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Effect of Organic Diet Intervention on Pesticide Exposures in Young Children Living in Low-Income Urban and Agricultural Communities

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BACKGROUND: Recent organic diet intervention studies suggest that diet is a significant source of pesticide exposure in young children. These studies have focused on children living in suburban communities.

OBJECTIVES: We aimed to determine whether consuming an organic diet reduced urinary pesticide metabolite concentrations in 40 Mexican-American children, 3–6 years of age, living in California urban and agricultural communities.

METHODS: In 2006, we collected urine samples over 16 consecutive days from children who consumed conventionally grown food for 4 days, organic food for 7 days, and then conventionally grown food for 5 days. We measured 23 metabolites, reflecting potential exposure to organophosphorous (OP), pyrethroid, and other pesticides used in homes and agriculture. We used linear mixed-effects models to evaluate the effects of diet on urinary metabolite concentrations.

RESULTS: For six metabolites with detection frequencies > 50%, adjusted geometric mean concentrations during the organic phase were generally lower for all children, and were significant for total dialkylphosphates (DAPs) and dimethyl DAPs (DMs; metabolites of OP insecticides) and 2,4-D (2,4-dichlorophenoxyacetic acid, a herbicide), with reductions of 40%, 49%, and 25%, respectively (p < 0.01). Chemical-specific metabolite concentrations for several OP pesticides, pyrethroids, and herbicides were either infrequently detected and/or not significantly affected by diet. Concentrations for most of the frequently detected metabolites were generally higher in Salinas compared with Oakland children, with DMs and metabololchol at near significance (p = 0.06 and 0.03, respectively).

CONCLUSION: An organic diet was significantly associated with reduced urinary concentrations of nonspecific dimethyl OP insecticide metabolites and the herbicide 2,4-D in children. Additional research is needed to clarify the relative importance of dietary and non-dietary sources of pesticide exposures to young children.

INTRODUCTION

Although most residential uses of many organophosphorous (OP) pesticides, including chlorpyrifos and diazinon, have been phased out since the mid-2000s due to potential health risks to children, they have continued to be used in agriculture [U.S. Environmental Protection Agency (EPA) 2000, 2001]. The use of OP pesticides in agriculture could result in ingestion of residues in food, and recent studies suggest that dietary intake of produce and juices may account for a significant proportion of OP pesticide exposure in young children (Lu et al. 2006b, 2008; Morgan et al. 2005; Smith-Spangler et al. 2012; Wilson et al. 2003). Some of the best evidence supporting these findings includes results from diet intervention studies where significant reductions in excreted urinary pesticide metabolites were observed in young children when they consumed an organic diet (Lu et al. 2006b, 2008). For example, Lu et al. (2008) showed that several OP pesticide metabolites in suburban children declined to undetectable concentrations during several days of eating organic food. These findings were consistent with an earlier observational study, which suggested that children consuming primarily organic food in non-agricultural households with no residential pesticide use have minimal or no pesticide exposures (Carl et al. 2003). Lu et al. (2006a, 2009) also reported a decrease in urinary pyrethroid pesticide metabolite concentrations in these children during the organic diet phase. The lower urinary pesticide metabolite concentrations found in children eating organic diets is consistent with food residue monitoring data that has shown lower pesticide residue levels in organic versus conventionally grown food [Baker et al. 2002; U.S. Department of Agriculture (USDA) 2008].

Other factors associated with children’s cumulative pesticide exposures include socioeconomic status and location of residence. For example, low-income children may experience higher exposures to pesticides, particularly pyrethroids, because of poor housing quality and associated pest infestations and home pesticide use (Bradman et al. 2005a; Quirós-Alcalá et al. 2011; Whyatt et al. 2002). Children living in agricultural areas, compared with children living in non-agricultural suburban areas, are exposed to higher ambient and residential contamination from drift or volatilization from nearby agricultural applications and take-home residue by farmworking parents (Bradman et al. 2011; Hamly et al. 2009; Lu et al. 2000; Quirós-Alcalá et al. 2011).

To date, organic food intervention studies have been conducted only in suburban/non-agricultural communities. Therefore, the relative contribution of dietary versus non-dietary pesticide exposures in low-income children living in urban and agricultural communities is not known. To address existing data gaps, we conducted an organic diet intervention study in young, low-income, Mexican-American children living in urban and agricultural communities. We report results for 23 urinary metabolites of OP and pyrethroid insecticides, as well as several herbicidal compounds.

METHODS

Study participants. We recruited a convenience sample of 40 children, 20 residing in an urban community in the Fruitvale area of Oakland, California, and 20 residing in...
Organic diet and child pesticide exposure

We measured 23 pesticide metabolites in urine specimens including specific and nonspecific metabolites for OP pyrethrin, and pyrethroid insecticides and select herbicides (Table 1 lists precursor compounds). Selection of metabolites was based on use of precursor compounds in the study locations at the county level, potential for pesticide exposure from residential and agricultural applications, and the availability of validated laboratory methods. We measured six dialkylphosphate (DAPs) metabolites, including diethyls (DEs) [diethylphosphate (DEP), diethylthiophosphate (DTP), and diethylthiophosphate (DDEDTP)] and dimethyls (DMs) [dimethylphosphate (DMP), dimethylthiophosphate (DMTP), and dimethylthiophosphate (DMDDTP). These nonspecific urinary OP pesticide metabolites derive from approximately 28 OP insecticides registered for use by the U.S. EPA, many of which are applied for agricultural and non-agricultural purposes (Table 1) in the participating communities.

We also measured five specific OP pesticide metabolites: CMH (coumaphos), IMPY (diazinon), CIT (isazophos), MDA (malathion), and DPY (pirimiphos-methyl). Other insecticide metabolites measured included two specific pyrethroid metabolites [4FP (cyfluthrin) and DBCA (deltamethrin)] and four nonspecific pyrethroid metabolites (3-PBA, CDCA, and cis-trans-DCCA). Specific metabolites of six herbicides were also measured [2,4-D (2,4-dichlorophenoxyacetic acid), 2,4,5-T (2,4,5-trichlorophenoxyacetic acid), acetochlor, alachlor, atrazine, and metolachlor], three of which (alachlor, atrazine, and metolachlor) were used only in the agricultural study location. All 594 samples were analyzed for DAP metabolites, whereas the number of samples analyzed for specific metabolites ranged between 532 and 594, depending on the amount of urine volume remaining after initial DAPs analysis.

Laboratory methods used to measure urinary DAPs have been described previously (Bravo et al. 2004). Briefly, we lyophilized specimens to remove water and redissolved...
Quality control (QC) procedures included repeat analysis of three in-house urine pools enriched with known amounts of metabolites residues with target values and confidence limits that were previously determined. Westgard rules for quality control were used to validate each analytical run (Caudill et al. 2008). We also analyzed duplicate samples within runs of the same sample (typically one sample per run, or about 5% replicates) to assess the precision of our analytical runs. These values were always within ±20% or else the samples were repeated. No metabolites were present in any blank samples, indicating that no contamination occurred in the field, during sample processing, or during shipment to the laboratory. Recoveries of repeat QC samples were 100 ± 10% and relative standard deviations were below 15%. Limits of detection (LOD) ranged between 0.05 and 0.30 ng/mL (Table 1). Metabolite concentrations below the LOD were imputed to LOD divided by the square root of 2 (Hornung and Reed 1990). Because individual OP pesticides can devolve to more than one DAP metabolite, we summed the DAPs on a molar basis to reflect total DM (i.e., molar sum of DMP, DMTP, and DMDTP) and total DE (i.e., molar sum of DEP, DETP, and DEDTP) metabolites. Total DAPs consisted of the molar sum of total DMs and total DEs. Creatinine concentrations were determined using a commercially available method (Vitros CREA slides; Ortho Clinical Diagnostics, Raritan, NJ).

**Data analysis.** We first summarized demographic characteristics for participating children. Before data analysis, we excluded urine samples collected during days 5 and 6 (considered washout days between the first conventional diet phase and the organic diet phase) and during days 12 and 13 (considered washout days between the organic diet phase and the second conventional diet phase) from the analysis. Overall analyte DF (%) is the ratio of the total number of urine samples with analyte concentration > LOD to the total number of urine samples collected in the study multiplied by 100.

### Table 1. Summary of precursor compounds, including reported usage by county, and measured metabolites in urine.

<table>
<thead>
<tr>
<th>Chemical class and precursor compounds</th>
<th>Metabolite measured (abbreviation)</th>
<th>Precursor compound use in 2006 (kg)</th>
<th>LOD (ng/mL)</th>
<th>Overall analyte DF (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Organophosphorus insecticides</td>
<td></td>
<td>Monterey</td>
<td>Alameda</td>
<td></td>
</tr>
<tr>
<td>C入户ophos</td>
<td>3-Chloro-4-methyl-7-hydroxycoumarin (CMH)</td>
<td>0</td>
<td>0</td>
<td>0.18</td>
</tr>
<tr>
<td>Diazinon</td>
<td>2-Isopropyl-4-methyl-6-hydroxprimidin (IMPY)</td>
<td>65,813</td>
<td>5</td>
<td>0.1</td>
</tr>
<tr>
<td>Isazorops</td>
<td>5-Chloro-1,2-dihydro-1-isopropyl-(3H)-1 (CI)</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Malathion</td>
<td>Malathion dicarboxylic acid (MDA)</td>
<td>16,940</td>
<td>8</td>
<td>0.05</td>
</tr>
<tr>
<td>Pirimiphos-methyl</td>
<td>2-Diethylamino-6-methylpyridin-4-ol (DPY)</td>
<td>0</td>
<td>0</td>
<td>0.2</td>
</tr>
<tr>
<td>Azinphos-methyl, chlorpyfros-methyl, dichlorvos, dicrotophos, dimethoxat, fenitrothion, fenthion, isazorops-methyl, malathion, methidathion, methyl parathion, naled, oxydemeton-methyl, phosmet, pirimiphos-methyl, temephos, tetrachlorvinphos, trichlorfon</td>
<td>Total diethylphosphates (total DMs = DMP + DMTP + DMDTP)</td>
<td>75,447</td>
<td>624</td>
<td>0.05</td>
</tr>
<tr>
<td>Chloroethoxyphos, chlorpyfros, coumaphos, diazinon, disulfoton, ethion, parathion, phorate, sulforoppe, terbufos</td>
<td>Total diethylphosphates (total DEs = DEP + DETP + DEDTP)</td>
<td>95,812</td>
<td>73</td>
<td>0.2</td>
</tr>
<tr>
<td>Pyrethrin and pyrethroid insecticides</td>
<td>Total diphosphanes (total DAPs = total DMs + total DEs)</td>
<td>171,259</td>
<td>697</td>
<td>ND</td>
</tr>
<tr>
<td>Cyfluthrin</td>
<td>4-Fluoro-3-phenoxbenzoic acid (4FP)</td>
<td>84</td>
<td>392</td>
<td>0.12</td>
</tr>
<tr>
<td>Deltamethrin</td>
<td>cis-2,2-(Dibromo)-2-dimethylvinylcyclopropane carboxylic acid (DBCA)</td>
<td>17</td>
<td>67</td>
<td>0.3</td>
</tr>
<tr>
<td>Allethrin, phenothrin, pallethrin, pyretrinins, resmethrin, tetramethrin</td>
<td>Chrysanthemum dicarboxylic acid (CDCA)</td>
<td>112</td>
<td>27</td>
<td>0.21</td>
</tr>
<tr>
<td>Allethrin</td>
<td>3-Phenoxybenzoic acid (3-PBA)</td>
<td>15,631</td>
<td>1,009</td>
<td>0.2</td>
</tr>
<tr>
<td>Allethrin, cihalothrin, cypemethrin deltamethrin, fenpropinox, permethrin, tralomethrin</td>
<td>cis-2,2-(Dichloro)-2-dimethylvinylcyclopropane carboxylic acid (cis-DDCA)</td>
<td>13,180</td>
<td>1,287</td>
<td>0.3</td>
</tr>
<tr>
<td>cis-Cypermethrin, cis-cyfluthrin, cis-permethrin</td>
<td>trans-2,2-(Dichloro)-2-dimethylvinylcyclopropane carboxylic acid (trans-DDCA)</td>
<td>13,180</td>
<td>1,287</td>
<td>0.3</td>
</tr>
<tr>
<td>Herbicides</td>
<td></td>
<td>Monterey</td>
<td>Alameda</td>
<td></td>
</tr>
<tr>
<td>2,4-Dichloroephenoxyacetic acid</td>
<td>2,4-Dichlorophenoxacetic acid (2,4-D)</td>
<td>0</td>
<td>0</td>
<td>0.1</td>
</tr>
<tr>
<td>2,4,5-Trichloroephenoxyacetic acid</td>
<td>2,4,5-Trichlorophenoxacetic acid (2,4,5-T)</td>
<td>0</td>
<td>0</td>
<td>0.1</td>
</tr>
<tr>
<td>Acetochlor</td>
<td>Acetochlor mercapturate (ACE)</td>
<td>0</td>
<td>0</td>
<td>0.21</td>
</tr>
<tr>
<td>Alachlor</td>
<td>Alachlor mercapturate (ALA)</td>
<td>46</td>
<td>46</td>
<td>0.3</td>
</tr>
<tr>
<td>Atrazine</td>
<td>Atrazine mercapturate (ATZ)</td>
<td>65</td>
<td>65</td>
<td>0.27</td>
</tr>
<tr>
<td>Metolachlor</td>
<td>Metolachlor mercapturate (MET)</td>
<td>1,550</td>
<td>1,550</td>
<td>0.1</td>
</tr>
</tbody>
</table>

**Abbreviations:** DF, detection frequency; LOD, limit of detection; ND, not detected.

* California EPA Department of Pesticide Regulation (Cal/EPA DPR) Pesticide Use Reporting (PUR) Database (CalEPA DPR 2006a, 2006b). *Amount applied (kg) includes agricultural, landscape maintenance, and structural pest control uses reported in each county; by law all agricultural pesticide use (includes applications to parks, golf courses, cemeteries, range- land, pastures, and along roadside and railroad rights-of-way) must be reported to the CalEPA DPR. The PUR database does not provide amount applied for each individual permethrin isomer (cis-, trans-); formulations applied may consist of a combination of two or more isomers. *The following insecticides were not applied in Monterey or Alameda County in 2006: Chlorpyfros-methyl, dichlorvos, dicrotophos, fenitrothion, fenthion, isazorops-methyl, pirimiphos-methyl, temephos, tetrachlorvinphos, trichlorfon, chlorethoxyphos, coumaphos, ethion, parathion, sulfoxep, and terbufos. *Overall analyte DF (%) is the ratio of the total number of urine samples with analyte concentration > LOD during the entire study to the total number of urine samples collected in the study multiplied by 100.
phase) from our analysis (Figure 1). We also excluded urine samples from children who reported eating at a restaurant or eating at a friend's house on the preceding day during the organic diet phase, or eating organic food on the preceding day during the second conventional diet phase. Additionally, if the family reported pesticide use in or around the home during the study period, then observations for metabolites of the active ingredients in the formulation applied were excluded for the day of and day after pesticide application. If the active ingredients were unknown, then observations were excluded for all metabolites. Last, we excluded pyrethroid metabolite results for one child in Salinas with mean 3-PBA concentrations > 3 SDs away from the mean of the other children. Based on the aforementioned exclusions, the total number of urine samples included in our final statistical models varied for each metabolite and ranged from 331 to 398 samples. Final statistical models focused on metabolites with an overall detection frequency in urine of ≥ 50%. Metabolite concentrations were log10-transformed for statistical analyses.

We computed descriptive statistics for frequently detected metabolites (n = 6; total DMs, total DEs, total DAPs, metolachlor mercapturate (MET), 2,4-D, 3-PBA) by diet phase. We then used linear mixed-effects models to account for the correlation among repeat urine samples collected from the same individual and determined whether mean metabolite concentrations differed between the organic diet phase and each of the conventional diet phases separately. These comparisons were conducted for all children and then stratified by location. To control for the large number of statistical tests (i.e., 36 multiple comparisons), we used the Hochberg procedure (Hochberg 1988).

We then conducted analyses using adjusted linear mixed-effects models to evaluate the effect of an organic diet on children's urinary metabolite concentrations controlling for location of residence and whether the sample was a first morning void or not. We allowed for random intercepts, and the covariance matrix was assumed to have identical variances and all covariances equal to zero. Effects from location, diet, and first morning void were fixed. The variance–covariance matrix of the parameter estimates was calculated using the robust sandwich estimator. For these analyses, we defined a binary diet variable, which combined all urine samples from both conventional diet phases into one category and the samples from the organic diet phase into another because no statistically significant differences were observed between metabolite concentrations in C1, the first conventional diet phase, compared with C2 (not shown). We also assessed whether the effect of the organic diet differed by location by including an interaction term; presence of interaction was established at p < 0.20. If the interaction term was not significant, we ran the model without it for our estimations. In those cases where interaction between location and diet was present, we used postestimation procedures to obtain the marginal effects of the organic diet for children at each location and the overall average marginal effect for all children. Unless otherwise noted, significance was set at an adjusted p < 0.05.

Additionally, we used the Kruskal–Wallis test to examine whether the number of servings of different food categories (e.g., vegetables, fruits) changed during the three diet phases. If this test suggested a significant difference, we then performed a pairwise Wilcoxon rank sum test to determine which diet phases differed.

For statistical analyses, we present results that are not adjusted for creatinine, to be consistent with other studies (Barr et al. 2005; Bradman et al. 2011; Lu et al. 2008). Analyses were repeated with creatinine-adjusted values to confirm our bivariate results. In addition, we performed all multivariate models with creatinine-adjusted urinary metabolite concentrations for comparison with final models.

Results

Demographic characteristics. Forty children 3–6 years of age participated in this study: 19 boys and 21 girls. Demographic characteristics were similar between study locations (see Supplemental Material, Table S2). The mean (± SD) age for children was 4.5 ± 1.1 years and 4.8 ± 1.2 years in the urban (Oakland, n = 20; 10 boys and 10 girls) and agricultural community (Salinas, n = 20; 9 boys and 11 girls), respectively. The majority of participants in both communities (65%) were within 200% of the U.S. federal poverty threshold, and almost all parents (95%) reported that they were married or living as married. Most homes (> 80%) in each community were located > 0.25 mile from the nearest agricultural field or golf course (homes in the agricultural community were generally removed from agricultural fields); four homes in Salinas were located < 0.25 mile from the nearest agricultural field. Overall, 23% of participants reported home pesticide use during the study period (30% and 15% of urban and farmworker households, respectively). However, participants in the urban community of Oakland reported more home pesticide use in the 3 months preceding the study compared with participants in the farmworker community of Salinas (65% vs. 30%, respectively).

Frequency of food consumption. Information collected in food frequency diaries indicated that overall dietary consumption habits did not change during the different diet phases (see Supplemental Material, Table S1). Fruit and grain consumption, however, was higher in the organic diet phase compared with the conventional diet phases (p < 0.05).

Pesticide metabolite levels in urine. Of the 13 pesticide-specific metabolites measured, 2 had overall detection frequencies (DFs) > 50%: 2,4-D at 90% and MET at 72% (Table 1). The nonspecific pyrethroid metabolite 3-PBA had an overall DF of 82%. Overall DFs for total DMs, DEs, and DAPs were 92%, 74%, and 95%, respectively. Thus, all but 6 metabolites measured in this study had DFs < 50%. The distributions of these 6 metabolites are presented in the Supplemental Material, Table S3. The DFs during conventional and organic diet phases for metabolites not frequently detected are presented in the Supplemental Material, Table S4.

Results of bivariate analyses. Among all children (n = 40), the geometric mean (GM) of total DMs, DAPs, and 2,4-D decreased during the organic diet phase by –48.7% [95% confidence interval (CI): –68.5%, –24.9%; p = 0.001], –45.3% (95% CI: –62.6%, –15.7%; p = 0.005), and –32.8% (95% CI: –45.4%, –14.1%; p = 0.001), respectively. After adjusting for multiple comparisons, only those results with p < 0.001 remained significant.

Between total DM concentrations were consistently much higher than DE concentrations, the findings for total DAPs were driven by total DMs, which break down from insecticides such as oxydemeton-methyl and malathion. Concentrations of total DEs, MET, and 3-PBA were not significantly affected by diet. No statistically significant differences were found between metabolite levels in C1 compared with C2 (see Supplemental Material, Table S3).

Estimated effect of diet on urinary metabolite levels using multivariate models. All children. Results from multivariate models showed significantly lower metabolite concentrations of total DMs (–48.7%; 95% CI: –65.7%, –23.2%), total DAPs (–39.9%; 95% CI: –58.6%, –12.6%), and 2,4-D (–25.2%, 95% CI: –38.0%, –9.7%) (p < 0.01)
during the organic diet phase compared with the combined conventional diet phases (Table 2 and Figure 2). We also observed lower metabolite concentrations during the organic diet phase, albeit not significantly, for total DEs, MET, and 3-PBA.

Except for 2,4-D, the linear mixed-effects model results using creatinine-adjusted metabolites were similar to models with metabolites not adjusted for creatinine (see Supplemental Material, Table S5 and Figure S1).

Among the six frequently detected metabolites, we observed significant interactions ($p < 0.20$) between location and diet only for total DEs and 3-PBA (see Table 2).

Analyses evaluating the effect of location are described below.

**Oakland children.** For children living in Oakland, switching from a conventional diet to an organic diet had the same effect on mean total DM, total DAP, and 2,4-D levels as observed for all children, because there was no interaction between diet and location for these compounds (results presented above). Switching to an organic diet was associated with a $-32.7\%$ (95% CI: $-48.8, -11.7\%$) decrease in mean 3-PBA levels ($p = 0.004$) among Oakland children. The opposite effect was observed for mean concentrations of total DEs, where an increase of 36.4% (95% CI: $-29.0, 162.0$) was observed during the organic diet phase. However, this large relative change reflected small absolute differences and was nonsignificant ($p = 0.352$) (Table 2 and Figure 2).

**Salinas children.** For children living in Salinas, the effect of switching from a conventional to an organic diet on the mean concentrations of total DMs, total DAPs, and 2,4-D metabolites was again the same as for all children because of the lack of interaction with location (results presented above). Contrary to what was observed in Oakland, among Salinas children, switching from a conventional to an organic diet was associated with a 21.5% (95% CI: $-8.9, 62.0$) increase in mean 3-PBA levels and a $-28.1\%$ (95% CI: $-48.8, -11.7\%$) decrease in mean 2,4-D levels.

### Table 2. Estimated effect of an organic diet (vs. conventional) on the geometric mean for frequently detected metabolites.a

<table>
<thead>
<tr>
<th>Children</th>
<th>Total DEs</th>
<th>Total DMs</th>
<th>Total DAPs</th>
<th>MET</th>
<th>2,4-D</th>
<th>3-PBA</th>
</tr>
</thead>
<tbody>
<tr>
<td>All ($n=40$)</td>
<td>$-1.2%$</td>
<td>$-48.7%$</td>
<td>$-39.9%$</td>
<td>$-6.4%$</td>
<td>$-25.2%$</td>
<td>$-13.3%$</td>
</tr>
<tr>
<td>Percent change (95% CI)</td>
<td>$-35.3, 51.0$</td>
<td>$-65.7, -23.2$</td>
<td>$-58.6, -12.6$</td>
<td>$-18.6, 7.5$</td>
<td>$-38.0, -9.7$</td>
<td>$-28.9, 5.8$</td>
</tr>
<tr>
<td>$p$-Value</td>
<td>0.957</td>
<td>0.001</td>
<td>0.008</td>
<td>0.350</td>
<td>0.002</td>
<td>0.159</td>
</tr>
<tr>
<td>Oakland ($n=20$)</td>
<td>$36.4%$</td>
<td>$-29.0, 162.0$</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Percent change (95% CI)</td>
<td>$32.1$</td>
<td>$-8.0, 23.1$</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>$-32.7, -11.7$</td>
</tr>
<tr>
<td>$p$-Value</td>
<td>0.352</td>
<td>0.004</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Salinas ($n=20$)</td>
<td>$-28.1%$</td>
<td>$-58.0, 23.1$</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Percent change (95% CI)</td>
<td>$-25.2$</td>
<td>$-8.9, 62.0$</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>$p$-Value</td>
<td>0.229</td>
<td>0.185</td>
<td>0.003</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Interaction</td>
<td>0.231</td>
<td>0.134</td>
<td>0.366</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
</tbody>
</table>

Abbreviations: CI, confidence interval. *Marginal results by location are omitted if observed interaction between location and diet was not significant ($p > 0.2$). In these cases only the model without interaction term is presented for all children.

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**Figure 2.** Estimated marginal adjusted GMs and 95% CIs for select urinary metabolites based on diet followed after fitting of linear mixed-effects models. All models were adjusted for type of void (first morning void vs. random spot sample). Models for “All children” were also adjusted for location (Oakland vs. Salinas); an interaction term for location and diet was included in these models for total DEs and 3-PBA ($p_{interaction} < 0.20$). $p$-Values reported in the figure indicate whether there were significant differences observed in metabolite concentrations between diet phases by location. $p$-Values reported in the x-axis indicate significance for the difference in metabolite concentrations between locations regardless of diet.
Salinas versus Oakland children. Among the six metabolites with detection frequencies > 50%, we found that concentrations were higher, with the exception of 3-PBA, in Salinas children compared with Oakland children, regardless of diet (p-values displayed on Figure 2). We observed somewhat higher adjusted GM concentrations for DM metabolites in Salinas versus Oakland children: GM = 97.7 (95% CI: 75.04, 127.09) versus GM = 66.9 (95% CI: 49.90, 89.67), respectively (p = 0.06); and significantly higher MET concentrations in Salinas versus Oakland children: GM = 0.18 (95% CI: 0.16, 0.20) versus GM = 0.14 (95% CI: 0.13, 0.17), respectively (p = 0.03). Conversely, 3-PBA concentrations were significantly higher in Oakland than in Salinas children; GM = 0.76 (95% CI: 0.60, 0.97) versus GM = 0.42 (95% CI: 0.34, 0.52), respectively (p < 0.001).

Discussion

Metabolites representing several classes of pesticides, including OP and pyrethroid insecticides and the herbicides 2,4-D and metolachlor, were frequently detected (> 72%) in urine samples collected from participating children. Multivariate analyses indicated that consuming an organic diet significantly lowered urinary concentrations of total DMs, total DAPs, and 2,4-D for children in both rural and urban locations. Among other frequently detected analytes, nonspecific diethyl OP pesticide metabolites and the pyrethroid metabolite 3-PBA, as well as MET, were not significantly lower for all children during the organic diet phase. Specific metabolites for other pesticides used on produce or grain were not frequently detected, including metabolites of the OP insecticides diazinon, malathion, and pirimiphos and the herbicides 2,4,5-T, acetochlor, and atrazine.

Similar findings of significantly reduced total DM and DAP metabolite levels during an organic diet phase were recently reported in a crossover study of 13 adults living in Melbourne, Australia (Oates et al. 2014). Results from that study showed that mean total DAP metabolite levels were 89% lower during the 1-week organic diet phase compared with the conventional diet phase. The researchers (Oates et al. 2014) found an even greater reduction (96%) for total DMs. In addition, Curl et al. (2015) recently found that urinary DAP concentrations were significantly lower among participants in the Multi-Ethnic Study of Atherosclerosis (MESA) who reported more frequent consumption of organic produce (p < 0.02). Our finding that DAP metabolite levels (i.e., nonspecific OP pesticide metabolites) declined during the organic diet phase is also consistent with findings for specific metabolites by Lu et al. (2006b, 2008) showing that fruit and vegetable consumption is associated with OP pesticide–specific urinary metabolite levels in children. As total urinary DM concentrations were consistently much higher than DE concentrations, findings for total DAPs in our study were driven by total DMs, which break down from insecticides such as azinphos-methyl, commonly used on tree-fruit crops consumed by children. In previous studies, we have observed much higher variability and instability of DE metabolites (Bradman et al. 2007, 2013), which may have limited our power to examine the effect of diet on total DE metabolite concentrations.

Our finding that an organic diet was not associated with a significant reduction in pyrethroid metabolite 3-PBA excretion for all children is not surprising given that these pesticides are primarily used in and around homes and not commonly applied to food crops; the finding is also consistent with Lu et al. (2006a), who reported that residential use is a more significant pyrethroid exposure factor for children than a conventional diet. Although we did observe a significant decrease in 3-PBA concentrations for Oakland children, we did not observe a significant reduction in metabolite levels for Salinas children during the organic diet phase. In contrast to prior organic diet intervention studies (Lu et al. 2006b, 2008), MDA detection frequencies were low in our population, and it was not possible to examine trends related to diet.

Several studies indicate that dietary intake is a potential route of exposure for herbicides. For example, Morgan et al. (2008) detected 2,4-D in approximately 46% of composite food samples at concentrations up to 20 ng/g. Similarly, Wilson et al. (2003) detected 2,4-D in 100% of liquid food samples and 89% of solid food samples, and estimated that dietary ingestion accounted for approximately 94% of total 2,4-D exposure in young children from combined dietary and nondietary ingestion and inhalation. In the 2003–2005 Food and Drug Administration's Total Diet Study (FDA 2003, 2005), the FDA detected 2,4-D in only a few types of cereals and grains; however, their laboratory detection limits were higher than other studies. Overall, these studies indicate that 2,4-D may be present in food and support our finding that the lower levels observed in our population during the organic diet phase were attributable to lower dietary exposure.

Herbicides such as metolachlor and atrazine have been tested in a wide variety of foods with very low detection frequencies (USDA 2007, 2013). Limited testing for other herbicides including alachlor and acetochlor also shows low detection frequencies in food (USDA 2007, 2013). Although many of these herbicides have been frequently detected in drinking water, the monitoring data suggest that solid foods are a less important source of exposure to these compounds (USDA 2007, 2013), consistent with our finding that organic diet was not associated with significantly reduced excretion of metabolites for these compounds. All of the herbicides for which we measured metabolites (2,4-D, 2,4,5-T, acetochlor, alachlor, atrazine, and metolachlor) had relatively low or no agricultural use in the regions we studied (Table 1) (California Environmental Protection Agency, Department of Pesticide Regulation (CalEPA DPR) 2006a, 2006b), and incidence of water contamination in our study regions is rare (CalEPA DPR 2006c).

The higher pyrethroid metabolite 3-PBA concentrations we observed in the Oakland children compared with Salinas were consistent with reported higher recent use of home pesticides, as well as higher pyrethroid pesticide residues that we measured previously in these children's homes (Quirós-Alcalá et al. 2011). Concentrations of metolachlor were higher in Salinas children compared with Oakland children regardless of diet. This finding is consistent with zero use of metolachlor reported in Alameda County (where the Oakland homes are located) (Table 1) compared with 1,550 kg applied in Monterey County (where Salinas homes are located). The somewhat higher levels of DMs in Salinas than in Oakland children is consistent with our results showing primarily higher levels of DMs than of DEs in Salinas pregnant women participating in the CHAMACOS (Center for the Health Assessment of Mothers and Children of Salinas) study compared with women of reproductive age in NHANES (Bradman et al. 2005b).

Overall, we found few differences in food choices between the conventional and organic diet phases in this study. However, there appeared to be slightly higher intake of fruit and grains among participants during the organic diet phase (see Supplemental Material, Table S1). These differences are unlikely to have confounded our results. For example, several studies have shown that higher produce intake is associated with higher urinary pesticide metabolite levels (Curl et al. 2015; Lu et al. 2005, 2006b; Oates et al. 2014). Thus, without the organic diet, we would expect that more fruit in general would lead to higher urinary metabolite levels, the opposite of what we found for several metabolites.

This study has several limitations. Urinary metabolite concentrations for some insecticides (e.g., DAPs) may reflect exposure to precursor pesticide compounds or preformed metabolites in food or the environment (Lu et al. 2005; Quirós-Alcalá et al. 2012; Zhang et al. 2008). Thus, reductions in urinary metabolite levels...
during the organic diet phase may, in part, be attributable to reduced intake of preformed metabolites from eating organic food, which presumably has fewer preformed metabolites because it was not treated with pesticides that can further breakdown. However, food monitoring data indicate that conventional foods have significantly higher pesticide residues compared with organic food (Baker et al. 2002; Forman et al. 2012; USDA 2008), suggesting that the reductions in child pesticide urinary metabolites during organic diet phases are at least partly attributable to reductions in precursor pesticide exposure. Given that OP urinary metabolite levels, especially in pregnant women, have been associated with poorer neurodevelopment in children (Bouchard et al. 2011; Muñoz-Quezada et al. 2013), future research clarifying the contribution of preformed metabolites to human exposure is critically needed to inform exposure and risk assessment studies. Finally, this study was conducted in primarily low-income Mexican-American children, which may limit its generalizability to other populations.

Conclusion

In summary, consistent with other studies, urinary 2,4-D and two measures of OP pesticide exposure (total DMs and total DAP metabolites) were lower in children eating an organic diet. Other frequently detected metabolites for pyrethroids, diethyl OP pesticides, and the herbicide metolachlor were not significantly lower during the organic diet phase. Further, several compound-specific herbicide and OP pesticide metabolites had low detection frequencies, indicating that diet was not an important exposure source for these pesticides (e.g., diazinon, malathion) in this population. Last, independent of diet, most frequently detected metabolites were generally higher in Salinas compared with Oakland children, with DMs and metolachlor at or near significance ($p = 0.06$ and 0.03, respectively), suggesting additional sources of pesticide exposure for children living in agricultural communities. Additional research is needed to clarify the relative importance of dietary and nondietary sources of pesticide exposures in young children and determine the proportion of urinary metabolite excretion attributable to preformed metabolites.

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