Metabolic Effects of a Chronic Dietary Exposure to a Low-Dose Pesticide Cocktail in Mice: Sexual Dimorphism and Role of the Constitutive Androstane Receptor

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BACKGROUND: Epidemiological evidence suggests a link between pesticide exposure and the development of metabolic diseases. Most experimental studies have evaluated the metabolic effects of pesticides using individual molecules, often at nonrelevent doses or in combination with other risk factors such as high-fat diets.

OBJECTIVES: To our knowledge, we are the first to demonstrate a sexually dimorphic obesogenic and diabetogenic effect of chronic dietary exposure to a common mixture of pesticides at TDI levels, and to provide evidence for a partial role for CAR in an in vivo mouse model. This raises questions about the relevance of TDI for individual pesticides when present in a mixture.

METHODS: A mixture of six pesticides commonly used in France, i.e., boscalid, captan, chlorpyrifos, thiofanate, thiacloprid, and ziram, was incorporated in a standard chow at doses exposing mice to the tolerable daily intake (TDI) of each pesticide. Wild-type (WT) and constitutive androstane receptor−deficient (CAR−/−) male and female mice were exposed for 52 wk. We assessed metabolic parameters [body weight (BW), food and water consumption, glucose tolerance, urinary metabolome] throughout the experiment. At the end of the experiment, we evaluated liver metabolism (histology, transcriptomics, metabolomics, lipidomics) and pesticide detoxification using liquid chromatography−mass spectrometry (LC-MS).

RESULTS: Compared to those fed control chow, WT male mice fed pesticide chow had greater BW gain and more adiposity. Moreover, these WT males fed pesticide chow exhibited characteristics of hepatic steatosis and glucose intolerance, which were not observed in those fed control chow. WT exposed female mice exhibited fasting hyperglycemia, higher reduced glutathione (GSH):oxidized glutathione (GSSG) liver ratio and perturbations of gut microbiota−related urinary metabolites compared to WT mice fed control chow. When we performed these experiments on CAR−/− mice, pesticide-exposed CAR−/− males did not exhibit BW gain or changes in glucose metabolism compared to the CAR−/− males fed control chow. Moreover, CAR−/− females fed pesticide chow exhibited pesticide toxicity with higher BWs and mortality rate compared to the CAR−/− females fed control chow.

CONCLUSIONS: To our knowledge, we are the first to demonstrate a sexually dimorphic obesogenic and diabetogenic effect of chronic dietary exposure to a common mixture of pesticides at TDI levels, and to provide evidence for a partial role for CAR in an in vivo mouse model. This raises questions about the relevance of TDI for individual pesticides when present in a mixture. https://doi.org/10.1289/EHP2877

Introduction

The rates of metabolic disorders, including obesity and its complications, such as type 2 diabetes (T2D) and nonalcoholic fatty liver disease (NAFLD), have increased dramatically over the past three decades, especially in developed countries. Genetic factors and changes in lifestyle are key factors in the development of metabolic syndrome. Exposure to environmental contaminants is also suspected to be part of the etiological factors. Experimental and epidemiological studies suggest that some environmental pollutants, such as endocrine-disrupting chemicals (EDCs), may contribute to the onset and progression of obesity, diabetes, and NAFLD by interfering with various aspects of metabolism (Hong et al. 2012). Diethylstilbestrol, bisphenol A, phthalates, or tributyltin, which act as endocrine disruptors, may modulate the expression of genes involved in xenobiotic and steroid hormone metabolism.

Pesticides represent an increasingly widespread environmental contamination. Much epidemiological evidence has linked pesticides in water sources, groundwater, and/or foodstuffs with obesity, diabetes, insulin resistance (Casals-Casas and Desvergne 2011; Grün and Blumberg 2009), and NAFLD (AI-Eryani et al. 2015; Arciello et al. 2013; Wahlang et al. 2013). In a human cohort, the weight and body mass index of the offspring significantly correlated with the prenatal maternal levels of dichlorodiphenyldichloroethylene (DDE), a metabolite of the organochlorine (OC) pesticide dichlorodiphenyldichloroethane (DDT) (Karmaus et al. 2009). A recent systematic review and meta-analysis clearly supports the hypothesis that environmental or occupational exposure to OC pesticides increases the risk of T2D (Evangelou et al. 2016). Similarly, in vivo and in vitro experimental studies support the notion that nonpersistent pesticides (e.g., pyrethroids, in weight homeostasis, have recently been reviewed as possible risk factors for obesity (De Long and Holloway 2017; Newbold et al. 2009). Moreover, EDCs are known to affect liver function and, therefore, may contribute to increasing the prevalence of NAFLD worldwide (Polyzos et al. 2012). Hepatic metabolic homeostasis is tightly controlled by various transcription factors sensing hormonal, nutritional, and environmental cues. Among transcription factors, nuclear receptors such as constitutive androstane receptor (CAR) (Qatanani and Moore 2005) and peroxisome proliferator−activated receptors (PPARs) (Desvergne et al. 2006) mediate the expression of genes involved in xenobiotic and energy homeostasis (Evans and Mangelsdorf 2014). Nuclear receptor transcriptional activity is regulated upon binding of small lipophilic ligands that include hormone and metabolites as well as endogenous compounds and xenobiotics.
organophosphorus, neonicotinoids, dithiocarbamates, carbamates, triazines) are potential contributors to general or hepatic metabolic changes (Bhaskar and Mohanty 2014; Jin et al. 2014; Kim et al. 2013; Seidler and Slotkin 2011; Wang et al. 2014) and obesity (De Long and Holloway 2017; Newbold et al. 2009). However, epidemiological studies have mainly focused on OC pesticides, and evidence on other pesticide classes is limited. Most experimental studies assess the effect of individual compounds, using doses that are often far from the environmental levels to which the general population is exposed. Consumers are mainly exposed through the food chain to mixtures of pesticides at low doses (EFSA 2015) throughout their lifetime, and the consequences of such a chronic exposure on metabolic homeostasis have not been well described. Moreover, the metabolic impact of pesticides has often been observed in combination with other risk factors, such as high-fat diets (Adigun et al. 2010b; Howell et al. 2015; Lasstier et al. 2010; Maqbool et al. 2016) and/or after a relatively short exposure period, never exceeding a few months (Mostafalou and Abdollahi 2017). Finally, animals were often exposed to pesticides via oral gavage, an exposure route that is not representative of consumer exposure, which mainly occurs through food intake.

In the present study, we assessed the metabolic consequences of chronic dietary exposure to a cocktail of pesticides in mice. We selected six pesticides among those used for the treatment of apple orchards in the south of France. Therefore, the pesticide cocktail used may be commonly found in apples in the European Union, as recently described in an European Food Safety Authority (EFSA) report (EFSA 2015). Pesticides were incorporated at nontoxic doses in a standard chow and mice exposed to the tolerable daily intake (TDI) for each of these six individual pesticides. We followed the body weight (BW) and metabolic parameters of male and female mice using both the wild-type (WT) and CAR-deficient (CAR−/−) strains over a period of 52 wk, determining, to our knowledge, for the first time, the in vivo effect of chronic exposure to a mixture of pesticides at very low doses.

Methods

Chemicals

High-purity analytical standards of boscalid, captan, chlorpyrifos (99.2%), thiacloprid, thiophanate, and ziram were purchased from Sigma-Aldrich. Information on the toxicity of pesticides was obtained from Agritox (http://www.agritox.anses.fr/). Methanol was liquid chromatography–mass spectrometry (LC-MS) grade and purchased from Fisher Scientific (Thermo Fisher Scientific). Ultrapure water from the Milli-Q® system (Millipore) was used for mobile phases.

Pesticide Chow

Ziram, thiofanate, captan, chlorpyrifos, boscalid, and thiacloprid belong to various chemical families and are either fungicides or insecticides. Mice were exposed at the TDI of each pesticide as defined in mg/kg BW per day; the TDIs were those defined by the EFSA until 2014 for humans by the Joint Food and Agriculture Organization of the United Nations/World Health Organization Meeting on Pesticide Residues and extrapolated for mice on the basis of mean BW. To calculate the quantity of pesticide needed to incorporate in rodent pellets to achieve TDI, we assumed a BW of 30 g and a daily food intake of 5 g for all mice. The properties and quantities of each pesticide incorporated in the mouse diet are described in Table 1. Pesticides were dissolved individually in a 9:1 volume/volume (v/v) mixture of methanol:acetone and then mixed together. The solution was dispersed over the vitamin powder [vitaminic mixture 200, Scientific Animal Food Engineering (SAFE)] and then homogenized in a rotavapor (Laborota 4000™; BUCHI Switzerland) for 30 min at 45°C to evaporate the solvents and at room temperature for 50 min. The vitamin powder enriched with pesticides was sent to the Animal Feed Preparation Unit [UPAE; National Institute of Agricultural Research (INRA)] and incorporated into the pellets. Control feed was prepared as described above with the 9:1 methanol:acetone mixture alone. Pesticides present in the diet were quantified by Eurofins using gas chromatography–tandem mass spectrometry and liquid chromatography–tandem mass spectrometry, and the final concentrations were confirmed, except for ziram, which could not be detected at such low concentration (Table 1). Analysis from Eurofins laboratory confirmed that control feed contained none of the pesticides being studied.

Animal Experiment

The experimental work on animals was conducted in the Toxicall Unit, INRA, in a conventional laboratory animal room following the European Union guidelines for laboratory animal use and care, and was approved by an independent ethics committee (authorization number 05.185.02). The animals were treated humanely with due consideration to the alleviation of distress and discomfort and housed in polycarbonate cages (Charles River Type S, 424 cm²).

Eight-week-old female and male C57BL/6J mice were purchased from Janvier Laboratories and acclimatized for 4 wk. CAR-deficient mice (backcrossed on the C57BL/6J background) were engineered in Wei et al. laboratory (Figure S1A-B) and are bred for 10 y in our animal room. Figure S1C confirms that mice were deficient in CAR compared the WT C57BL/6J mice (Wei et al. 2000). The C57BL/6J mice served as WT strain for comparison with the CAR−/− mice on the same genetic background (C57Bl/6d). Both strains were treated in the same manner and experiments conducted concurrently to allow direct comparison of results. After the initial acclimatization period, all mice were maintained on standard chow for 4 wk (weeks −4 through 0), during which time we monitored BW weekly. At 16 wk of age (week 0), diets were switched to either a pesticide-enriched or -free diet, which was continued for a total of 52 wk to mimic a 30-y exposure in humans (Flurkey et al. 2007). Mice were randomly housed five or four per cage at their receipt (72 WT mice) or at their transfer from the breeding house.

Table 1. Chemical families, functions, and tolerable daily intake (TDI) (mg/kg body weight (BW)/day) of each pesticide and the expected and measured pesticides concentrations (determined level) (µg/kg food) in the animal pellets.

| Pesticide name | Chemical family | Function  | TDI (mg/kg BW/day) | Expected quantity (µg/kg food)a | Determined level (µg/kg food) |
|----------------|----------------|----------|--------------------|---------------------------------|-----------------------------|
| Ziram          | Dithiocarbamate| Fungicide| 0.006              | 36                              | NDa                         |
| Chlorpyrifos   | Organophosphorus| Insecticide| 0.01              | 60                              | 47                          |
| Thiacloprid    | Neonicotinoid  | Insecticide| 0.01              | 60                              | 56                          |
| Boscalid       | Carboxamide    | Fungicide| 0.04               | 240                             | 240                         |
| Thiofanate     | Benzimidazole  | Fungicide| 0.08               | 480                             | 205                         |
| Captan         | Dicarboximide  | Fungicide| 0.1                | 600                             | 165                         |

Note: BW, Body weight; ND, not determined; TDI, tolerable daily intake (http://www.agritox.anses.fr/).

aExpected quantity refers to the incorporated quantities of pesticide in mice pellets.

bZiram was not present at a detectible level (<0.01 mg/kg).

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into the animal room (72 CAR−/− mice) and allowed ad libitum access to food and water with a 12-h light/dark cycle (23 ± 2°C). Eight groups of 18 animals each were used in this study: WT males exposed, WT males nonexposed, WT females exposed, WT females nonexposed, CAR−/− males exposed, CAR−/− males nonexposed, CAR−/− females exposed, and CAR−/− females nonexposed. Mice were randomly distributed into unexposed and exposed groups. In this experiment, we used a standard synthetic diet from Animal Feed Preparation Unit (UPAE, INRA). This diet is composed of 63% carbohydrate, 5% fat, 22% protein, 2% cellulose, 1% vitamins (Safe 200, SAFE) and 7% minerals (Safe 205b, Safe France).

During the 6th, 26th, and 48th weeks of exposure, the mice were placed in individual metabolic cages (Tecniplast France) to obtain 24-h urine samples to use for metabolomics and mass spectrometry. The samples were stored at −80°C until analysis. After 52 wk of exposure, 4 to 5 mice from each experimental group were daily randomly sacrificed (in total, 16 to 20 animals from the four experimental groups were sacrificed per day) by cervical dislocation at Zeitgeber time 17 (i.e., in the fed state) at which hepatic activities are optimal (Lu et al. 2013). Blood was collected from the submandibular vein with a lancet into lithium heparin–coated tubes (BD Microtainer®, BD). Plasma was prepared by centrifugation (1,500 g; 10 min; 4°C) and stored at −80°C to be used for enzyme-linked immunosorbent assay (ELISA), biochemical analysis, and metabolomics. Liver samples were collected, weighed, snap-frozen in liquid nitrogen, and kept at −80°C for further analyses (liver neutral lipid analyses, lipidomics, microarray, and metabolomics). Subcutaneous and epidydimal white adipose tissue (WAT) were collected and weighed. Liver samples (50 mg) were fixed in formaldehyde (4%) for 24 h and embedded in paraffin for histology.

**Glucose Tolerance Test**

All experiments were performed on conscious mice. For the oral glucose tolerance test at weeks 36 and 48 or the intraperitoneal glucose tolerance test at week 16, mice were fasted for 6 h and received an oral (2 g/kg BW) or intraperitoneal (1 g/kg BW) glucose load. Blood glucose was measured at the tail vein using an Accu-Check® Performa glucometer (Roche Diagnostics) 15 min prior to and at 0, 15, 30, 45, 60, 90, and 120 min after the glucose load.

**Plasma Analysis**

Plasma insulin was assayed after 52 wk of pesticide exposure using the ultrasensitive mouse insulin ELISA kit (EMD Millipore) following the manufacturer’s instructions. Aspartate transaminase, alanine transaminase (ALT), total cholesterol, low-density lipoprotein (LDL) cholesterol, high-density lipoprotein (HDL) cholesterol, free fatty acids (FFAs), and triglycerides (TGs) were determined using a Pentra 400 biochemical analyzer (HORIBA, Montpellier, France; Anexplo facility, Toulouse, France).

**Liver Neutral Lipid Analysis**

Hepatic lipid contents were determined at the end of the experiment as described elsewhere (Bligh and Dyer 1959). Briefly, following homogenization of tissue samples in 2:1 (v/v) methanol/ethylene glycol-bis(β-aminoethyl ether)-N,N,N',N'-tetraacetic acid (EGTA; 5 mM), lipids corresponding to an equivalent of 2 mg of tissue were extracted according to Bligh and Dyer in chlorofrom: methanol:water (2:5:2:5:2:1, v/v/v) in the presence of the internal standards glyceryltrimonoacanole, stigmasterol, and cholesterol esters were analyzed by gas chromatography on a Focus Thermo Electron system using a Zebron-1 Phenomenex (Phenomenex Zebron-1, England) fused-silica capillary column [5 m; 0.32 mm internal diameter (i.d.); 0.50 μm film thickness]. The oven temperature was programmed from 200 to 350°C at a rate of 5°C/min, and the carrier gas was hydrogen (0.5 bar). The injector and detector were at 315°C and 345°C, respectively.

**Liver Fatty Acid Analysis**

To measure all hepatic fatty acid methyl ester (FAME) molecular species, lipids that corresponded to an equivalent of 1 mg of liver were extracted in the presence of the internal standard glyceryl triheptadecanole (2 μg). The lipid extract was transmethyalted with 1 mL boron trifluoride (BF3) in methanol (14% solution; Sigma) and 1 mL heptane for 60 min at 80°C and evaporated to dryness. The FAMEs were extracted with heptane/water (2:1). The organic phase was evaporated to dryness and dissolved in 50 μL ethyl acetate. A sample (1 μL) of total FAME was analyzed with gas–liquid chromatography [Clarus 600 system (PerkinElmer), with FAMEWAX fused silica capillary columns (Restek), 30 m × 0.32 mm i.d., 0.25 μm film thickness]. Oven temperature was programmed to increase from 110°C to 220°C at a rate of 2°C/min, and the carrier gas was hydrogen [7.25 pounds per square inch (psi)]. Injector and detector temperatures were 225°C and 245°C, respectively.

**Liver Phospholipid and Sphingolipid Analysis**

The liquid chromatography solvent acetonitrile was high-performance liquid chromatography (HPLC) grade and purchased from Acros Organics. Ammonium formate (>99.9%) was supplied by Sigma-Aldrich. Synthetic lipid standards [ceramide (Cer) d18:1/18:0, Cer d18:1/15:0, phosphatidylethanolamine (PE) 12:0/12:0, PE 16:0/16:0, phosphatidylcholine (PC) 13:0/13:0, PC 16:0/16:0, sphingomyelin (SM) d18:1/18:0, SM d18:1/12:0] were purchased from Avanti Polar Lipids.

Lipids were extracted from the liver (2 mg) as described by Bligh and Dyer (1959) in dichloromethane/methanol (2%/acetic acid)/water (2.5:2.5:2 v/v/v). Internal standards were added (Cer d18:1/15:0, 16:0; PE 12:0/12:0, 18:0; PC 13:0/13:0, 16:0; SM d18:1/12:0, 16:0; PI 16:0/17:0, 30:0; PS 12:0/12:0, 156:25). The solution was centrifuged at 1,500 rpm for 3 min. The organic phase was collected and dried under azote, then dissolved in 50 μL MeOH. Sample solutions were analyzed with an Agilent 1290 UPLC system coupled to a G6460 triple quadrupole spectrometer (Agilent, version 2.1). MassHunter software was used for data acquisition and analysis. A Kinetex® HILIC column (Phenomenex; 50 × 4.6 mm; 2.6 μm) was used for LC separations. The column temperature was maintained at 40°C. The mobile phase, A, was Acetonitrile; and B was 10 mM ammonium formate in water at pH 3.2. The gradient was as follows: from 10% to 30% B in 10 min, 50% B from 10 to 12 min, and then back to 10% B at 13 min for 1 min to re-equilibrate prior to the next injection. The flow rate of the mobile phase was 0.3 mL/min, and the injection volume was 5 μL. An electrospray source was employed in positive (for Cer, PE, PC, and SM analysis) or negative ion mode (for PI and PS analysis). The collision gas was nitrogen. Needle voltage was set at +4,000 V. Several scan modes were used. First, to obtain the naturally different masses of different species, we analyzed cell lipid extracts with a precursor ion scan at 184-mass-to-charge ratio (m/z), 241 m/z, and 264 m/z for PC/SM, PI, and Cer, respectively. We performed a neutral loss scan at 141 m/z and 87 m/z for PE and PS, respectively. The collision energy optimums for Cer, PE, PC, SM, PI, and PS were 25 evolt (eV), 20 eV, 30 eV, 25 eV, 45 eV, and 22 eV, respectively. Then the corresponding Single Reaction Monitoring (SRM) transitions were used to quantify different phospholipid species for each class. Two Multiple Reaction Monitoring (MRM) acquisitions were necessary, due to important differences between

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phospholipid classes. Data were treated with QqQ Quantitative (version B.05.00; Agilent) and Qualitative analysis software (version B.04.00; Agilent).

**Histology**

Paraformaldehyde-fixed, paraffin-embedded liver tissue was sliced into 3-μm sections, deparaffinized, and stained with hematoxylin and eosin for histopathological analysis. The staining was visualized with a Leica microscope DM4000 B equipped with a Leica DFC450 C camera (Leica Microsystems). The 72 slides (one slide per animal, 18 slides per group, and four groups: WT males and females fed control or pesticide chow) were full-blind analyzed by two individuals (C.L. and L.G.P.), and observations were approved by an histopathologist (T.A.S.). The score of steatosis was calculated according to the Kleiner method (Kleiner et al. 2005) and was the sum of three scores using the entire slice field: (a) the first score was defined according to the surface covered by lipid droplets (i.e., a surface of lipid droplet <5% of the entire slice correspond to a score 0; a surface of lipid droplet >66% of the entire slice correspond to a score 3); (b) for the second score, a predominant azonal distribution pattern of lipid droplets was graded with the highest value (score 3); and (c) the third score quantified the presence (score 1) or absence (score 0) of microvesicles.

**Gene Expression Studies**

RNAs were extracted from liver samples collected at the end of the exposure in WT and CAR−/− females and males fed control or pesticide chow using the method (Chomczynski and Sacchi 1987) with TRI Reagent® (Sigma-Aldrich). RNAs were quantified using nanodrop (NanoDrop™ 1000; Thermo Scientific). Two micrograms of total RNA were reverse transcribed using the High-Capacity cDNA Reverse Transcription Kit (Applied Biosystems™). The SYBR Green (Applied Biosystems, California) assay primers are presented in Table S1. Amplification was performed using an ABI Prism 7,300 Real-Time PCR System (Applied Biosystems™). Quantitative real-time polymerase chain reaction (qPCR) data were normalized to TATA-box-binding protein mRNA levels and analyzed with LinRegPCR (version 2015.3; Jan Ruijter) to get mean efficiency (NO), which is calculated as follows: NO = threshold/[(Eff mean)k] with Eff mean: mean PCR efficiency and CQ: quantification cycle.

**Microarray Gene Expression Studies**

Gene expression profiles were obtained for six liver samples per group at the GeT-TRiX facility (GéneToul, Génopole Toulouse Midi-Pyrénées, France) using Sureprint G3 Mouse GE v2 microarrays (8x60K; design 074,809; Agilent) following the manufacturer’s instructions. For each sample, Cyanine-3 (Cy3)–labeled complementary RNA (cRNA) was prepared from 200 ng of total RNA using the One-Color Quick Amp Labeling kit (Agilent) according to the manufacturer’s instructions, followed by Agencourt RNAclean XP (Beckman Coulter Inc.). Dye incorporation and cRNA yield were checked using a Dynabeads™ 96 ultraviolet/visible (VIS) droplet reader (Trinean). A total of 600 ng of Cy3-labeled cRNA was hybridized on the microarray slides following the manufacturer’s instructions. Immediately after washing, the slides were scanned on an Agilent G2505C Microarray Scanner using Agilent Scan Control A.8.5.1 software and the fluorescence signal extracted using Agilent Feature Extraction software (version 10.10.1.1) with default parameters. Microarray data and experimental details are available in National Center for Biotechnology Information Gene Expression Omnibus (Edgar et al. 2002) and are accessible through GEO Series accession number GSE101405; http://www.ncbi.nlm.nih.gov/geo/query/acc.cgi?acc=GSE101405).

Microarray data were analyzed using R (version 3.1.2; R Development Core Team) Bioconductor packages (version 3.0) as described in GEO accession GSE 101,405. Raw data (median signal intensity) were filtered, log2 transformed, and normalized using the quantile method (Bolstad et al. 2003). A model was fitted using the limma lmFit function (Smyth 2004). Pair-wise comparisons between biological conditions were applied using specific contrasts. A correction for multiple testing was applied using the Benjamini-Hochberg procedure (Klipper-Aurbach et al. 1995) for false discovery rate. Hierarchical clustering was applied to the samples and the differentially expressed probes using 1-Pearson correlation coefficient as distance and Ward’s criterion for agglomeration. The clustering results are illustrated as a heatmap of expression signals. Functional annotation clustering of GO Biological Process were performed using DAVID Bioinformatics Resources 6.7 (Huang et al. 2009a, 2009b).

**Proton Nuclear Magnetic Resonance (1H-NMR) Based Metabolomics**

Urine, plasma, and liver polar extracts were prepared for NMR analysis as described previously (Beckonert et al. 2007). All 1H-NMR spectra were obtained on a Bruker DRX-600-Avance NMR spectrometer (Bruker) using the AXIOM metabolomics platform (MetaToul) operating at 600.13 MHz for 1H resonance frequency using an inverse detection 5-mm 1H-13C-15N cryorobe attached to a cryoplatform (the preamplifier cooling unit). The 1H-NMR spectra were acquired at 300 K using a standard one-dimensional noesypr1D pulse sequence with water presaturation and a total spin-echo delay (2 nt) of 100 ms. A total of 128 transients were collected into 64,000 data points using a spectral width of 12 pulses/s, a relaxation delay of 2.5 s, and an acquisition time of 2.28 s. 1H-1H COSY, 1H-1H TCO SY, and 1H-13C Heteronuclear Single Quantum Coherence (HSQC) were obtained for each biological matrix on one representative sample for metabolite identification.

Data were analyzed by applying an exponential window function with a 0.3-Hz line broadening prior to Fourier transformation. The resultant spectra were phased, baseline corrected, and calibrated to trimethylsilylpropanoic acid (TSP) (δ 0.00 ppm) manually using Mnova NMR (version 9.0; Mestrelab Research S.L.). The spectra were subsequently imported into MatLab (R2014a; MathWorks, Inc.). All data were analyzed using full-resolution spectra. The region containing the water resonance (δ 4.6–5.2 ppm) was removed, and the spectra were normalized to the probabilistic quotient (Dieterle et al. 2006) and aligned using a previously published function (Veselkov et al. 2009).

Data were mean-centered and scaled using the unit variance scaling prior to analysis with orthogonal projection on latent structure-discriminant analysis (O-PLS-DA). 1H-NMR data were used as independent variables (X matrix) and regressed against a dummy matrix (Y matrix) indicating the class of samples (mice fed control or pesticide chow) (Trygg and Wold 2002). The O-PLS-derived model was evaluated for goodness of prediction (Q2Y). The loading and O-PLS-DA correlation coefficients were calculated for each variable and back-scaled into a spectral domain so that the shapes of the NMR spectra and the signs of the coefficients were preserved (Cloarec et al. 2005b, 2005a). The weights of the variables were color-coded according to the square of the O-PLS-DA correlation coefficients. Correlation coefficients extracted from significant models were filtered so that only
significant correlations above the threshold defined by Pearson’s critical correlation coefficient ($p < 0.05; |r| > 0.7; for n = 8 per group) were considered significant. For illustration purposes, the area under the curve of several signals of interest was integrated and significance tested with the $t$-test.

NMR data are freely available at MetaboLights (http://www.efi.ac.uk/metabolights/) under the accession number MTBLS602.

**Mass Spectrometry**

Urine from seven to eight mice per group was screened for pesticides and their metabolites using liquid chromatography coupled to high-resolution mass spectrometry (HPLC-HRMS). The chromatographic system consisted of a RSLC3000 UHPLC (Thermo Scientific) operating with an Hypersil Gold C18 column (100 × 2.1 mm; 1.9 μm) (Thermo Scientific). A gradient program of (A) H2O/CH3OH/CH3COOH (95:5:0.1, v/v/v) and (B) CH3OH/CH3COOH (1,000:1, v/v) was used at 40°C and a flow rate of 0.3 mL/min as follows: 0 min, 0% B; from 30 to 35 min, 100% B; and from 36 to 45 min, 0% B. A volume of 10 μL of each sample diluted twice with mobile phase A injected. The mass spectrometer was an LTQ Orbitrap XL™ (Thermo Scientific) operating in the negative mode. ESI parameters were set as follows: spray voltage, 4 kV; sheath gas flow rate (N2), 55 arbitrary units (a.u.); auxiliary gas flow rate (N2), 10 a.u.; capillary temperature, 300°C; capillary voltage, −40 V; and tube lens offset, −200 V. High-resolution mass spectra were acquired between m/z 80 and 800 at a resolution of 30,000. The mass spectrometer was calibrated using the Thermo Scientific calibration solution in agreement with their protocol. Tandem mass spectrometry (MS-MS) spectra were obtained using the collision-induced dissociation mode of the ion trap analyzer at low resolution and a normalized collision energy of 25%.

Data were extracted from mass spectrometry acquisition files using xcms software, version 3.5 (Scripps Research Institute, La Jolla, CA) (Smith et al. 2006) with centwave algorithm (Tautenhahn et al. 2008). To avoid the effect of urine dilution, variation among sample data was normalized using probabilistic quotient normalization (Dieterle et al. 2006). In the resulting normalized data, univariate analysis of variance was used to study the effect of sex on the different molecule intensities. All these data analysis steps were performed using workflow4metabolomics, a collaborative portal dedicated to metabolomics data processing (Giacomoni et al. 2015).

HPLC-HRMS data were analyzed using a methodology previously developed for exposomics studies (Jamin et al. 2014). From the six studied pesticides, a list of suspected metabolites was compiled from previously described metabolites of these pesticides, as well as all other putative phase II metabolites (Table S2). Thus, the signals of 16 compounds were monitored for the metabolism of boscalid (15 metabolites + the parent molecule), 13 for thiacloprid (12 metabolites + the parent molecule), 30 for captant (29 metabolites + the parent molecule), 9 for thiophanate (8 metabolites + the parent molecule), 4 for chlorpyrifos (3 metabolites + the parent molecule), and 9 for ziram (9 metabolites). However, standards of these metabolites are not commercially available. In this study, a direct comparison with nonexposed animals was used as a validation strategy. From data extracted with xcms, 16 features matched the theoretical m/z of monitored compounds with a mass precision <5 ppm. MS-MS experiments were performed to confirm the identity of suspected compounds. Conjugation to glucuronic acid was confirmed by detection of the characteristic loss of 176 unified atomic mass (u), the conjugation to sulphate confirmed by the loss of 80 u, and the conjugation to mercapturic acid confirmed by the loss of 129 u. Concerning captant metabolites, MS-MS spectra of the ion detected at m/z 150.0564, corresponding to the in-source fragmentation of sulfate and glucuronic conjugates of tetramethylsilane (THPI), showed a loss of 43 u corresponding to neutral isocyanic acid CONH, validating the identification of these metabolites. Finally, seven features were confirmed by MS-MS experiments, corresponding to five metabolites: two metabolites of captan ([M-H]- and [M-SO3-H]- of THPI conjugated to sulphate and [M-GlAc-H]- of THPI conjugated to glucuronic acid), one metabolite of chlorpyrifos ([M-H]- of 3,5,6-trichloro-2-pyridinol), and two metabolites of boscalid ([M-H]- and [M-GlAc-H]- of 2-chloro-N-(4’-chloro-5-hydroxybiphenyl-2-yl) nicotinamide conjugated to glucuronic acid and the [M-H]- of the mercapturic conjugation of boscalid). Structures are available in Figure S2A-B. HRMS and MS-MS data are freely available at MetaboLights (http://www.efi.ac.uk/metabolights/) under the accession number MTBLS596.

**Statistical Analysis**

Statistical analyses were performed using GraphPad Prism for Windows (version 4.00; GraphPad Software). One-way or two-way analysis of variance (ANOVA) was performed, followed by appropriate posthoc tests (Dunn’s or Bonferroni) when differences were found statistically significant. When only two groups were compared, the student’s $t$-test was used; $p < 0.05$ was considered significant.

**Results**

**Body Weight and White Adipose Tissue Gain in Wild-Type Mice Exposed to Pesticide Chow Compared to Those Fed Control Chow**

Pesticide levels were assessed in pellets before the beginning of the experiment (Table 1), and we confirmed that pesticides were present in the diet at levels significantly below the expected quantities. We calculated the actual pesticide levels to which mice were exposed after accounting for actual animal BW and food intake during the exposure (Figure S3A). We confirmed that males and females were exposed to similar levels of pesticides and that these levels were below the TDI for each pesticide (Figure S3B).

Beginning at 20 wk postexposure and through the end of the exposure period, male WT mice fed the pesticide mixture gained significantly more weight than those fed the control chow (Figure 1A). After 52 wk of exposure, the BW gain in exposed male mice was twice that of the male mice fed control chow (7.3 ± 1.2 g vs. 3.65 ± 0.7 g, respectively). Epididymal and subcutaneous WAT weights were significantly higher in exposed WT male mice than the WT males fed control chow group (subcutaneous WAT: 0.019 ± 0.003 g vs. 0.012 ± 0.001 g; epididymal WAT: 0.033 ± 0.004 g vs. 0.026 ± 0.003 g; Figure 1A). As shown in Figure S4, WT male and female fed control chow mice had distinct food and water consumption patterns, and specifically female mice fed control chow consumed more food and less water over the period evaluated. Food intake was not different in WT male or female mice fed pesticide chow compared to those fed control chow (Figure S4). WT female mice fed pesticide chow consumed more water than those fed the control chow. In contrast, WT female mice did not exhibit any significant changes in BW or fat mass upon exposure to pesticides (Figure 1B). No cage effect in BW was observed during the experience in WT male fed control or pesticide chow (Figure S5A) and in WT female fed control or pesticide chow (Figure S5B).

**Glucose Tolerance and Fasting Glucose Levels in Mice Fed Control or Pesticide Chow**

We evaluated the fasting blood glucose and glucose tolerance of WT male and female mice fed control or pesticide chow (Figure 2).
At 16, 36, and 48 wk of exposure, WT male mice fed pesticide chow exhibited significantly higher blood glucose following a glucose load compared to those fed control chow (Figure 2A). In addition, at 36 and 48 wk of exposure, WT male mice fed pesticide chow had significantly higher fasting blood glucose (145 ± 6 mg/dL and 163 ± 8 mg/dL, respectively) compared to those fed control chow (124 ± 4 mg/dL and 118 ± 9 mg/dL, respectively) (Figure 2C). WT female mice fed pesticide chow also exhibited significantly higher fasting blood glucose at 36 and 48 wk (Figure 2D) and significantly higher blood glucose after a glucose load at 48 wk (Figure 2B) (182 ± 8 mg/dL and 172 ± 9 mg/dL at 36 and 48 wk, respectively) compared to those fed control chow (142 ± 6 mg/dL and 144 ± 7 mg/dL at 36 and 48 wk, respectively). After 16, 36, and 48 wk of exposure, WT female mice exhibited significantly higher fasting blood glucose than WT male mice (Table S3). This difference was observed in both mice that were fed pesticide chow and those fed control chow. No significant differences in insulin levels were observed in WT males (Figure S6A) or females (Figure S6B) fed pesticide chow compared to those fed control chow. Further biochemical analyses of serum revealed significantly higher ALT activity in WT males fed pesticide chow than in those fed control chow (Figure 3A).

**Effect of Eating Pesticide Chow on Liver Metabolism**

We further investigated pesticide-induced metabolic perturbations in the liver. Hematoxylin and eosin staining revealed that the livers from pesticide-exposed male mice presented hepatocellular vacuolizations localized predominantly in the centrilobular zone, characteristic of an emerging steatosis (Figure 3A). The steatosis score, calculated using the Kleiner method as described in the “Material and Methods” section (Kleiner et al. 2005), was 0.47 ± 0.32 in WT males fed control chow and 2.56 ± 0.71 in exposed WT male mice. Importantly, exposed male mice had significantly higher levels of hepatic TGs than unexposed male mice (445.7 ± 90.8 vs. 195.9 ± 16.6 mg/mg liver) (Figure 3A). This difference in hepatic TGs between exposed and unexposed males (Figure 3A) correlated with weak but significant differences in serum FFAs and TG levels (Figure S7A) between the two groups without significant impact on the plasma metabolome (Figure S8A). Plasma analysis revealed no significant differences in total or HDL and LDL cholesterol (Figure S7B) between male mice fed pesticide and those fed control chow.

By contrast, females fed the pesticide chow did not exhibit observable differences in liver histology, with steatosis scores of 2.53 ± 0.39 compared to 2.23 ± 0.49 for mice fed control chow (Figure 3C). Moreover, female mice fed pesticide chow did not exhibit higher hepatic TGs or ALT than those fed control chow. Plasma analysis revealed no significant differences in FFA or TG (Figure S7C) or total or HDL and LDL cholesterol (Figure S7D) between female mice fed pesticide and those fed control chow, and 1H-NMR showed no significant differences in plasma metabolomics (Figure S8B). Exposed females exhibited a slight decrease of LDL cholesterol compared to animals fed control chow (Figure S7D).

Extensive lipid profiling revealed distinct hepatic lipid profiles for WT male and female mice that were fed control and pesticide chow for 52 wk (Figure S9A). Cluster 5 showed lipid species specifically down-regulated in WT males exposed to pesticides. Exposure to pesticide mixture led in WT males to slight but significant changes in the relative PS32:0, PS34:0, and PS36:1 abundance (Figure S9B). Cluster 4 showed lipid species...
up-regulated in WT males and females exposed to pesticides. It is noteworthy that pesticide exposure led in both WT males and females to increased abundance of SM18:1/16:0, PC30:0, and PC32:0 (Figure S9C). Lipid species down-regulated in WT males and females exposed to pesticides (cluster 2) were C18:2n-6, Cer d18:1/26:1, and SM18:1/24:1 (Figure S9D). The relative TG51, TG53, PI36:0 abundance (Cluster 1) was up-regulated in WT males exposed to pesticides, while exposed WT females exhibited higher PI36:0 (Figure S9E).

1H-NMR Metabolomics

We used 1H-NMR-based metabolomics of aqueous liver extracts to evaluate the metabolic profiles of WT male (Figure 3B) and female (Figure 3D) mice fed control or pesticide chow for 52 wk. Whereas there was no significant difference detected in male mice (Figure 3B), we observed a clear discrimination between the metabolic profiles of female mice fed control and pesticide chow (see also partially assigned 600 MHz 1D NMR spectra of aqueous liver extract in Figure S10A and 1H and 13C assignments for identified metabolites in Table S4). Analysis of the discriminant metabolites revealed higher levels of fumarate and oxidized glutathione (GSSG) and lower levels of nicotinamide adenine dinucleotide phosphate and reduced glutathione (GSH) in the livers of pesticide-exposed females (Figures 3E and 3F). Thus, the GSH/GSSG ratio was significantly lower in the livers of exposed females (Figure 3F).

1H-NMR metabolomics of urinary samples was performed after 6 wk, 24 wk, and 48 wk of exposure (Figure 4, Figure S10B, and Table S5). We did not observe any significant difference in the endogenous urinary metabolic profiles of exposed and nonexposed males (Figure 4A). In contrast, and confirming our previous metabolomics observations in liver extracts, pesticide exposure induced significant urinary metabolic changes in females. After only 6 wk of exposure, an O-PLS-DA statistical model was fitted that could significantly discriminate exposed females from their nonexposed counterparts (parameters of the O-PLS-DA model: $Q^2=0.23; p=0.04$; Figure 4B). The metabolic profiles were differentiated mainly by the level of...
methylamine and trimethylamine oxide, which are both derived from
than WT females fed control chow (Table S5). We also observed in
females fed control chow (Craciun and Balskus 2012). Other metabo-
dietary choline fermentation by commensal bacteria than in WT
a
48 wk of pesticide exposure.
WT females fed pesticide and WT females fed control chow after only
in 2-KAA levels between exposed and nonexposed female mice at 48
wk (Figure S12). At the end of the experiment, the metabolic pro-
duced dimorphic response to pesticides, we examined the hepatic tran-
Nuclear Receptor Activation in Female and Male Mice
To further investigate the mechanisms involved in the sexually
dimorphic response to pesticides, we examined the hepatic tran-
scriptome through microarray analysis of six samples per group.
Differentially expressed genes were classified using hierarchical
clustering, which illustrated distinct gene regulation in response to
pesticides between the two sexes (Figure 5A). One hundred and
two probes were selected as differentially regulated in response to
pesticide exposure in males and females (p < 0.001 and
|fold change| > 1.5). Six clusters of genes were identified (Figure
5A); clusters 1 and 4 clearly highlight genes down-regulated upon
pesticide exposure in WT males and females, respectively. Clusters
5 in WT females and clusters 3 and 6 in WT males show
genes whose expression was up-regulated in response to pestic-
ides. The Venn diagram in Figure 5B represents the numbers of
hepatic genes specifically upregulated in response to pesticides
in male and female mice selected with a
$p$-value < 0.05.
A total of 536 genes (43%) were
significantly up-regulated in response to pesticides in
male and female mice selected with a $p$-value < 0.05 in order to get
a larger regulated gene sampling. A total of 536 genes (43%) were
significantly up-regulated in females and 670 (54%) in males upon
pesticide exposure, among which only 43 were common between
the two sexes (Figure 5B). For the down-regulated genes, 511
(36.4%) were significantly altered in females, 853 (60.8%) signifi-
cantly altered in males, and 38 were common (Figure S13).
Pathway enrichment analysis identified the top biological
functions associated with the genes identified as up-regulated in
response to pesticide exposure for WT male and female mice
(Figure 5C). In WT pesticide-exposed male mice, we observed
2-keto adipic acid (2-KAA), the identification of which was confirmed
by additional two dimensional sequences and spike in NMR experi-
ments (Figures S11A and B). This metabolite was undetectable in
urine samples from male mice (both those fed control and those fed
pesticide chow), whereas females mice (both those fed control and
those fed pesticide chow) exhibited high levels of 2-KAA at the be-
ginning of the experiment (e.g., week 6; Figure S12). This metabolite
decreased gradually upon pesticide exposure, with a 10-fold di-
ference in 2-KAA levels between exposed and nonexposed female mice at 48
wk (Figure S12). At the end of the experiment, the metabolic profiles
of WT females fed control and pesticide chow were distinct (parame-
ters of the PLS-DA model: Q2Y = 0.65). The direction of the metabolite indicates the group with which it is posi-
tively associated as labeled on the diagram. (F) Area under the curve of the $^1$H-NMR spectra was integrated for the glutathione signals (GSSG, oxidized form, multiplet at 4.75 ppm; GSH, reduced form, multiplet at 4.56 ppm). Data are presented as the mean ± standard error of the mean. *$p$ < 0.05. **$p$ < 0.001. $p$-Values represent the difference between mice fed pesticide chow and those fed control chow as determined using a student’s $t$-test. $n$ = 18 mice per group.
increased expression of genes linked to DNA replication (19 genes), extracellular matrix (25 genes), cytoskeleton (10 genes), endoplasmic reticulum membrane (35 genes), oxidoreductase activity (30 genes), and nucleotide binding (74 genes) compared to WT male mice fed control chow (Figure 5C). Among the genes involved in the oxidoreductase activity pathway, Cyp2b9 and Cyp2b10 exhibited the largest differences in mRNA expression between WT male mice fed control chow and those fed pesticide chow (3.4-fold and 2.5-fold increases, respectively) (Figure 5D).

In WT female mice, the genes up-regulated by pesticide exposure were associated primarily with the peroxisome (23 genes), mitochondrion (83 genes), fatty acid oxidation and acyl-coA metabolic processes (20 genes), isomerase activity (8 genes), and translation processes (14 genes) (Figure 5C). The upregulation of Ehhadh and Acot5, two genes involved in fatty acid oxidation, was confirmed by Quantitative Reverse Transcription - Polymerase Chain Reaction (RT-qPCR) of the 18 liver samples (Figure 5E). Additional comparison of our regulated genes upon exposure to pesticide with previously published lists of PPARα-target genes show that 7.6% of genes regulated in response to pesticides in females are well-described target genes of PPARα (Figure S14A). Analysis of the expression profile for the 41 hepatic genes identified in our study and reported as related to PPARα-dependent pathways revealed that pesticide exposure changed several gene expressions, mainly in WT females (Figure S14B).

**Evaluation of the Pesticide-Detoxifying Capacities of Male and Female Mice**

Next, we focused the analysis of the hepatic microarray on genes involved in xenobiotic metabolism in males compared to females. Analysis of 82 genes present in the microarray and involved in this pathway revealed that pesticide exposure significantly affected 14 xenobiotic metabolism enzymes in WT males compared to only three in WT females (Figure 6A). These data show higher xenobiotic metabolic activity in the livers of exposed WT males compared to WT females.

Thus, we used an untargeted LC-MS–based exposomics approach (Jamin et al. 2014) to compare the urinary end products of pesticide metabolism after 52 wk of exposure (Figures 6B–D). Among the six assayed pesticides, only metabolites of boscalid, captan, and chlorpyrifos were clearly identified (Figure S2). Pesticides were confirmed to be absent in controls. Only one suspected metabolite of thiachlopride and two metabolites of thiophanate were detected according to their exact mass (oxazole derivative of thiachlopride, hydroxylated and oxidized derivatives of thiophanate), but could not be confirmed by MS-MS due to their weak...
signals. Three suspected metabolites of ziram were detected (dithiocarbamic acid conjugated or not to sulfate or glucuronic acid), but these nonspecific structures were also detected in urine of nonexposed mice. Therefore, metabolites of thiachlorpride, thiophanate, and ziram were not further considered for the exposomic data analysis. The metabolism of chlorpyrifos, boscalid, and captan differed between WT male and female mice fed pesticide chow for 52 wk (Figures 6B–D). Exposed WT females exhibited higher levels than exposed WT males of 3,5,6-trichloro-2-pyridinol (TCPy), a metabolite resulting from either the dearylation of chlorpyrifos oxon or its oxidation by Cyp3a11 (Croom et al. 2010). WT females also presented higher levels of sulfated captan and boscalid–glucuronide metabolites than exposed WT males. Therefore, despite less transcriptional regulation of hepatic xenobiotic metabolizing enzymes in the liver, WT females appeared to efficiently metabolize these three pesticides, suggesting that the metabolism of pesticides did not only occur in the liver. Interestingly, the mercapturic acid metabolite of boscalid was only found in WT males and was almost undetectable in WT females.
Role of Constitutive Androstane Receptor in Pesticide-Induced Metabolic Perturbations

We investigated the involvement of a key player in the transcriptional regulation of xenobiotic metabolizing enzymes, the nuclear receptor CAR. We utilized a constitutive CAR knockout mouse model to evaluate the role of CAR in pesticide-induced metabolic perturbations. We compared male and female CAR−/− mice fed pesticide chow for 52 wk with male and female CAR+/- mice fed control chow for 52 wk (18 mice per group). Male CAR−/− mice fed pesticide chow did not have significantly different BWs than those fed control chow (Figure 7A). Moreover, CAR−/− male mice fed control chow and those fed pesticide chow exhibited similar blood glucose levels following a glucose bolus at 16, 36, and 48 wk (Figure S15A). In the liver of CAR−/− males, pesticide exposure decreased the expression of Cyp2b9 and did not alter the expression of Cyp2b10 (Figure 8C). It is noteworthy that pesticide exposure significantly decreased the prototypical PPARα-target gene (Cyp4a10) in the CAR−/− male mice, which was not observed in WT male mice (Figure 5D). Finally, in male mice, knock out of CAR did not result in significant differences in the urinary levels of pesticide metabolites (Figure 8A). 1H-NMR analysis of plasma samples showed no significant differences in serum metabolomics between CAR−/− male mice fed control chow and those fed pesticide chow (Figure S8C). In contrast, female CAR−/− mice fed pesticide chow had significantly larger BWs than CAR−/− mice fed control chow beginning at 4 wk and lasting through 48 wk (Figure 7A). At week 12, the weight gain of exposed CAR−/− females was 7.49 ± 1.14 g compared to 3.39 ± 0.65 g in unexposed females. This significant difference in BW was maintained over 40 wk (Figure 7A). However, the BW of CAR−/− mice fed control chow gradually increased over the duration of the experiment and was not significantly different from that of CAR−/− mice fed pesticide chow from week 44 through week 52. As observed in male CAR−/− mice fed pesticide chow, female CAR−/− mice did not exhibit significantly higher serum glucose levels (either fasting or after a glucose bolus), as was observed for WT animals fed pesticide chow (Figure S15B). Moreover, pesticide exposure led to a high death rate in CAR−/− females (Figure 7B). After 52 wk of pesticide exposure, only 50% of CAR−/− female mice survived, whereas the survival rate was nearly 88.8% in unexposed CAR−/− females. In female WT mice, the survival rate was similar between mice fed control chow and those fed pesticide chow (Figure S16). Knockout of CAR in female mice modified the levels of almost all measured urinary pesticide metabolites (Figure 8B); CAR−/− females had lower levels of chlorpyrifos, captant (capton SO3) and boscalid metabolites compared to their WT counterparts, and higher levels of boscalid mercapturic acid. 1H-NMR analysis of plasma samples showed no significant differences in plasma metabolomics between CAR−/− female mice fed control chow and those fed pesticide chow (Figure S8D).

In contrast to WT female mice, for which we identified several PPARα-associated genes as up-regulated in mice fed pesticide chow compared to those fed control chow (Figure 5E), we detected significantly lower mRNA expression levels of PPARα target genes in female CAR−/− mice fed pesticide chow (Cyp4a10, Acox1, and Cpt1; Figure 8D). Finally, 1H-NMR-based metabolic profiling of
liver extracts of male CAR−/− mice did not show any significant difference between animals fed control chow and those fed pesticide chow (Figure 8E). Similarly, we observed no significant difference in the metabolic profiles of liver extracts for female CAR−/− mice fed control chow compared to those fed pesticide chow (Figure 8F), in contrast to the results in WT mice (Figure 3D).

Discussion

Metabolic diseases have multifactorial origins, often resulting from a combination of sedentary lifestyles and high-energy diets. The involvement of endocrine-disruptive environmental contaminants in the onset of diabetes and obesity is of growing interest. Various epidemiological (Evangelou et al. 2016), animal (Grün and Blumberg 2009), and mechanistic data (Grün and Blumberg 2009; Kim et al. 2013) support the hypothesis that exposure to several classes of pesticides, including organophosphorus and OCs, may be considered a risk factor for obesity and diabetes. However, in previous animal studies, metabolic disruptions were observed with specific individual compounds, such as DDT, DDE, dieldrin, glyphosate, mancozeb, imidacloprid, or chlorpyrifos (Androutsopoulos et al. 2013; Bhaskar and Mohanty 2014; De Long and Holloway 2017), and often at doses that are too high to be relevant for human exposure (Wang et al. 2014). Moreover, most studies investigate the obesogenic effect of pesticides in a context of high-fat diets (Adigun et al. 2010b; Howell et al. 2015; Lasram et al. 2014; Lassiter et al. 2010; Maqbool et al. 2016) or using a genetically induced model of obesity (Ob/Ob mice) (Mulligan et al. 2017; Peris-Sampedro et al. 2015). In our study, we aimed to use a pesticide exposure protocol relevant to consumers’ exposure. A mixture of six pesticides was incorporated into a standard rodent diet to mimic the exposure of consumers (EFSA), which occurs mainly as a mixture of several chemicals through food intake. We chose to assess a reference dose, the TDI, an estimated amount of a substance in food or drinking water that can be consumed over a lifetime without presenting an appreciable risk to health (EFSA).

Obesogenic Impact of the Pesticide Mixture in Wild-Type Males

We observed that chronic dietary exposure of WT male mice to a mixture of pesticides at levels that are below the TDI resulted in a greater BW gain and adiposity than in WT males fed control chow. This suggests that adipose tissue might be a possible target of pesticides in male mice, as observed previously with several compounds (Chapados et al. 2012). The proadipogenic properties of pesticides has been correlated with their capacity to induce oxidative stress in 3T3-L1 adipocytes in vitro (Shen et al. 2017) and in mice (Armstrong et al. 2013). Importantly, adipose PPARγ, a major player in WAT homeostasis, has been reported to be activated by several environmental pollutants, including mono-(2-ethylhexyl)phthalate in COS7, C2C12, and 3T3-L1 cell lines (Feige et al. 2007) and triflumizole in mouse 3T3-L1 preadipocytes and human multipotent mesenchymal stromal stem cells (Li et al. 2012).

Hepatic lipid accumulation has also been reported in mice upon exposure to perfluoroalkyl acids (Das et al. 2017), perfluorooctanoic acid (Hui et al. 2017), and cadmium (Zhang et al. 2015), and this raises the question whether the hepatic alterations observed in our study are specific to pesticide exposure. In our study, pesticide exposure led not only to steatosis glucose intolerance and elevated blood glucose, but also resulted in greater WAT mass and BW and more circulating FFAs in WT male mice compared to unexposed WT male mice. Previous studies showed that steatosis induced in mice by perfluoroalkyl acids (Das et al. 2017) and perfluorooctanoic acid (Yan et al. 2015) correlates with an increased expression of lipogenic genes in the liver. In the present work, we did not detect significant differences in the mRNA expression of hepatic genes involved in de novo fatty acid synthesis between mice fed control and pesticide chow. It is therefore possible that TGs...

Figure 7. Body weight (A) and survival (B) of male and female constitutive androstane receptor (CAR−/−) mice fed control (C) or pesticide (P) chow for 52 wk. (A) The line graphs show body weight (BW) for pesticide-exposed mice (black circles) and mice fed control chow (white circles) from 4 wk prior to exposure through 52 wk. The bar graphs show the gains in BW for pesticide-exposed mice after 52 wk. (B) Kaplan-Meier survival curves.
Figure 8. (A and B) Levels of high-performance liquid chromatography (HPLC)–high-resolution mass spectrometry (HRMS) features of pesticide metabolites identified in urine samples from male (A) and female (B) wild-type (WT) and constitutive androstane receptor (CAR)−/− mice fed pesticide chow for 52 wk. **p<0.01; ***p<0.001 compared to WT as determined using a student’s t-test. (C and D) Relative hepatic expression of CAR and peroxisome proliferator–activated receptor (PPARα) target genes as determined by RT-qPCR in male (C) and female (D) CAR−/− mice fed control (C) or pesticide chow (P) for 52 wk. (E and F) Orthogonal projection on latent structure-discriminant analysis (O-PLS-DA) score plots derived from the liver extract 1H-NMR–based spectra of male (E) and female (F) CAR−/− mice fed control or pesticide chow for 52 wk. Q2Y represents the goodness of fit for the PLS-DA models, and p-values were derived using 1,000 permutations of the Y matrix. Area under the curve of the 1H-NMR spectra was integrated for the glutathione signals (GSSG, oxidized form, multiplet at 4.75 ppm; GSH, reduced form, multiplet at 4.56 ppm). Data are presented as mean ± standard error of the mean. *p<0.05; **p<0.01; ***p<0.001. p-values represent differences between mice fed control chow and those fed pesticide chow as determined using a student’s t-test. Note: C, control; P, pesticide mixtures. n=18 mice per group.
accumulating in the liver of WT male mice exposed to pesticides originate from adipose tissue lipolysis rather than from hepatic de novo synthesis. Therefore, the molecular mechanism leading to steatosis in our model might be distinct from those described in other studies. However, whether hepatic steatosis is a generic adaptive response of this organ to chronic exposure to pollutants remains an open question. These results demonstrate that chronic low-dose pesticide exposure leads to sex-specific hepatic metabolic disturbances; WT male mice present with significantly increased hepatic TGs, whereas WT females present with metabolic changes related to oxidative stress.

Male WT mice fed pesticide chow exhibited higher mRNA expression of genes involved in hepatic detoxifying pathways compared to those fed control chow, suggesting that pesticide metabolism occurs mainly in the liver. This metabolism was independent of CAR, as male CAR−/− mice exhibited the similar urinary pesticide metabolite profiles as WT males. Liver CAR activation may be linked to the observed increased expression of hepatic genes involved in DNA replication, extracellular matrix, and cytoskeleton constituents, intracellular events that could predispose an individual to hepatocellular adenomas or carcinomas (Huang et al. 2005). The organophosphates, pyrethroids, triazole, and carbamate pesticides have been shown to activate CAR in vitro (Baldwin and Roling 2009; Tamura et al. 2013) or in vivo (Tamura et al. 2013). Thus, it is likely that, at least in our pesticide cocktail, chlorpyrifos (OP) and/or thiophanate (carbamate) may be involved in the activation of CAR in male mice, though one cannot rule out the interaction of the other compounds with CAR.

**Diabetogenic Effect of Pesticide Exposure in Females**

In female WT mice, there were no significant differences in BW or WAT mass between WT mice fed control chow and those fed pesticide chow. Moreover, female WT mice fed pesticide chow did not experience hepatic steatosis as defined by histology and hepatic TG quantification. However, transcriptomics and metabolomics profiling revealed a significant impact of pesticides in female livers, including a marked oxidative status and activation of PPARα signaling pathways, a transcription factor crucial for liver fatty acid degradation (Montagner et al. 2016; Régnier et al. [in press]). Therefore, pesticide-induced activation of PPARα is in good agreement with a reduced hepatic lipid accumulation in the livers of female mice. PPARα is also involved in controlling the expression of antioxidant defense enzymes (Anderson et al. 2004), such as those leading to the conjugation of GSH, which could explain at least part of the drop in reduced glutathione liver content in exposed female mice. β-oxidation also leads to the production of free radicals, which could also induce significant oxidative stress (Pessayre 2007). Therefore, we can hypothesize that some of the pesticides or their metabolites may interact with PPARα in WT females, leading to increased hepatic lipid catabolism. Previous studies demonstrated that some pesticides can activate PPARα. Pyrethrin and diclofop-methyl, for example, has been shown to induce PPARα-mediated transcriptional activities in in vitro reported gene assays using CV-1 monkey kidney cells and in mice (Takeuchi et al. 2006). The pesticides methiocarb and carbaryl exhibited PPARα agonist activities in in vitro reporter gene assays using COS1 cells (Abass and Pelkonen 2013; Fujino et al. 2016). The pesticide adjuvant Toximul® (Stepan Company) was reported to modulate hepatic metabolism through the activation of PPARα (Upham et al. 2007). However, no data are yet available regarding a possible interaction of one of the six pesticides of our study with this nuclear receptor.

Blood metabolomic assay on samples from WT exposed and nonexposed male and female mice displayed similar metabolite profiles. However, urinary metabolic profiling using 3H-NMR identified 2-KAA as an early biomarker that discriminates pesticide-treated WT females from WT females fed control chow and that may be linked to the observed increase in blood glucose in WT female exposed to pesticides. There was a trend of lower 2-KAA levels in pesticide-exposed WT females after only 6 wk of exposure and a significant difference after 36 and 48 wk of pesticide exposure. A product of lysine and tryptophan degradation, 2-KAA is further degraded in the mitochondria via the DHTKD1 enzyme, and 2-aminoacideric and 2-KAA aciduria have been linked to mutations in DHTKD1 (Hagen et al. 2015). Wang et al. (2013) identified 2-aminoacidic acid, the immediate precursor of 2-KAA, as a predictor of diabetes development in normoglycemic individuals in observational study in human and demonstrated in vivo in mice that it modulates glucose homeostasis, whereas in vitro studies in murine and human islets support an effect of 2-aminoacidic acid on insulin secretion (Wang et al. 2013). Using 40 strains of a well-characterized murine genetic reference population and a multilayered omics approach, Wu et al. (2014) identified Dhtkd1 as a primary regulator of piasmatic 2-aminoacidic acid, explaining variance in fasted glucose and diabetes status. Changes in 2-aminoacidate levels were highly correlated with changes in the Homeostatic Model Assessment (HOMA) index and therefore with insulin sensitivity. Finally, in a cohort of 835 Caucasian adults in the CoLaus study, including 43 diabetics and 792 nondiabetics males and females, 2-aminoacidic acid levels were also lower in the urine of diabetic patients compared to healthy patients (Wu et al. 2014). In this context, our findings suggest that 2-KAA may be an early biomarker of altered glucose homeostasis in WT female mice, and point out the pancreas as a possible early target of pesticides (Kamath and Rajini 2007).

We also observed significant differences in concentration of gut microbiota–related metabolites in the urine of WT female mice fed pesticide chow compared to those fed control chow. We found lower levels of metabolites from the methyamine pathway; trimethylamine is derived from dietary choline fermentation by commensal bacteria and metabolized into trimethylamine oxide in the liver (Craciun and Balskus 2012). We also observed higher levels of 3-indoxylsulfate, a metabolite derived from the conversion of dietary tryptophan to indole by enteric bacteria and further conversion of indole in the liver (Wikoff et al. 2009). Finally, the phenyl derivatives p-cresol glucuronide and phenylacetylglcine, which were also lower in the urine of pesticide-treated WT females, are well-described host gut microbiota metabolites (Wikoff et al. 2009). The gut microbiota possesses extensive xenobiotic metabolizing capacities (Claus et al. 2016) and may be involved in pesticide metabolism in females, thereby modifying their toxicity. A recent study demonstrated that the gut microbiota is involved in organophosphate metabolism, thereby inducing hepatic gluconeogenesis and glucose intolerance (Velmurugan et al. 2017). Our results suggest that pesticide treatment impacted the gut microbiota of WT female mice. However, these perturbations were observed only after 48 wk of pesticide exposure, after metabolic perturbations (such as hyperglycemia) were set up, which could imply that perturbations in the gut microbiota are a consequence of the pesticide-induced metabolic disorders and not the cause. Complementary experiments would be necessary to fully understand the role of the gut microbiota in the diabetogenic impact of our pesticide mixture in females.

**The Role of Constitutive Androstane Receptor in the Dimorphic Response to Pesticide Exposure**

One of the main findings of our experiment was that the response to pesticide exposure was sexually dimorphic. This sexual
dimorphism appeared to be related to CAR activity. Compared to WT male mice fed control chow, those fed pesticide chow exhibited higher expression levels of xenobiotic metabolizing enzymes in the liver, particularly those reflecting increased CAR activity (Cyp2b9 and Cyp2b10) (Maglich et al. 2002). As expected, this difference in the expression CAR-dependent xenobiotic metabolizing enzymes was not seen with CAR−/− males fed control or pesticide chow. Moreover, pesticide-exposed CAR−/− males did not exhibit greater BW gains or changes in glucose metabolism compared to those fed control chow. As urinary profiles of pesticide metabolites were similar between CAR−/− and WT males, we hypothesize that CAR was likely not involved in pesticide metabolism in males and that CAR-dependent changes in pesticide metabolism cannot account for the observed protective effects in CAR−/− males upon pesticide exposure.

Unlike males, CAR activity and other xenobiotic detoxifying enzymes were not induced in livers from WT females upon pesticide exposure, therefore suggesting extrahepatic pesticide metabolism. However, the urinary pesticide metabolite profile of female CAR−/− mice was different from that of their WT counterparts, suggesting that CAR played a role in pesticide detoxification in females. CAR−/− females fed pesticide chow also exhibited larger BWs throughout most of the experiment and exhibited higher mortality than those fed control chow. Reduced metabolic capacities in female CAR−/− mice may lead to accumulation of the parent compounds and may be more toxic than their metabolites or may accumulate in vital organs, explaining the high rate of mortality. Given our observation that gut microbiota–related metabolites are changed in urine of pesticide-exposed WT females and the fact that CAR is highly expressed in the intestine (Bookout et al. 2006), we speculate that CAR-dependent metabolism occurs in the gut. We additionally found no considerable differences in GSH:GSSG ratios between CAR−/− mice fed control and pesticide chow, suggesting no significant increase in oxidative stress. This may be attributed to the lack of PPARα activation, as determined by the similar pattern of expression of its prototypical targets genes in CAR−/− mice fed control and pesticide chow. Moreover, activation of PPARα has been shown to be involved in the control of the expression of antioxidant defense enzymes, such as those leading to the conjugation of GSH (Anderson et al. 2004). More experiments would be necessary to confirm these hypothesis. Moreover, the findings of altered PPARα target genes in the CAR−/− mice after pesticide exposure may reflect cross-talks between these two nuclear receptors as previously reported by others (Anderson et al. 2004; Rosen et al. 2017). The dimorphic effect of pesticides may also be linked to estrogenic protection. A growing body of evidence demonstrates a protective role of estrogenic signaling in the development of metabolic syndrome (Matic et al. 2013; Riant et al. 2009; Zhu et al. 2013, 2014). Thus, we postulate that in our model, females may be protected from pesticide-induced obesity and hepatic fat accumulation through estrogenic signaling.

**A Cocktail Effect of Pesticides**

We observed significant metabolic dysregulation in animals fed a pesticide mixture at doses at which each individual pesticide is supposed to not exert any health impact. The observed disturbances may be linked to a cocktail effect resulting from dose addition or a synergistic interaction between two or more compounds. Interactions between the pesticides in the mixture may occur at various levels or on various targets and/or cell signaling pathways (Delfosse et al. 2015; Rizzati et al. 2016). Organophosphates are well known endocrine–metabolic disruptors (Adigun et al. 2010a) acting on liver adenylyl cyclase, oxidative status, and proinflammatory markers (Adigun et al. 2010a; Lasram et al. 2015). Neonicotinoids have been shown to disrupt the thyroid (Bhaskar and Mohanty 2014). Carbamate may alter liver glucose, fat, and oxidative balance (Wang et al. 2014), and dicarboximide (captan) has been shown to reduce liver reduced glutathione (Della Morte et al. 1994). Comparison of the metabolic effects of the pesticide mixture vs. those of each of the single pesticides requires further investigations. Such comparisons are needed to understand whether a synergistic deregulation of metabolic pathways or an additive effect of pesticides on the induction of oxidative stress could explain the observed metabolic disruptions.

**Conclusions**

We demonstrated that a pesticide cocktail to which consumers may be exposed through food intake, induced at a nontoxic dose (TDI level as defined for human exposure but adjusted to the BW of mice) in a mouse model, caused metabolic disruption consistent with diabetic status. Moreover, the sexual dimorphic impact of this pesticide cocktail may be governed by distinct xenobiotic metabolic capacities and distinct nuclear receptor activation. Our results thus question the relevance of TDI levels for individual pesticides when present in a mixture.

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