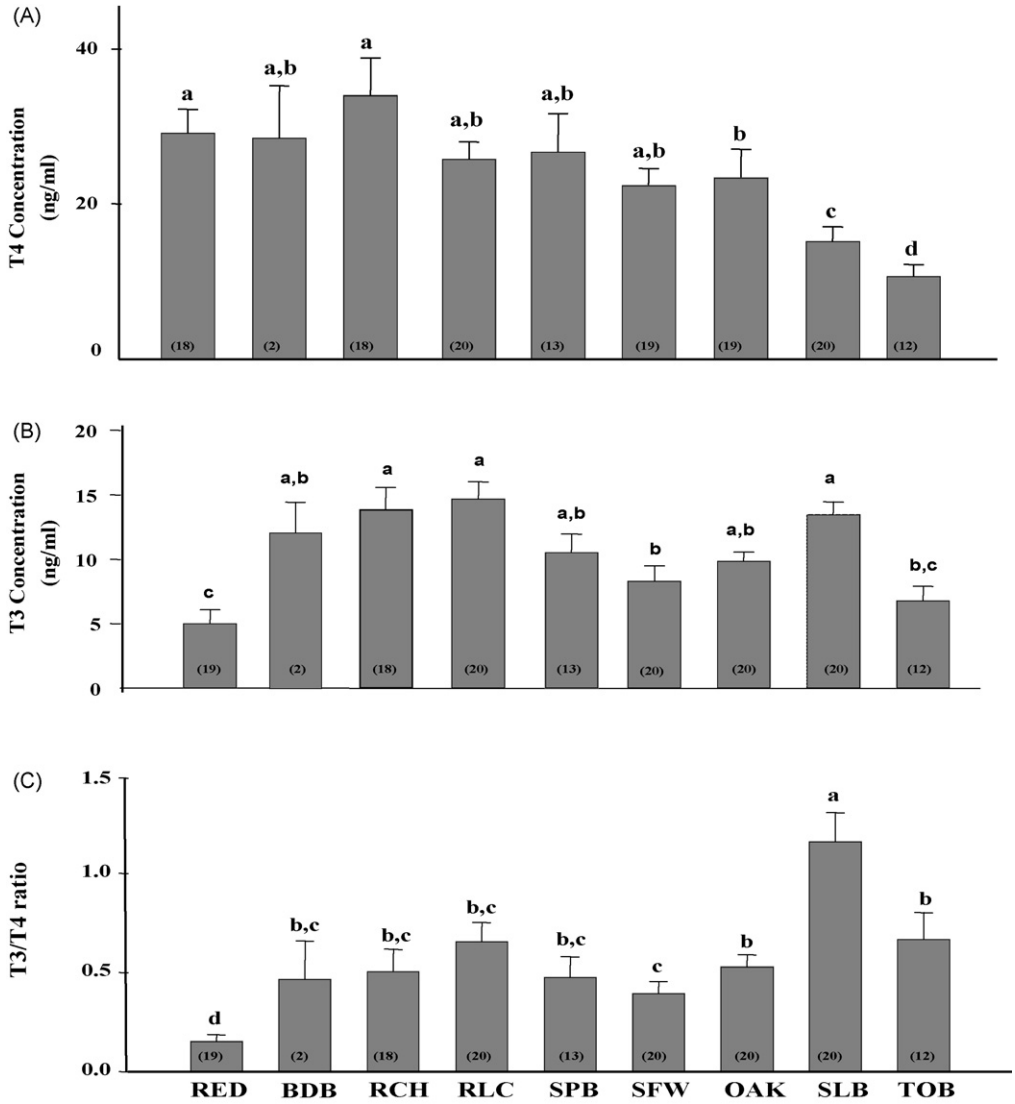

Fig. 3. Plasma T4 concentrations (panel A), T3 concentrations (panel B), and T3/T4 ratio (panel C) in shiner surfperch collected from field sites in San Francisco Bay (RED = Redwood City; RCH = Richmond Harbor; RLC = Richmond Lauritzen Channel; SPB = San Pablo Bay; SFW = San Francisco Waterfront; OAK = Oakland Inner Harbor; SLB = San Leandro Bay) and in Bodega Bay (BDB) and Tomales Bay (TOB). Study site locations are shown in map in Fig. 1. Bars represent mean ng/ml values ± SE, with the number of individuals per mean shown in parentheses. (a–d) Superscripts indicate significant differences between means ($p < 0.05$).

3.3. Hepatic contaminant concentrations in fish

Preliminary studies using liver samples from the 2006 pilot study demonstrated that a large number of contaminants, in all three targeted classes (PCBs, pesticides, PAHs), were detectable in single fish livers of both species. The rationale for selecting the 2007 study sites in which fish liver contaminant measurements would be undertaken was based upon obtaining groups of fish that were likely to be exposed to distinct environmental chemical signatures, as well as emerging data on the endocrine status for thyroxine and other hormones in fish residing at different study sites. Given the above factors, cost considerations of GC/MS analysis, and representation of both species at such study sites, the RED, OAK, SLB, and RLC groups were chosen for these analyses. As stated earlier (references cited in Section 2.2), RLC is a DDT-contaminated site and SLB is a PCB-contaminated site, while OAK was already known to have endocrine effects in the fish, as compared with less or no observable effects in fish from RED. In addition to these groups, livers from four surfperch from CAT were also analyzed; although the pooled plasma samples meant that these samples could not be included in the correlation analyses, this group provided a measure of contaminant exposures in fish from a location subjected to very minimal human influence.

Among 49 measured PCB congeners, 43 were detected overall between both species (Tables 1 and 2). In surfperch, 26 PCB congeners were present in liver at >10 ng/g in one or more of the groups, while 16 congeners were >20 ng/g in one or more of the groups (Table 1). Of the 12 dioxin-like co-planar PCBs (Safe, 1994), only PCB 118 was present in concentrations >20 ng/g in one or more of the groups, while three (PCBs 105, 126, and 156) were ≥10 ng/g in one or more of the groups. All detected PCB congeners were generally present at ≥2-fold higher concentrations in the OAK and SLB groups as compared with surfperch from RED and RLC, while in the fish from CAT, hepatic PCBs were very low (total PCBs were <20 ng/g). Overall, the highest congener concentrations measured were for PCB 153 and PCB 138 (both >100 ng/g), followed by PCBs 101, 118, 180, 187, 149, and 170, most of which were >40 ng/g in fish from OAK and SLB. In sculpin, on the other hand, a greater number of PCB congeners (32) were present at >10 ng/g in one or more groups, and 25 PCBs were >20 ng/g in one or more groups (Table 2). As in surfperch, sculpin had highest hepatic concentrations of PCB 153 and PCB 138 (both >100 ng/g), while PCBs 101, 118 (co-planar), 180, 187, 149, 158 and 170 were >40 ng/g in groups from all sites. Total PCBs were higher in sculpin as compared with surfperch at RED and RLC ($p < 0.05$), but at similar levels between species in SLB and OAK.

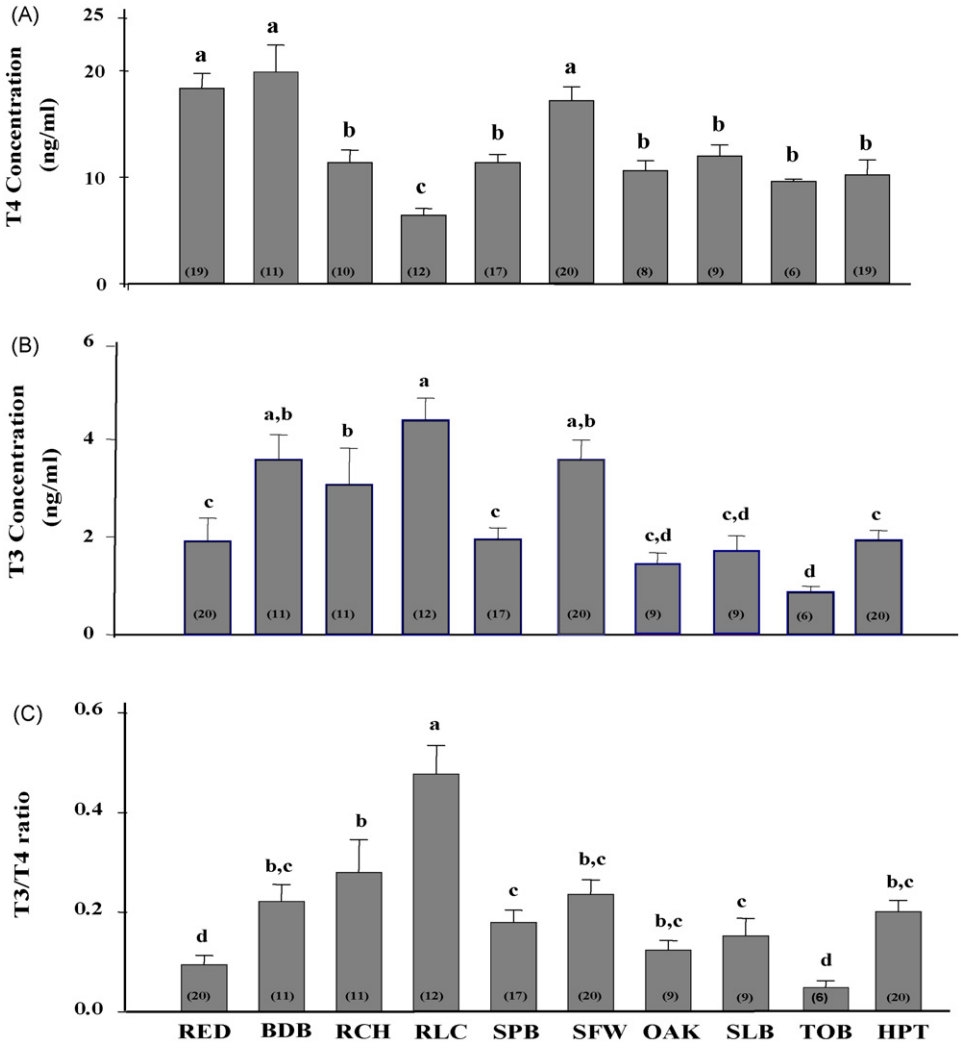

Fig. 4. Plasma T4 concentrations (panel A), T3 concentrations (panel B), and T3/T4 ratio (panel C) in Pacific staghorn sculpin collected from field sites in San Francisco Bay (RED = Redwood City; RCH = Richmond Harbor; RLC = Richmond Lauritzen Channel; SPB = San Pablo Bay; SFW = San Francisco Waterfront; OAK = Oakland Inner Harbor; SLB = San Leandro Bay; HPT = Hunters Point) and in Bodega Bay (BDB) and Tomales Bay (TOB). Study site locations are shown in map in Fig. 1. Bars represent mean ng/ml values ± SE, with the number of individuals per mean shown in parentheses. (a–d) Superscripts indicate significant differences between means ($p < 0.05$).

In surfperch, total hepatic DDT concentrations were 7–12-fold higher at RLC, the DDT-contaminated site, as compared with the other locations ($p < 0.001$; Table 3). Concentrations of 2,4-DDT, 2,4-DDD and 4,4-DDT were mostly non-detectable in surfperch, except in the RLC group; 4,4-DDE and 4,4-DDD were present in particularly high concentrations in RLC fish (293.8 ± 61.1 and 379.2 ± 89.4 ng/g, respectively). Sculpin exhibited a similar overall spatial pattern of concentrations (Table 4), with the exception that sculpin from RLC exhibited yet higher concentrations as compared with surfperch (e.g., 612.0 ± 83.6 ng/g 4,4-DDE and 1072.9 ± 302.9 ng/g 4,4-DDD; $p < 0.05$ vs. respective values in surfperch).

Gamma chlordane and alpha chlordane were detected in all surfperch groups from within San Francisco Bay, with highest concentrations in the SLB group ($p < 0.05$; Table 3). Trans-nonachlor and cis-nonachlor were also widely detected, and present in higher concentrations in the SLB group ($p < 0.05$). No chlordanes were detected in fish from CAT. In sculpin, chlordanes were similarly elevated in the SLB group as compared with RED (Table 4); however, sculpin from RLC and OAK also had elevated chlordane levels, which were 2–4-fold higher than in surfperch from the same locations ($p < 0.05$).

Hepatic concentrations of different PAH compounds were variable across the different groups of surfperch and sculpin. In surfperch, total PAHs were higher in the RLC group as compared with surfperch from the other locations ($p < 0.05$), while lowest total PAH levels were seen in the group from CAT (Table 3). In the sculpin, total PAHs were >3-fold higher as compared with corresponding surfperch groups ($p < 0.01$), except in the SLB group (Table 4). In both species, naphthalene typically constituted a major contributor (20–50%) of the total PAHs present, while other PAHs showing high hepatic concentrations in selected groups included acenaphthene, phenanthrene, anthracene, the methylnaphthalenes, and fluoranthrene.

3.4. Relationships between thyroid endocrine status and contaminant exposure

In surfperch, 9 PCB congeners exhibited significant inverse correlations with plasma T4 concentrations ($R = -0.36$ to -0.46; $p < 0.05$; Table 5), including PCBs 28, 31, 44, 49, 87, 95, 97, 110, and 167 (co-planar). All other PCB congeners also showed inverse relationships with plasma T4, with PCBs 52, 66, 101, 105 (co-planar),

Table 1
Hepatic PCB concentrations (mean ng/g ww ± SE) in shiner surfperch sampled from Redwood City (RED), San Leandro Bay (SLB), Richmond Lauritzen Channel (RLC), Oakland Inner Harbor (OAK) and Catalina Island (CAT). In parentheses, the numbers of samples with concentrations above the MDL are given in the numerator, with denominator indicating the number of individuals measured. Different (a–c) letters represent significant differences in total PCBs. PCB congeners 008, 018, 037, 081, 119 and 189 were not detected in any surfperch samples.

Congener	RED	SLB	RLC	OAK	CAT
028	1.7 ± 0.6 (2/6)	3.2 ± 0.5 (5/6)	2.1 ± 0.4 (4/5)	ND (6)	ND (4)
031	2.1 ± 0.8 (2/6)	5.7 ± 0.5 (6/6)	4.9 ± 0.7 (5/5)	ND (6)	ND (4)
033	1.4 ± 0.4 (1/6)	ND (6)	ND (5)	ND (6)	ND (4)
044	1.7 ± 0.3 (3/6)	8.4 ± 1.0 (6/6)	6.2 ± 0.7 (5/5)	ND (6)	ND (4)
049	4.0 ± 0.3 (6/6)	14.0 ± 1.3 (6/6)	6.7 ± 0.9 (5/5)	9.7 ± 1.1 (6/6)	ND (4)
052	7.7 ± 0.6 (6/6)	23.5 ± 2.6 (6/6)	13.5 ± 1.8 (5/5)	14.2 ± 1.6 (6/6)	ND (4)
066	5.8 ± 0.6 (6/6)	10.1 ± 1.1 (6/6)	8.1 ± 1.7 (5/5)	5.2 ± 0.8 (6/6)	ND (4)
070	4.9 ± 0.8 (6/6)	9.2 ± 1.0 (6/6)	5.9 ± 1.2 (5/5)	9.5 ± 1.1 (6/6)	ND (4)
074	3.8 ± 0.8 (5/6)	6.0 ± 0.8 (6/6)	6.2 ± 1.2 (5/5)	14.3 ± 1.7 (6/6)	ND (4)
077	1.2 ± 0.2 (1/6)	ND (6)	ND (5)	ND (6)	ND (4)
087	1.8 ± 0.5 (2/6)	11.2 ± 0.5 (6/6)	5.5 ± 1.4 (4/5)	ND (6)	ND (4)
095	5.2 ± 1.0 (5/6)	19.7 ± 1.7 (6/6)	8.4 ± 1.2 (5/5)	16.3 ± 3.6 (6/6)	1.8 ± 0.7 (1/4)
097	2.7 ± 1.0 (2/6)	15.6 ± 1.4 (6/6)	11.1 ± 2.1 (5/5)	3.8 ± 1.5 (5/6)	ND (4)
099	15.8 ± 2.1 (6/6)	37.4 ± 3.9 (6/6)	16.7 ± 3.1 (5/5)	28.7 ± 4.3 (6/6)	ND (4)
101	22.1 ± 4.1 (6/6)	66.9 ± 4.4 (6/6)	28.7 ± 3.8 (5/5)	53.1 ± 7.4 (6/6)	2.8 ± 1.4 (2/4)
105	3.0 ± 1.2 (2/6)	16.4 ± 1.4 (6/6)	7.3 ± 2.0 (4/5)	12.3 ± 2.9 (5/6)	ND (4)
110	11.1 ± 0.7 (6/6)	33.3 ± 2.1 (6/6)	13.7 ± 1.4 (5/5)	23.3 ± 4.1 (6/6)	2.7 ± 0.9 (2/4)
114	1.3 ± 0.3 (1/6)	2.9 ± 0.8 (3/6)	1.7 ± 0.6 (1/5)	6.3 ± 4.9 (1/6)	ND (4)
118	16.7 ± 3.6 (6/6)	53.7 ± 4.3 (6/6)	21.5 ± 4.4 (5/5)	52.3 ± 6.9 (6/6)	1.5 ± 0.5 (1/4)
123	2.3 ± 0.8 (2/6)	ND (6)	ND (5)	5.7 ± 2.8 (2/6)	ND (4)
126	ND (6)	ND (6)	ND (5)	10.3 ± 5.9 (2/6)	ND (4)
128	4.8 ± 0.5 (6/6)	10.9 ± 0.5 (6/6)	4.9 ± 0.4 (5/5)	26.5 ± 3.5 (6/6)	2.2 ± 1.0 (1/4)
138	63.1 ± 12.9 (6/6)	111.9 ± 8.9 (6/6)	48.3 ± 6.8 (5/5)	107.9 ± 29.8 (5/6)	3.9 ± 2.6 (1/4)
141	8.9 ± 1.4 (6/6)	16.4 ± 1.1 (6/6)	4.4 ± 2.0 (4/5)	18.3 ± 7.6 (5/6)	1.1 ± 0.1 (1/4)
149	20.5 ± 2.3 (6/6)	40.0 ± 3.6 (6/6)	15.1 ± 2.4 (5/5)	37.9 ± 17.8 (5/6)	2.7 ± 1.4 (1/4)
151	14.1 ± 2.1 (6/6)	19.9 ± 1.3 (6/6)	8.8 ± 0.8 (5/5)	19.6 ± 5.2 (5/6)	0.2 ± 0.1 (2/4)
153	72.8 ± 17.9 (6/6)	118.7 ± 10.9 (6/6)	47.5 ± 6.2 (5/5)	135.0 ± 36.6 (5/6)	1.9 ± 0.8 (1/4)
156	4.9 ± 1.6 (4/6)	8.6 ± 1.5 (5/6)	4.1 ± 0.8 (5/5)	13.6 ± 3.5 (5/6)	ND (4)
157	1.7 ± 0.6 (1/6)	ND (6)	ND (5)	ND (6)	ND (4)
158	13.5 ± 2.4 (6/6)	18.3 ± 1.9 (6/6)	11.1 ± 2.1 (5/5)	28.3 ± 10.1 (6/6)	ND (4)
167	2.7 ± 0.5 (6/6)	9.3 ± 0.5 (6/6)	4.8 ± 0.5 (5/5)	9.5 ± 2.0 (6/6)	ND (4)
168 + 132	4.3 ± 0.8 (5/6)	6.1 ± 0.4 (6/6)	2.7 ± 0.6 (5/5)	21.9 ± 13.9 (6/6)	1.5 ± 0.5 (3/4)
169	ND (6)	9.5 ± 7.7 (1/6)	ND (5)	ND (6)	ND (4)
170	16.3 ± 3.9 (6/6)	19.9 ± 3.7 (5/6)	8.0 ± 1.2 (5/5)	44.1 ± 8.0 (6/6)	ND (4)
177	15.6 ± 2.4 (6/6)	17.3 ± 1.2 (6/6)	7.8 ± 1.3 (5/5)	31.2 ± 11.3 (6/6)	2.7 ± 1.5 (1/4)
180	33.9 ± 8.2 (6/6)	35.7 ± 7.1 (5/6)	18.5 ± 2.9 (5/5)	59.7 ± 17.7 (6/6)	1.4 ± 0.3 (1/4)
183	15.8 ± 1.9 (6/6)	18.3 ± 1.1 (6/6)	9.3 ± 1.3 (5/5)	29.3 ± 7.2 (6/6)	ND (4)
187	34.4 ± 7.4 (6/6)	38.4 ± 2.9 (6/6)	17.2 ± 1.6 (5/5)	49.0 ± 13.9 (5/6)	1.5 ± 0.9 (2/4)
194	3.2 ± 1.9 (1/6)	ND (6)	ND (5)	4.6 ± 3.3 (1/6)	ND (4)
200	2.9 ± 0.8 (3/6)	1.5 ± 0.3 (1/6)	1.6 ± 0.3 (2/5)	2.2 ± 1.1 (1/6)	ND (4)
201	4.5 ± 0.4 (1/6)	6.1 ± 3.1 (2/6)	2.2 ± 0.6 (2/5)	8.6 ± 4.4 (2/6)	ND (4)
206	1.7 ± 0.7 (1/6)	ND (6)	ND (5)	ND (6)	ND (4)
209	1.6 ± 0.4 (2/6)	ND (6)	ND (5)	ND (6)	ND (4)
Total PCBs	**445.2 ± 79.5 (6)[b]**	**840.0 ± 56.5 (6)[c]**	**381.1 ± 43.0 (5)[b]**	**906.1 ± 175.1 (6)[c]**	**19.5 ± 9.1 (4)[a]**

and 200 showing nearly significant correlation ($p = 0.06$–0.08). There were no significant relationships between hepatic PCBs and plasma T3 concentrations; however T3/T4 ratios generally showed a positive relationship with PCBs, and were significantly correlated ($R = 0.35$–0.48, $p < 0.05$) with PCBs 49, 52, 97, 99, 101, 105 (co-planar), 110, and 118 (co-planar).

In sculpin, which had greater overall PCB exposures, thyroid endocrine parameters exhibited more significant correlations (Table 5). Plasma T4 concentrations were inversely correlated with 14 PCB congeners ($R = -0.43$ to -0.64, $p < 0.05$), including PCBs 28, 31, 49, 52, 87, 95, 97, 99, 101, 105 (co-planar), 110, 118 (co-planar), 128, and 157 (co-planar). Seven of these PCBs were similarly correlated in surfperch, all of which showed higher correlation coefficients in sculpin. Plasma T3 in sculpin was positively correlated with 15 PCB congeners ($R = 0.42$–0.60, $p < 0.05$), including PCBs 28, 44, 52, 66, 70, 74, 87, 95, 97, 99, 101, 105, 128, 157, and 177; all of these PCBs were also correlated with T3/T4 ratio ($R = 0.42$–0.76, $p < 0.05$; Table 5), and ten were from the group (of 14) that was inversely correlated with T4 (28, 52, 87, 95, 97, 99, 101, 105, 128, and 157). PCBs 52, 95, 101, 105, 128, and 157 were correlated with all thyroid parameters (inverse with T4, positive with T3 and T3/T4 ratio) in sculpin.

In contrast to the numerous significant correlations identified for PCBs, there were no correlations between hepatic PAH concentrations and T4 or T3/T4 ratio in surfperch (Table 6); however, three PAHs were positively correlated with T3 concentrations (acenaphthene, fluorene, and phenanthrene; $p < 0.05$). In sculpin, T3 and T3/T4 ratio showed significant positive correlations with acenaphthene, anthracene and acenaphthylene ($R = 0.47$–0.64). Acenaphthene was the only PAH to be correlated with all three thyroid parameters, inversely with T4 ($R = -0.54$) and positively with T3 ($R = 0.76$) and T3/T4 ratio ($R = 0.89$). In contrast, pyrene was positively correlated with plasma T4 ($R = 0.61$), but not correlated with the other measures.

Chordanes were not significantly related to plasma T4 concentrations in either species (Table 6), and they were also not correlated with T3 or T3/T4 ratio in the sculpin. In surfperch, chlordanes were significantly correlated with T3 ($R = 0.42$–0.49, $p < 0.03$) and T3/T4 ratio ($R = 0.62$–0.75, $p < 0.001$). Three DDT metabolites were positively correlated with plasma T3 in surfperch ($R = 0.44$–0.53, $p < 0.02$), but not with T3/T4 ratio or T4 (Table 6). In sculpin, all detected DDT congeners were positively correlated with T3 ($R = 0.53$–0.69, $p < 0.01$) and T3/T4 ratio ($R = 0.53$–0.72, $p < 0.01$). A negative correlation between T4 and 4,4-DDE ($R = -0.451$, $p = 0.03$) and a similar trend for the other DDT congeners ($p = 0.07$–0.08) was also observed in sculpin.

4. Discussion

The findings of the present study and several former studies (Fairey et al., 1997; Davis et al., 2004, 2007; Oros et al., 2005, 2007; Hunt et al., 2008) have consistently demonstrated that wild fish residing in San Francisco Bay Area habitats are exposed to a variety of anthropogenic contaminants. It has not been well understood,

Table 2
Hepatic PCB concentrations (mean ng/g ww ± SE) in Pacific staghorn sculpin sampled from Redwood City (RED), San Leandro Bay (SLB), Richmond Lauritzen Channel (RLC), and Oakland Inner Harbor (OAK). In parentheses, the numbers of samples with concentrations above the MDL are given in the numerator, with denominator indicating the number of individuals measured. PCB congeners 008, 018, 033, 037, 081, 119, 126, 169, 189, 201, 206, and 209 were not detected in any scuplin samples.

Congener	RED	SLB	RLC	OAK
028	1.5 ± 1.4 (1/6)	3.2 ± 3.1 (3/7)	11.9 ± 2.5 (4/4)	4.3 ± 2.7 (2/5)
031	1.5 ± 1.4 (1/6)	4.9 ± 2.3 (3/7)	18.0 ± 3.3 (4/4)	14.7 ± 6.7 (3/5)
044	5.3 ± 3.4 (2/6)	3.3 ± 2.1 (2/7)	20.3 ± 3.7 (4/4)	ND (5)
049	8.1 ± 3.0 (4/6)	14.8 ± 3.2 (6/7)	23.6 ± 3.7 (4/4)	16.8 ± 5.4 (5/5)
052	11.6 ± 3.5 (4/6)	20.8 ± 5.5 (6/7)	38.5 ± 7.3 (4/4)	29.9 ± 5.6 (5/5)
066	12.1 ± 7.3 (3/6)	11.9 ± 3.6 (5/7)	31.1 ± 5.8 (4/4)	3.7 ± 2.1 (2/5)
070	9.3 ± 4.5 (3/6)	9.7 ± 3.2 (5/7)	23.8 ± 4.1 (4/4)	1.7 ± 1.0 (2/5)
074	8.8 ± 4.5 (3/6)	6.3 ± 1.8 (5/7)	20.2 ± 3.5 (4/4)	1.5 ± 0.8 (2/5)
077	3.0 ± 2.7 (1/6)	ND (7)	1.2 ± 1.0 (1/4)	ND (5)
087	3.8 ± 3.5 (1/6)	10.8 ± 3.7 (4/7)	27.8 ± 4.3 (4/4)	13.8 ± 6.7 (3/5)
095	18.1 ± 8.1 (4/6)	15.8 ± 5.0 (5/7)	43.0 ± 7.3 (4/4)	25.7 ± 5.1 (5/5)
097	3.8 ± 3.5 (1/6)	11.1 ± 3.9 (4/7)	32.4 ± 4.1 (4/4)	31.0 ± 10.9 (4/5)
099	16.4 ± 5.0 (4/6)	36.1 ± 7.8 (7/7)	46.1 ± 5.9 (4/4)	37.4 ± 11.6 (4/5)
101	21.7 ± 6.8 (4/6)	69.6 ± 13.2 (7/7)	77.3 ± 9.1 (4/4)	60.1 ± 19.3 (4/5)
105	1.5 ± 1.4 (1/6)	9.7 ± 3.4 (5/7)	22.6 ± 2.2 (4/4)	7.8 ± 4.3 (2/5)
110	16.8 ± 5.5 (4/6)	41.9 ± 7.2 (7/7)	47.7 ± 5.7 (4/4)	43.2 ± 7.7 (5/5)
114	ND (6)	ND (7)	ND (4)	1.2 ± 1.1 (1/6)
118	17.1 ± 8.0 (2/6)	49.8 ± 7.3 (7/7)	56.4 ± 2.7 (4/4)	47.8 ± 15.4 (4/5)
123	ND (6)	ND (7)	6.7 ± 2.3 (3/4)	ND (5)
128	5.7 ± 3.3 (2/6)	9.4 ± 2.4 (6/7)	25.1 ± 2.4 (4/4)	7.1 ± 2.7 (3/5)
138	122.1 ± 21.4 (6/6)	108.1 ± 15.8 (7/7)	112.2 ± 7.0 (4/4)	137.1 ± 23.4 (5/5)
141	7.7 ± 4.5 (2/6)	7.9 ± 2.7 (5/7)	16.3 ± 3.3 (4/4)	5.3 ± 3.0 (2/5)
149	65.7 ± 12.2 (5/6)	51.1 ± 7.9 (7/7)	60.2 ± 6.4 (4/4)	67.5 ± 10.5 (5/5)
151	23.0 ± 4.5 (6/6)	19.3 ± 2.5 (7/7)	25.7 ± 5.1 (4/4)	27.8 ± 6.2 (5/5)
153	112.1 ± 16.5 (6/6)	119.4 ± 18.9 (7/7)	119.6 ± 8.2 (4/4)	154.5 ± 25.3 (5/5)
156	2.3 ± 2.1 (1/6)	6.0 ± 2.5 (4/7)	12.9 ± 1.8 (4/4)	5.5 ± 3.0 (2/5)
157	ND (6)	1.0 ± 0.8 (1/7)	6.6 ± 1.2 (4/4)	ND (5)
158	42.5 ± 7.6 (6/6)	27.6 ± 3.2 (7/7)	22.2 ± 2.9 (4/4)	53.4 ± 16.7 (5/5)
167	8.3 ± 5.7 (2/6)	8.5 ± 2.4 (6/7)	12.4 ± 0.8 (4/4)	7.1 ± 2.7 (3/5)
168 + 132	19.8 ± 2.5 (6/6)	7.3 ± 1.5 (6/7)	10.8 ± 1.2 (4/4)	14.3 ± 3.2 (5/5)
170	14.8 ± 8.7 (2/6)	21.6 ± 5.6 (6/7)	13.2 ± 4.0 (3/4)	27.9 ± 7.5 (4/5)
177	11.5 ± 7.1 (2/6)	17.9 ± 4.0 (6/7)	29.1 ± 4.4 (4/4)	17.9 ± 5.4 (4/5)
180	58.8 ± 5.9 (6/6)	46.7 ± 6.7 (7/7)	55.3 ± 6.8 (4/4)	80.7 ± 14.7 (5/5)
183	33.9 ± 4.4 (6/6)	19.5 ± 2.9 (7/7)	26.6 ± 0.8 (4/4)	30.2 ± 6.3 (5/5)
187	62.1 ± 9.3 (6/6)	44.7 ± 6.1 (7/7)	57.8 ± 4.8 (4/4)	55.9 ± 10.3 (5/5)
194	ND (6)	3.3 ± 3.1 (1/7)	17.9 ± 15.5 (1/4)	5.3 ± 4.7 (1/5)
200	ND (6)	ND (7)	ND (4)	1.9 ± 1.7 (1/5)
Total PCBs	**752.3 ± 94.9 (6)**	**839.0 ± 144.4 (7)**	**1872.9 ± 614.7 (4)**	**1040.3 ± 159.2 (5)**

Table 3
Hepatic concentrations of pesticides and PAHs (mean ng/g ww ± SE) in shiner surfperch sampled from Redwood City (RED), San Leandro Bay (SLB), Richmond Lauritzen Channel (RLC), Oakland Inner Harbor (OAK), and Santa Catalina Island (CAT). In parentheses, the numbers of samples with concentrations above the MDL are given in the numerator, with denominator indicating the number of individuals measured. Different (a–c) letters represent significant differences in total chlordanes, total DDTs, or total PAHs. The following chemicals were not detected in any surfperch samples: 4,4′-DDT, aldrin, BHC-α, -β, -γ, and -δ, DCPA (dacthal), dicofol, dieldrin, endosulfan-I and -II, endrin, endrin aldehyde, endrin ketone, heptachlor, heptachlor epoxide, methoxychlor, mirex, oxychlordane, perthane, 1-methylphenanthrene, benz[a]anthracene, benzo[a]pyrene, benzo[e]pyrene, benzo[g,h,i]perylene, chrysene, dibenz[a,h]anthracene, dibenzothiophene, fluoranthene, indeno[1,2,3-c,d]pyrene, perylene.

	RED	SLB	RLC	OAK	CAT
Pesticides					
Gamma chlordane	3.0 ± 0.41 (3/6)	12.3 ± 1.4 (6/6)	4.8 ± 1.4 (4/5)	6.0 ± 0.8 (6/6)	ND (4)
Alpha chlordane	8.7 ± 2.6 (4/6)	24.1 ± 2.6 (6/6)	7.4 ± 1.8 (4/5)	11.4 ± 1.7 (6/6)	ND (4)
trans-Nonachlor	10.8 ± 3.8 (4/6)	25.4 ± 2.2 (6/6)	6.8 ± 1.9 (4/5)	14.7 ± 3.3 (6/6)	ND (4)
cis-Nonachlor	8.0 ± 2.8 (4/6)	18.2 ± 0.7 (6/6)	4.6 ± 1.2 (4/5)	9.6 ± 2.2 (5/6)	ND (4)
Endosulfan sulfate	ND (6)	ND (6)	ND (5)	ND (6)	ND (4)
4,4-DDE	58.9 ± 14.1 (6/6)	80.7 ± 5.3 (6/6)	293.8 ± 61.1 (5/5)	81.5 ± 20.9 (6/6)	16.4 ± 5.2 (4/4)
4,4-DDD	8.8 ± 4.2 (3/6)	35.8 ± 4.0 (6/6)	379.3 ± 89.4 (5/5)	28.1 ± 4.3 (6/6)	ND (4)
2,4-DDT	ND (6)	ND (6)	7.9 ± 3.5 (1/5)	0.81 ± 0.7 (1/6)	ND (4)
2,4-DDD	ND (6)	3.1 ± 2.8 (1/6)	84.4 ± 20.6 (5/5)	ND (6)	ND (4)
4,4-DDT	ND (6)	ND (6)	47.8 ± 18.0 (4/5)	ND (6)	ND (4)
Total chlordanes	**30.6 ± 9.9 (6)[a,b]**	**79.9 ± 5.4 (6)[c]**	**23.6 ± 6.3 (6)[a]**	**41.8 ± 5.6 (6)[b]**	**ND (4)**
Total DDTs	**67.7 ± 18.1 (6)[b]**	**119.6 ± 10.5 (6)[b]**	**813.1 ± 192.6 (6)[c]**	**110.4 ± 23.9 (6)[b]**	**16.4 ± 5.2 (4)[b]**
PAHs					
Naphthalene	141.5 ± 38.9 (6/6)	111.0 ± 13.0 (6/6)	67.7 ± 11.7 (5/5)	13.8 ± 4.7 (4/6)	20.0 ± 10.0 (2/4)
2-Methylnaphthalene	34.6 ± 7.0 (6/6)	42.7 ± 6.4 (6/6)	38.9 ± 2.8 (5/5)	17.3 ± 3.4 (6/6)	8.82 ± 3.8 (3/4)
1-Methylnaphthalene	40.3 ± 8.7 (6/6)	32.7 ± 4.8 (6/6)	28.1 ± 2.9 (5/5)	17.2 ± 3.9 (6/6)	10.1 ± 1.9 (4/4)
2,6-Dimethylnaphthalene	ND (6)	ND (6)	2.5 ± 2.2 (1/5)	ND (6)	2.96 ± 1.7 (1/4)
2,3,5-Trimethylnaphthlene biphenyl	19.5 ± 17.8 (1/6)	7.4 ± 5.8 (1/6)	ND (5)	11.4 ± 3.2 (4/6)	2.57 ± 0.98 (2/4)
Acenaphthylene	ND (6)	ND (6)	7.2 ± 3.2 (3/5)	ND (6)	ND (4)
Acenaphthene	7.2 ± 3.6 (2/6)	36.5 ± 8.9 (5/6)	97.5 ± 14.7 (5/5)	ND (6)	ND (4)
Flurorene	ND (6)	ND (6)	42.5 ± 5.2 (5/5)	10.1 ± 3.3 (5/6)	ND (4)
Phenanthrene	ND (6)	15.9 ± 9.5 (2/6)	88.0 ± 2.7 (5/5)	28.7 ± 9.8 (6/6)	2.29 ± 1.1 (1/5)
Anthracene	ND (6)	1.2 ± 1.1 (1/6)	20.2 ± 2.2 (5/5)	13.4 ± 2.2 (6/6)	2.87 ± 0.88 (4/4)
Fluoranthene	ND (6)	ND (6)	ND (5)	50.5 ± 5.8 (6/6)	0.89 ± 0.10 (1/4)
Pyrene	ND (6)	ND (6)	ND (5)	12.9 ± 4.6 (6/6)	2.2 ± 1.1 (2/4)
Benzo[b]fluoranthene	ND (6)	ND (6)	ND (5)	6.3 ± 4.8 (1/6)	ND (4)
Benzo[k]fluoranthene	ND (6)	ND (6)	ND (5)	9.9 ± 9.1 (1/6)	ND (4)
Total PAHs	**242.4 ± 64.7 (6)[b]**	**246.4 ± 19.1 (6)[b]**	**392.6 ± 11.1 (5)[c]**	**191.6 ± 33.7 (5)[b]**	**28.6 ± 16.7 (4)[a]**

Table 4
Hepatic concentrations of pesticides and PAHs (mean ng/g ww ± SE) in Pacific staghorn sculpin sampled from Redwood City (RED), San Leandro Bay (SLB), Richmond Lauritzen Channel (RLC), and Oakland Inner Harbor (OAK). In parentheses, the numbers of samples with concentrations above the MDL are given in the numerator, with denominator indicating the number of individuals measured. Different (a–c) letters represent significant differences in total chlordanes, total DDTs, or total PAHs. The following chemicals were not detected in any sculpin samples: aldrin, BHC-α, -β, -γ, and -δ, DCPA (dacthal), dicofol, dieldrin, endosulfan-I and -II, endrin, endrin aldehyde, endrin ketone, heptachlor, heptachlor epoxide, methoxychlor, mirex, oxychlordane, perthane, 1-methylphenanthrene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[k]fluoranthene benzo[e]pyrene, benzo[g,h,i]perylene, chrysene, dibenz[a,h]anthracene, dibenzothiophene, fluorene, indeno[1,2,3-c,d]pyrene, perylene.

	RED	SLB	RLC	OAK
Pesticides				
Gamma chlordane	9.3 ± 4.5 (3/6)	18.0 ± 5.7 (4/7)	16.2 ± 2.8 (4/4)	13.6 ± 7.6 (3/5)
Alpha chlordane	12.6 ± 5.2 (3/6)	29.9 ± 8.7 (4/7)	24.2 ± 3.4 (4/4)	20.3 ± 9.7 (3/5)
trans-Nonachlor	17.0 ± 8.3 (3/6)	30.1 ± 8.3 (5/7)	18.3 ± 2.7 (4/4)	32.3 ± 14.6 (4/5)
cis-Nonachlor	16.0 ± 6.8 (3/6)	21.1 ± 4.6 (5/7)	28.9 ± 7.1 (4/4)	19.0 ± 6.5 (4/5)
Endosulfan sulfate	ND (6)	ND (7)	ND (5)	101.8 ± 91.1 (1/5) (509.2)
4,4-DDE	42.9 ± 15.9 (4/6)	88.0 ± 21.4 (5/7)	612.0 ± 83.6 (4/4)	86.1 ± 22.6 (4/5)
4,4-DDD	ND (6)	47.9 ± 12.6 (5/7)	1072.9 ± 302.9 (4/4)	29.2 ± 11.2 (3/5)
2,4-DDT	ND (6)	ND (6)	49.1 ± 18.8 (4/4)	0.80 ± 0.6 (1/5)
2,4-DDD	ND (6)	3.2 ± 2.9 (1/7)	104.0 ± 29.6 (4/4)	ND (5)
4,4-DDT	ND (6)	ND (7)	88.8 ± 39.0 (4/4)	ND (5)
2,4-DDE	ND (6)	ND (7)	18.5 ± 4.7 (4/4)	ND (5)
Total chlordanes	**54.9 ± 23.5 (3/6)[a]**	**99.2 ± 26.9 (5/7)[b]**	**87.7 ± 15.7 (4)[b]**	**85.2 ± 38.0 (4/5)[b]**
Total DDTs	**42.9 ± 42.9 (4/6)[a]**	**139.0 ± 34.3 (5/7)[b]**	**1945.4 ± 471.8 (4)[c]**	**217.1 ± 101.8 (4/5)[b]**
PAHs				
Naphthalene	193.0 ± 29.9 (6/6)	167.6 ± 30.5 (7/7)	135.6 ± 31.1 (4/4)	377.8 ± 177.6 (5/5)
2-Methylnaphthalene	34.6 ± 18.5 (6/6)	69.7 ± 9.2 (7/7)	85.6 ± 11.1 (4/4)	164.2 ± 69.2 (5/5)
1-Methylnaphthalene	95.7 ± 11.1 (6/6)	53.6 ± 13.8 (6/7)	48.3 ± 9.2 (4/4)	99.3 ± 48.9 (5/5)
2,6-Dimethylnaphthalene	ND (6)	ND (7)	15.1 ± 8.3 (2/4)	ND (5)
2,3,5-Trimethylnaphthlene	147.9 ± 135.1 (1/6) (888.0)	ND (7)	ND (4)	ND (5)
Biphenyl	63.9 ± 10.1 (6/6)	7.4 ± 5.8 (1/6)	49.9 ± 13.4 (4/4)	ND (5)
Acenaphthylene	4.5 ± 2.9 (2/6)	ND (6)	54.6 ± 12.2 (4/4)	ND (6)
Acenaphthene	6.9 ± 4.3 (2/6)	27.1 ± 9.0 (4/7)	128.6 ± 24.5 (4/4)	27.9 ± 12.0 (3/5)
Flurorene	25.4 ± 18.4 (2/5)	ND (6)	56.8 ± 7.9 (4/4)	12.4 ± 11.1 (1/5)
Phenanthrene	106.3 ± 26.8 (6/6)	30.2 ± 13.9 (3/7)	61.9 ± 6.8 (4/4)	17.4 ± 15.6 (1/5)
Anthracene	15.6 ± 12.8 (2/6)	6.1 ± 3.8 (2/7)	156.7 ± 37.3 (4/4)	ND (5)
Fluoranthene	81.7 ± 16.6 (6/6)	12.0 ± 11.1 (1/7)	75.6 ± 12.2 (4/4)	ND (5)
Pyrene	94.7 ± 15.6 (6/6)	ND (6)	33.9 ± 2.7 (4/4)	ND (5)
Total PAHs	**1009.1 ± 172.1 (6)[b]**	**366.3 ± 48.3 (7)[a]**	**902.6 ± 141.0 (4)[b]**	**698.9 ± 279.7 (5)[b]**

Table 5
Correlation coefficients (*R*) and *p*-values of the relationships among hepatic concentrations of PCB congeners (*identified in center column*) and thyroid endocrine parameters (*T4 and T3 concentrations*, *T3/T4 ratio*) in shiner surfperch (*left*) and Pacific staghorn sculpin (*right*). Asterisk indicates that the Pearson correlation coefficient is significant ($p < 0.05$); dagger symbol (†) indicates $p = 0.5–0.1$.

Shiner surfperch						PCB congeners	Pacific staghorn sculpin					
T4		T3		T3/T4			T4		T3		T3/T4.	
R	*p*	*R*	*p*	*R*	*p*		*R*	*p*	*R*	*P*	*R*	*p*
−0.37*	0.04	−0.14	0.46	0.23	0.22	**028**	**−0.44***	0.04	**0.42***	0.05	**0.54***	0.01
−0.39*	0.03	−0.09	0.61	0.25	0.17	**031**	**−0.50***	0.02	0.22	0.33	0.39†	0.08
−0.43*	0.02	−0.06	0.73	0.33†	0.07	**044**	−0.40†	0.07	**0.59***	0.01	**0.67***	0.01
−0.37*	0.04	−0.11	0.56	**0.39***	0.03	**049**	**−0.62***	0.01	0.27	0.22	**0.42***	0.05
−0.32†	0.08	−0.06	0.77	**0.41***	0.02	**052**	**−0.64***	0.01	**0.46***	0.03	**0.59***	0.01
−0.34†	0.06	−0.20	0.27	0.17	0.36	**066**	−0.35	0.12	**0.46***	0.03	**0.53***	0.01
−0.27	0.14	−0.23	0.22	0.17	0.36	**070**	−0.37†	0.09	**0.55***	0.01	**0.58***	0.01
−0.05	0.78	−0.25	0.18	−0.05	0.77	**074**	−0.30	0.18	**0.48***	0.02	**0.51***	0.02
−0.46*	0.01	−011	0.58	0.33†	0.07	**087**	**−0.50***	0.02	**0.56***	0.01	**0.59***	0.01
−0.41*	0.02	−0.22	0.25	0.30†	0.10	**095**	**−0.51***	0.02	**0.48***	0.02	**0.59***	0.01
−0.45*	0.01	0.01	0.95	**0.35***	0.05	**097**	**−0.45***	0.03	**0.44***	0.04	**0.44***	0.04
−0.23	0.22	−0.01	0.98	**0.41***	0.02	**099**	**−0.42***	0.05	**0.42***	0.05	**0.48***	0.02
−0.32†	0.08	0.03	0.86	**0.48***	0.01	**101**	**−0.50***	0.02	**0.45***	0.03	**0.49***	0.02
−0.32†	0.08	0.07	0.73	**0.42***	0.02	**105**	**−0.50***	0.02	**0.55***	0.01	**0.66***	0.01
−0.44*	0.02	−0.06	0.76	**0.47***	0.01	**110**	**−0.56***	0.01	0.35	0.11	0.41†	0.06
−0.22	0.24	0.05	0.80	**0.41***	0.02	**118**	**−0.50***	0.02	0.41†	0.06	0.41†	0.06
−0.17	0.35	−0.29	0.120	0.02	0.91	**128**	**−0.47***	0.03	**0.60***	0.01	**0.69***	0.01
−0.13	0.47	0.06	0.74	0.32†	0.08	**138**	0.01	0.97	−0.01	0.97	0.01	0.96
−0.17	0.36	−0.04	0.85	0.27	0.15	**141**	−0.28	0.21	0.33	0.13	**0.45***	0.04
−0.26	0.16	−0.08	0.68	0.29	0.12	**149**	−0.16	0.47	0.13	0.58	0.12	0.60
−0.08	0.65	−0.06	0.77	0.22	0.23	**151**	−0.11	0.62	0.08	0.71	0.13	0.56
−0.06	0.76	0.03	0.87	0.24	0.20	**153**	−0.16	0.48	−0.01	0.98	0.06	0.78
−0.13	0.48	0.05	0.79	0.15	0.43	**156**	−0.40†	0.06	0.41†	0.06	**0.49***	0.02
−0.30	0.11	−0.30	0.11	−0.001	0.96	**157**	**−0.51***	0.02	**0.55***	0.01	**0.76***	0.01
−0.13	0.48	−0.13	0.49	0.04	0.86	**158**	0.02	0.92	−0.37†	0.09	−0.29	0.20
−0.43*	0.02	0.03	0.87	0.37	0.04	**167**	−0.19	0.41	0.23	0.29	0.23	0.31
−0.18	0.35	−0.09	0.63	0.01	0.96	**168 + 132**	0.34	0.13	−0.19	0.39	−0.19	0.39
0.14	0.46	0.04	0.81	0.05	0.80	**170**	−0.08	0.73	0.19	0.39	0.03	0.89
−0.11	0.54	−0.12	0.52	0.04	0.81	**177**	−0.36†	0.11	**0.50***	0.02	**0.43***	0.04
−0.01	0.96	−0.05	0.80	0.07	0.70	**180**	−0.07	0.75	−0.08	0.73	−0.05	0.82
−0.07	0.70	−0.15	0.42	0.06	0.76	**183**	0.24	0.28	−0.10	0.67	−0.09	0.69
−0.06	0.77	−0.10	0.61	0.10	0.60	**187**	−0.09	0.69	−0.05	0.82	−0.01	0.98
−0.13	0.49	−0.20	0.29	−0.11	0.57	**194**	−0.19	0.40	0.28	0.21	0.21	0.35
−0.36†	0.06	0.07	0.71	−0.25	0.18	**200**	ND	ND	ND	ND	ND	ND
−0.25	0.17	−0.09	0.63	0.25	0.18	**Total PCBs**	−0.39†	0.072	**0.80***	0.01	**0.81***	0.01

however, the extent to which such contaminant exposures may be related to phenotypic effects. This study provided an initial characterization of potential endocrine-related effects and their relationship to accumulated contaminants in two indigenous fish species from this region.

Dependent upon the environmental location studied, plasma T4 concentrations exhibited significant differences in both surfperch and sculpin. In surfperch residing in RED, BDB, RCH, and the remote site at CAT, T4 concentrations were comparable to other fish species under control conditions as reported in the literature (e.g., LeRoy et al., 2006; Yang et al., 2007). In contrast, surfperch sampled from OAK, SLB, and TOB exhibited significantly reduced T4. In sculpin, a similar spatial pattern of differences was observed (higher T4 at RED and BDB, lower T4 at OAK, SLB and TOB). However, the sculpin exhibited additional site-associated reductions in T4 at RCH, RLC, and SPB, suggesting this species may be more sensitive to environmental effects as compared with surfperch. Nonetheless, both species exhibited several consistent spatial differences in T4, including some over two years of study (e.g., reduced T4 at OAK). As discussed further below, differences in T4 were significantly related to environmental contaminant exposures.

Both species also exhibited significant differences in T3 and T3/T4 ratio across the different study locations. Relatively low T3 and T3/T4 ratio were observed at some locations (e.g., both species at RED, sculpin at TOB), while relatively values were observed at other locations (e.g., surfperch at SLB, sculpin at RLC). Such differences in T3 and T3/T4 ratio may point to environment-related alterations in the peripheral conversion of T4 into T3.

In all vertebrates including fish, the bulk of plasma T3 concentrations are derived from peripheral conversions of T4 by 5′-deiodinases (Eales et al., 1999; Plohman et al., 2002; Eales, 2006; Zoeller et al., 2007). PCBs, pesticides and other chemicals may alter 5′-deiodinase activity in fish, which can lead to increased T3 and reduced T4 (Adams et al., 2000; Brown et al., 2004a; Coimbra et al., 2005; Picard-Aitken et al., 2007). At RLC, sculpin had the highest PCB exposures (total PCBs >1870 ng/g) and the highest T3/T4 ratio. In surfperch at SLB, high T3/T4 ratio was associated with higher levels of a distinct profile of hepatic PCB congeners (several lower chlorinated PCBs) as compared with surfperch from OAK. In addition, significant correlations between T3/T4 ratio and PCBs, as well as some other chemicals, was observed in both species. We therefore hypothesize that certain locations in the San Francisco Bay Area may be associated with contaminant effects on peripheral deiodination of T4 in fish. In future studies, it will be worthwhile to measure fish deiodinase activities in specific environmental locations (e.g., those identified with high T3/T4 ratio in fish) and evaluate relationships with exposures to specific contaminants. Deiodination measures have been recommended in a suite of assays to screen for thyroid disruption in fish (Eales et al., 1999).

Laboratory-based studies are increasingly demonstrating direct effects of selected contaminant compounds on thyroid endocrine parameters in fish. In lake trout (*Salvelinus namaycush*), chronic exposure to co-planar (dioxin-like) PCB 126 reduces plasma T4

Table 6
Correlation coefficients (R) and p-values (p) of the significant relationships among thyroid endocrine parameters (*T4 and T3 concentrations*, *T3/T4 ratio*) and hepatic concentrations of PAHs (panel A) and pesticides (panel B) in shiner surfperch and Pacific staghorn sculpin. Asterisk indicates that the Pearson correlation coefficient is significant ($p < 0.05$), dagger symbol (†) indicates $p = 0.5–0.1$. The PAHs and pesticides that were measured are listed in Tables 3 and 4; correlation results are given only for values with $p \leq 0.10$.

	T4		T3		T3/T4	
	R	*p*	*R*	*p*	*R*	*p*
A: PAHs						
Shiner surfperch						
Biphenyl	0.07	0.73	−0.36†	0.06	−0.14	0.48
Acenaphthene	0.05	0.81	**0.64***	0.01	0.18	0.35
Fluorine	0.22	0.27	**0.47***	0.01	−0.19	0.34
Phenanthrene	0.07	0.72	**0.52***	0.01	−0.03	0.87
Pacific staghorn sculpin						
Biphenyl	0.42†	0.06	0.10	0.64	0.18	0.41
Acenapthylene	−0.38†	0.09	**0.75***	0.01	**0.81***	0.01
Acenaphthene	**−0.54***	0.01	**0.77***	0.01	**0.89***	0.01
Anthracene	−0.39†	0.07	**0.75***	0.01	**0.80***	0.01
Pyrene	**0.61***	0.01	−0.10	0.66	−0.11	0.63
B: Pesticides						
Shiner surfperch						
Alpha chlordane	−0.30	0.16	**0.42***	0.03	**0.75***	0.01
Gamma chlordane	−0.27	0.16	**0.49***	0.01	**0.71***	0.01
trans-Nonachlor	−0.21	0.29	**0.45***	0.02	**0.65***	0.01
cis-Nonachlor	−0.19	0.32	**0.44***	0.02	**0.62***	0.01
Total chlordanes	−0.26	0.19	**0.47***	0.01	**0.72***	0.01
4,4-DDT	−0.19	0.33	**0.35**†	0.07	−0.12	0.53
4,4-DDD	0.17	0.38	**0.47***	0.01	−0.08	0.70
2,4-DDD	0.19	0.34	**0.44***	0.02	−0.09	0.66
4,4-DDE	0.19	0.33	**0.53***	0.01	−0.03	0.87
Total DDTs	0.19	0.34	**0.48***	0.01	−0.07	0.73
Pacific staghorn sculpin						
cis-Nonachlor	−0.22	0.33	0.30	0.18	0.39†	0.08
4,4-DDT	−0.29	0.19	**0.58***	0.01	**0.55***	0.01
2,4-DDT	−0.32	0.14	**0.53***	0.01	**0.53***	0.01
4,4-DDD	−0.39†	0.08	**0.61***	0.01	**0.63***	0.01
2,4-DDD	−0.39†	0.08	**0.62***	0.01	**0.68***	0.01
4,4-DDE	**−0.45***	0.03	**0.69***	0.01	**0.72***	0.01
2,4-DDE	−0.38†	0.08	**0.69***	0.01	**0.72***	0.01
Total DDTs	−0.40†	0.07	**0.64***	0.01	**0.65***	0.01

concentrations (Brown et al., 2004b). Similarly, Atlantic croaker (*Micropogonias undulates*) treated with the non-co-planar PCB 153 also shows reduced T4 concentrations, as well as reduced T3 concentrations (LeRoy et al., 2006). However, treatment with another co-planar congener, PCB 77, may have no effects in American plaice (*Hippoglossoides platessoides*; Adams et al., 2000) or in Atlantic croaker, except at an unrealistically high dose (100 μg/g body weight; LeRoy et al., 2006). In the present study, concentrations of PCBs 77 and 126 were mostly non-detectable in liver of surfperch and sculpin, although other co-planar PCBs were detected (e.g., PCBs 105, 118, 156). While hepatic concentrations of PCB 153 were relatively high (50–135 ng/g ww) in both species, this congener did not significantly correlate with thyroid endocrine parameters. However, other non-co-planar PCB congeners did show correlations. It was notable that many of the correlated PCB congeners were not necessarily the highest in tissue concentration in the fish. PCBs 138, 149, 153, 158, 180, and 187 were among the highest in liver, yet they were not significantly correlated with thyroid endocrine parameters. Thus, highly accumulated PCBs were not necessarily associated with altered thyroid status, a result that would not be readily evident by consideration of overall hepatic concentrations alone. This suggests that certain PCBs, if indeed causative in altering thyroid status, may act at relatively lower concentrations.

Of the correlated PCBs, five were co-planar PCBs (105, 118, 156, 157, 167), which presumably would activate aryl hydrocarbon receptor (AhR)-mediated responses in the fish (Safe, 1994; Van den Berg et al., 1988; Burgin et al., 2001; Arukwe and Nordbø, 2008). The effects of these responses may include reduced thyroid hormone levels resulting from induced hepatic biotransformation enzymes (UDPGT and sulfatase) and excretion (Kohn et al., 1996; Crofton, 2008), and/or changes in expression of genes involved in thyroid hormone production (Pocar et al., 2006). It has also been reported that activation of PCBs 105 and 118 by CYP1A may result in formation of agonists that bind to and interfere with thyroid hormone receptors (Gauger et al., 2007). While some co-planar PCBs were implicated in thyroid alterations in this study, it was clear that several non-co-planar PCBs were also significantly correlated with thyroid parameters in both species. Non-co-planar PCBs, some of which are prevalent in aquatic environments and in fish (including this study) may induce CYP 2B and 3A enzymes, but their mechanisms of action are far less clear (Safe, 1994; Hansen, 1998; Buckman et al., 2007). It has also been proposed that some PCBs may alter thyroid hormone levels via competitive binding to transthyretin (TTR), a T4-binding protein present in plasma (Brouwer et al., 1998; Purkey et al., 2004; Ucán-Marín et al., 2009). Such binding can displace T4 from TTR, altering T4 turnover and feedback signaling to the HPT axis. While the mechanism(s) of action are not known, results of the present study point to candidate PCB congeners implicated in thyroid disruption in the fish.

In contrast to the correlations between thyroid endocrine parameters and several PCB congeners, fewer relationships with PAHs were elucidated. In surfperch, no relationships with T4 or the T3/T4 ratio were found among the PAHs detected in liver, while only three PAHs were positively correlated with T3. In sculpin, which exhibited higher overall hepatic PAH concentrations, some

additional correlations emerged. Increasing PAH exposures would be expected to activate AhR-mediated pathways which, as discussed earlier, may possibly lead to thyroid endocrine system effects (Teles et al., 2005). However, the present study also identified several relationships between chlorinated pesticides and altered thyroid parameters in both species. In surfperch, the chlordanes were positively correlated with T3 and T3/T4 ratio and showed an inverse (non-significant) trend with T4 levels. In sculpin, a similar pattern of response was evident, although low variation in hepatic chlordane concentrations across the groups appears to have resulted in a lack of significance. DDT and DDT metabolites were positively correlated with T3 in both species, and in sculpin, T3/T4 ratio (positive) and T4 levels (inverse) also showed several significant correlations. The potential mechanisms by which chlorinated pesticides may exert effects on the thyroid endocrine system are not understood. Recent mammalian research suggests that DDT metabolites may interfere with TSH receptors on thyroid follicular cells (Rossi et al., 2007; Picchietti et al., 2009), while an older study reported that DDTs decreased iodine uptake leading to reduced T4 production by the thyroid gland (Goldman, 1981). DDT and DDT metabolites may also be associated with thyroid disruption in fish (Brown et al., 2004a,b; Boas et al., 2006; Schnitzler et al., 2008). In addition, a potential for DDT estrogenic actions in disrupting thyroid endocrine function may be suggested by some studies in which estrogenic compounds have been observed to interfere with TTR binding to T4 (e.g., Morgado et al., 2009). Given the relationships between chlorinated pesticides and thyroid parameters implied by this study, their potential mechanisms of action in the fish are of interest in future studies.

The results for TOB, a rural location with a small human population, were initially surprising, as it was expected that fish from this location would show thyroid hormone concentrations more comparable to those in fish from CAT and BDB. In both species, concentrations of T4 and T3 were at or near the lowest levels measured. Although hepatic contaminants were not measured in fish from TOB, most of the organic contaminants considered in the present study would not be expected to be present at elevated levels, with the possible exception of PAHs derived from recreational and commercial boating and/or local human activities. However, previous work has demonstrated that environmental mercury, derived from historic mining activities in the TOB watershed, is high and accumulates in tissues of resident fish including surfperch (Gassel et al., 2004). Mercury and methyl-mercury, in addition to some other metals (e.g., cadmium), have been reported to decrease plasma T4 and T3, possibly via direct thyroid cell effects, as well as by interfering with deiodinase activity (Eales et al., 1999; Brown et al., 2004a; Soldin et al., 2008). Future study of the relationship between metal contamination and thyroid endocrine function may be prompted by these findings, possibly comparing fish populations from TOB and nearby BDB, given their disparate thyroid endocrine phenotypics.

5. Conclusion

In conclusion, this study identified significant alterations in the thyroid endocrine system of two fish species in association with different environments in the San Francisco Bay Area. Several of the alterations were consistent between the species as well as between two years of study. Differing patterns of response in thyroid endocrine parameters suggested more than one underlying mechanism of effect. Exposure to PCBs appears to be at least one possible cause of the thyroidal alterations, given numerous correlations with hepatic PCB concentrations. Both AhR- and non-AhR-mediated responses to PCBs were implicated, since both co-planar and non-co-planar PCBs were significantly related. Chlorinated pesticides, and possibly certain PAHs (acting via AhR-mediated mechanisms?), were also implicated in thyroid endocrine alterations in the fish. We conclude that certain aquatic environments in the San Francisco Bay Area are associated with significant alterations in fish thyroid endocrine status, which can be significantly related to current-day exposures to contaminant chemicals such as PCBs.

Conflict of interest

The authors declare that no conflict of interest exists that would prejudice the impartiality of this scientific work.

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