*Ann. Occup. Hyg.*, Vol. 51, No. 1, pp. 53–65, 2007
Published by Oxford University Press
doi:10.1093/annhyg/mel062

# Urinary Pesticide Concentrations Among Children, Mothers and Fathers Living in Farm and Non-Farm Households in Iowa

**BRIAN D. CURWIN[1]*, MISTY J. HEIN[1], WAYNE T. SANDERSON[2], CYNTHIA STRILEY[3], DICK HEEDERIK[4], HANS KROMHOUT[4], STEPHEN J. REYNOLDS[5] and MICHAEL C. ALAVANJA[6]**

*[1]National Institute for Occupational Safety and Health, Division of Surveillance, Hazard Evaluations and Field Studies, Cincinnati, OH, USA; [2]University of Iowa, Department of Occupational and Environmental Health, Iowa City, IA, USA; [3]National Institute for Occupational Safety and Health, Division of Applied Research and Technology, Cincinnati, OH, USA; [4]Institute for Risk Assessment Sciences, Utrecht University, Utrecht, The Netherlands; [5]Colorado State University, Department of Environmental and Radiological Health Sciences, Fort Collins, CO, USA; [6]National Cancer Institute, Division of Cancer Epidemiology and Genetics, Rockville, MD, USA*

Received 10 March 2006; in final form 17 July 2006; published online 19 September 2006

**In the spring and summer of 2001, 47 fathers, 48 mothers and 117 children of Iowa farm and non-farm households were recruited to participate in a study investigating take-home pesticide exposure. On two occasions ~1 month apart, urine samples from each participant and dust samples from various rooms were collected from each household and were analyzed for atrazine, metolachlor, glyphosate and chlorpyrifos or their metabolites. The adjusted geometric mean (GM) level of the urine metabolite of atrazine was significantly higher in fathers, mothers and children from farm households compared with those from non-farm households ($P \leqslant 0.0001$). Urine metabolites of chlorpyrifos were significantly higher in farm fathers ($P = 0.02$) and marginally higher in farm mothers ($P = 0.05$) when compared with non-farm fathers and mothers, but metolachlor and glyphosate levels were similar between the two groups. GM levels of the urinary metabolites for chlorpyrifos, metolachlor and glyphosate were not significantly different between farm children and non-farm children. Farm children had significantly higher urinary atrazine and chlorpyrifos levels ($P = 0.03$ and $P = 0.03$ respectively) when these pesticides were applied by their fathers prior to sample collection than those of farm children where these pesticides were not recently applied. Urinary metabolite concentration was positively associated with pesticide dust concentration in the homes for all pesticides except atrazine in farm mothers; however, the associations were generally not significant. There were generally good correlations for urinary metabolite levels among members of the same family.**

*Keywords:* biological monitoring; herbicides; insecticides; pesticides; pesticide exposure; take-home; urine

## INTRODUCTION

Farmers are the biggest users of pesticides applying ~540 million kilograms in 1999 in the United States; herbicides accounted for the largest proportion of this amount with ~240 million kilograms applied (USEPA 2002a). Concern for pesticide exposure among the children of farmers and farm workers was raised by the National Institute for Occupational Safety and Health (NIOSH) with the Report to Congress on Workers' Home Contamination Study Conducted Under the Workers' Family Protection Act (29 U.S.C. 671a) (NIOSH, 1995). The Natural Resources Defense Council (NRDC) considers pesticides to be one of the top five environmental threats to children's health and considers farm children to be the most highly pesticide-exposed subgroup in the United States (NRDC, 1998).

*Author to whom correspondence should be addressed:
Tel: +1 513 841 4432; fax: +1 513 841 4486;
e-mail: bcurwin@cdc.gov

Children and spouses of farmers are potentially exposed to pesticides indirectly by take-home contamination; pesticides can be tracked into farm homes on the clothing and shoes of farmers. We previously reported that the majority of farmers changed out of their work clothes and shoes inside the home (Curwin *et al*., 2002). Pesticide track-in has been clearly demonstrated after residential application of herbicides to lawns. Nishioka *et al*. (1999, 2001) measured the distribution of the herbicide 2,4-D in homes within a week of a lawn application and showed that transport mechanisms were dominated by track-in from active dogs, the home-owner's contaminated shoes and the children's shoes when worn indoors. Lewis *et al*. (2001) found that chlorpyrifos residues in indoor air and in carpet dust were higher within a few days after an exterior residential application than before the application and suggested that track-in was the principal source of these residues.

Several studies have found that farm homes have a greater frequency of detectable residues of pesticides and higher concentrations of pesticides in dust than in reference homes, potentially leading to greater exposure to pesticides among family members (Simcox *et al*., 1995; Bradman *et al*., 1997; Lu *et al*., 2000; Curl *et al*., 2002; Fenske *et al*., 2002; McCauley *et al*., 2003). Pesticide urine concentrations among the children of farmers and farm workers have been shown to be elevated when compared with children of non-farm families (Loewenherz *et al*, 1997; Lu *et al*., 2000) and pesticide levels in house dust have been correlated with urinary pesticide levels in children and adults living in the home.

Although the literature is inconclusive, pesticide exposure is thought to be associated with a variety of health effects including cancer, reproductive disorders, neurotoxicity and endocrine disruption (Maroni and Fait, 1993; Dich *et al*., 1997; Zahm *et al*., 1997; Kirkhorn and Schenker 2002; Richter and Chlamtac 2002; Alavanja *et al*., 2004a). More specifically, phenoxy herbicides (e.g. 2,4-D) have been associated with a number of cancers including soft tissue sarcomas, non-Hodgkin's lymphoma (NHL), cancer of stomach, colon and prostate; triazine herbicides (e.g. atrazine) have been associated with ovarian cancer; organophosphate insecticides (e.g. chlorpyrifos) have been associated with delayed neuropathy, chromosome aberrations, central nervous system alterations and NHL (Maroni and Fait, 1993); metolachlor has been associated with lung cancer (Alavanja *et al*., 2004b) and interuterine growth retardation (Munger *et al*., 1997); and glyphosate has been associated with adverse neurobehavioral development (Garry *et al*., 2002). Further, parental occupation involving pesticide application has been associated with childhood cancers (Daniels *et al*., 1997; Zahm and Ward, 1998; Flower *et al*., 2004) and household pesticide use has been associated with childhood leukemia (Ma *et al*., 2002).

Differences in children's physiology, behavior patterns and hygiene may result in significantly greater exposures of children to environmental contaminants than adults (National Academy of Sciences, 1993; Bearer, 1995; Health Council of the Netherlands, 2004). Small children spend much of their time on the floor or ground and are very likely to come into contact with pesticide residues on carpets or uncovered floors when playing inside and yard dirt when playing outside (Renwick, 1998). These factors can result in different sources and levels of pesticide exposure for children than adults in the same scenario (Garry, 2004). Children may also be more susceptible than adults to the toxic effects of pesticides, due to the sensitivity of developing organ systems. Older children, through their increased mobility and ability to assist with farm work, may have opportunities for direct contact with pesticide products. Although the public health importance of preventing injury to farm families has been well recognized, the hazards of exposure to pesticides and other chemicals to families in the farm environment have received relatively little attention.

A study was initiated to investigate agricultural pesticide contamination inside farm homes and family exposure to agricultural pesticides (Curwin *et al*., 2002, 2005a, b). The goal of the study was to evaluate pesticide exposure among farm families and compare their exposure to non-farm controls. The objectives presented in this paper are 2-fold: (i) to measure urinary pesticide levels among farm and non-farm families in Iowa and (ii) to ascertain what factors may influence these levels.

## METHODS

In Iowa in the spring and summer of 2001 farm and non-farm households were recruited to participate in the study. Participant recruitment has been described in more detail previously (Curwin *et al*., 2002). In short, recruitment was conducted by convenience sampling. To be eligible for the study, households had to have at least one child under the age of 16 years. Non-farm households had to be on land that was not used for farming, and nobody in the household could be working in agriculture or commercial pesticide application. Farm households had to be using at least one of the seven target pesticides—atrazine, acetochlor, metolachlor, alachlor, chlorpyrifos, glyphosate and 2,4-D. The target pesticides were selected because of their extensive use in Iowa agriculture. All the pesticides are corn or soybean herbicides, with the exception of chlorpyrifos, which is an insecticide used on corn. A total of 25 farm households [24 fathers, 24 mothers and 66 children

(29 female and 37 male)] and 25 non-farm households [23 fathers, 24 mothers, and 51 children (19 female and 32 male)] were enrolled in the study. Only the results for atrazine, metolachlor, chlorpyrifos and glyphosate are reported to due to limitations of analytical methods for the other pesticides in urine (e.g. cross-reactivity with other chemicals, poor analytical methods). NIOSH Human Subject Review Board approved the study.

*Sample collection*

During May–August, 2001, each household was visited on two occasions. The first visit was shortly after a pesticide application event (within 1–5 days) and the second visit was ~4 weeks later (average 4 weeks, range 3–5 weeks). Two spot urine samples were collected from the participants at each visit, one in the evening on the day of the visit and one the following morning. Urine samples were collected in 500 ml Nalgene® bottles and participants were asked to store the urine in their refrigerator or in a provided cooler with ice packs. Samples were collected the day after the visit and 25 ml aliquots were removed, stored on dry ice and shipped to the laboratory. The total volume of each urine void was recorded. Dust sample collection and analysis have been described previously (Curwin *et al*., 2005a). Briefly, dust samples were collected at each visit from various rooms in the homes using the HVS3 vacuum sampler [Cascade Stamp Sampling Systems (CS3) Inc., Sandpoint, ID] according to the American Society for Testing Material (ASTM) Standard Practice for Collection of Dust from Carpeted Floors for Chemicals (ASTM, 2000).

A questionnaire was administered to all participants at the first visit and re-administered at the second visit. Questions were asked about crops grown, use of personal protective equipment (PPE), crop size, pesticides used, dates and hours of application, who applied the pesticide, and the number of acres applied. This information was gathered from the start of the 2001 growing season until the second visit and generally reflected the early 2001 growing season among the participants. Information on children's age, weight, height and sex was also collected.

*Sample analysis*

The metabolites of four pesticides—atrazine (atrazine mercapturate), chlorpyrifos [3,5,6-trichloro-2-pyridinol (TCP)], metolachlor (metolachlor mercapturate), and glyphosate (parent glyphosate)—were analyzed in urine samples using immunoassay techniques. Immunoassay is cheaper, requires less sample and is faster than traditional GC or HPLC methods. The analytical limits of detection (LOD) varied by analyte and were 1.16, 3.32, 0.3 and 0.9 μg/l for atrazine mercapturate, TCP, metolachlor mercapturate and glyphosate, respectively. Urinary creatinine was measured using a commercially available enzyme slide technology (Vitros 250 Chemistry System, Ortho-Clinical Diagnostics, Raritan, NJ). All the methods described below have been validated and published elsewhere (Biagini *et al*., 1995, 2004; Striley *et al*., 1999; MacKenzie *et al*., 2000; Hines *et al*., 2003; Rubio *et al*., 2003).

Immunoassay for Atrazine (A00071) RaPID Assay® enzyme-linked immunosorbent assay (ELISA) kit (Strategic Diagnostics, Newtown, PA) was used to determine the metabolite atrazine mercapturate according to the manufacturer's instructions with the following exception: calibration standards (0.0, 0.1, 0.5, 1.0, 2.0 and 5.0 ppb) were prepared by fortifying pooled urine from anonymous volunteers diluted 1:10 with UriSub (CST Technologies Inc., Great Neck, NY) with synthesized atrazine mercapturate. All participant urine samples were diluted 1:10 with UriSub.

A previously published immunoassay for TCP (A00208) RaPID Assay® ELISA (Strategic Diagnostics, Newtown, PA) was used to determine the urinary metabolite of chlorpyrifos (MacKenzie *et al*., 2000) according to the manufacturer's instructions with the following exception: calibration standards (0.0, 0.0156, 0.3125, 0.625, 1.25, 2.5, 5.0 and 10.0 ppb) were prepared by fortifying UriSub with 3,5,6 trichloro-2-pyridinol. All participant urine samples were diluted 1:10 with UriSub. In addition, each sample was treated with 20 μl of β-glucuronidase (Roche Diagnostics, Part# 1-585-665, Mannheim, Germany) for 30 min at room temperature prior to the analysis in order to cleave 3,5,6 trichloro-2-pyridinol from its glucuronide conjugate form.

Glyphosate and metolachlor mercapturate were measured simultaneously in urine using a newly developed fluorescence covalent microbead immunoassay (FCMIA) (Biagini *et al*., 2004). Pesticide–protein conjugates for each of the pesticides were coupled to separate addressable sets of microbeads. The conjugate coupled microbeads were then used in a competitive assay for the pesticides. The pesticide in solution competed with the bead-bound conjugate for fluorescently labeled anti-pesticide antibodies. Thus increasing concentrations of a given pesticide in urine resulted in decreasing fluorescence signals from the microbead for that pesticide. The coupling of different pesticide conjugates to separate addressable sets of microbeads allows simultaneous measurement of the two pesticides. Calibration standards (0.0, 0.1, 0.3, 1.0, 3.0, 10.0, 30.0, 100 and 300 ppb) were prepared. Pooled urine diluted 1:10 in a mixture of assay buffer (Abraxis LLC, Hatboro, PA) and UriSub (1:3) were fortified with glyphosate and metolachlor mercapturate. An aliquot of 250 μl of the fortified mixture was treated with 20 μl of derivitizing agent

**Table 1.** Number and percentage of urine levels reported as non-detect (ND), positive but below the limit of detection (LOD), or greater than or equal to the LOD

| Pesticide subject | Household type | Number of | | | Urine level | | | *P*-value[b] |
|---|---|---|---|---|---|---|---|---|
| | | Homes | Subjects | Samples | ND | <LOD[a] | ⩾LOD | |
| Atrazine | | | | | | | | |
| Father | Non-farm | 23 | 23 | 89 | 34 (38%) | 39 (44%) | 16 (18%) | 0.0153 |
| | Farm | 24 | 24 | 92 | 4 (4%) | 47 (51%) | 41 (45%) | |
| Mother | Non-farm | 24 | 24 | 93 | 36 (39%) | 43 (46%) | 14 (15%) | 0.0601 |
| | Farm | 24 | 24 | 94 | 7 (7%) | 59 (63%) | 28 (30%) | |
| Child | Non-farm | 25 | 51 | 182 | 59 (32%) | 101 (55%) | 22 (12%) | 0.0355 |
| | Farm | 25 | 65 | 235 | 18 (8%) | 157 (67%) | 60 (26%) | |
| Chlorpyrifos | | | | | | | | |
| Father | Non-farm | 23 | 23 | 89 | 0 (0%) | 5 (6%) | 84 (94%) | —[c] |
| | Farm | 24 | 24 | 92 | 0 (0%) | 0 (0%) | 92 (100%) | |
| Mother | Non-farm | 24 | 24 | 93 | 0 (0%) | 5 (5%) | 88 (95%) | — |
| | Farm | 24 | 24 | 94 | 0 (0%) | 0 (0%) | 94 (100%) | |
| Child | Non-farm | 25 | 51 | 182 | 0 (0%) | 0 (0%) | 182 (100%) | — |
| | Farm | 25 | 65 | 235 | 0 (0%) | 1 (<1%) | 234 (100%) | |
| Metolachlor | | | | | | | | |
| Father | Non-farm | 23 | 23 | 89 | 23 (26%) | 22 (25%) | 44 (49%) | 0.34 |
| | Farm | 24 | 24 | 92 | 8 (9%) | 28 (30%) | 56 (61%) | |
| Mother | Non-farm | 24 | 24 | 93 | 22 (24%) | 28 (30%) | 43 (46%) | 0.82 |
| | Farm | 24 | 24 | 94 | 13 (14%) | 40 (43%) | 41 (44%) | |
| Child | Non-farm | 25 | 51 | 182 | 22 (12%) | 53 (29%) | 107 (59%) | 0.65 |
| | Farm | 25 | 65 | 235 | 24 (10%) | 64 (27%) | 147 (63%) | |
| Glyphosate | | | | | | | | |
| Father | Non-farm | 23 | 23 | 89 | 5 (6%) | 25 (28%) | 59 (66%) | 0.34 |
| | Farm | 24 | 24 | 92 | 2 (2%) | 21 (23%) | 69 (75%) | |
| Mother | Non-farm | 24 | 24 | 93 | 5 (5%) | 28 (30%) | 60 (65%) | 0.79 |
| | Farm | 24 | 24 | 94 | 7 (7%) | 24 (26%) | 63 (67%) | |
| Child | Non-farm | 25 | 51 | 182 | 2 (1%) | 20 (11%) | 160 (88%) | 0.29 |
| | Farm | 25 | 65 | 235 | 7 (3%) | 37 (16%) | 191 (81%) | |

[a]The laboratory did not censor values detected below the LOD. These values may be within the error around a zero value and are not reliably quantifiable.
[b]*P*-value for comparing the proportion of samples detected above the LOD for farm subjects versus non-farm subjects obtained using the GENMOD procedure in SAS with a REPEATED effect of household ID to account for the correlated nature of the data.
[c]Tests were not conducted due to the high proportion of samples detecting chlorpyrifos.

(Abraxis LLC, Hatboro, PA) for 10 min at room temperature; 50 µl of the derivitized mixture was analyzed by FMCIA. All participant urine samples were diluted 1:10 in a mixture of assay buffer and UriSub (1:3). An aliquot of 250 µl of the mixture was treated with derivitizing agent for 10 min and 50 µl of the derivitized sample analyzed by FMCIA. There was no measurable cross-reactivity between the pesticides allowing simultaneous measurement.

### *Data analysis*

Statistical analyses were performed using SAS 9 Software® (SAS Institute Inc., Cary, NC). Methods needed to address two concerns: first, since participants from each household provided evening and morning urine samples at two visits and multiple children were sampled from each household, concentrations could not be treated as independent. A second concern was that concentrations were frequently below the analytical LOD, particularly for atrazine, metolachlor and glyphosate (Table 1). The laboratory did not censor values below the LOD; rather, they were reported as non-detect, a level below the LOD, or a level greater than or equal to the LOD. Methods are commonly available for dealing with correlated data (e.g. mixed-effects regression modeling) and highly censored data (e.g. maximum likelihood estimation); however, methods are not readily available for simultaneously dealing with these problems.

Initially, maximum likelihood estimation, shown to work well even in the presence of high censoring rates (Helsel, 2005), was used to estimate geometric means separately for farm and non-farm family members via the LIFEREG procedure in SAS. In this analysis, urinary concentrations reported below the LOD were considered to be left-censored at the LOD

and the log-normal distribution was specified as the underlying distribution. The procedure does not work well when there are fewer than 50 detected values; consequently, estimates should be considered less reliable for atrazine, which had the fewest number of samples detected above the LOD. Since standard errors were known to be underestimated by the LIFEREG procedure, which assumes independence, it was not used for significance testing.

Mixed-effects modeling via the MIXED procedure in SAS was used to test for associations between the concentrations and covariates, estimate variance components, and estimate correlation coefficients between visit 1 and visit 2 for each family member and among the family members within each visit. In the mixed-effects models, concentrations below the LOD were used if reported and concentrations reported as non-detect were replaced with one-half of the minimum reported positive level. Urinary concentrations were skewed to the right; therefore, concentrations were natural log transformed prior to the analysis. A majority of the participants provided both an evening and a morning void; however, there were instances where only a single void (evening or morning) was provided at a particular visit (7 out of 94 father-visits, 3 out of 95 mother-visits and 19 out of 218 child-visits). To simplify the covariance structures, evening and morning voids, which were not significantly different, were averaged to give a single result for each visit.

All mixed-effects models assumed that the households were independent. Data models utilized a compound symmetric covariance structure. For children, data models utilized a compound symmetric covariance structure for children from the same household within a particular visit. That is, parameters were estimated for the variance of the levels and the covariance between levels obtained at the same visit but from different children. Parameters were also estimated for the covariance between levels obtained from the same child at different visits and from different children at different visits. Covariance parameters for farm and non-farm subjects were allowed to vary. The model with the lowest Akaike's Information Criterion (AIC) was deemed to best fit the data. Estimates of the variance and covariance parameters were used to estimate inter- and intra-individual variability (Kromhout and Heederik, 2005). In turn, these estimates were used to estimate the attenuation ratio expected when assessing associations with exposure based on two repeated observations (Liu *et al*., 1978).

The pesticide concentration in urine (μg/l), log transformed but unadjusted for creatinine, was the dependent variable for all models; creatinine adjustment was accomplished by including the creatinine level (mg/dl) as an independent variable in the model (Barr *et al*., 2005). In the mixed-effects models, since the dependent variable was the mean of the evening and morning pesticide concentrations, adjustment for creatinine was accomplished by including the mean of the evening and morning creatinine levels as an independent variable in the model. When modeling pesticide levels in urine from children, the age and sex of the child were considered potential confounders. Covariates of interest included household type (farm, non-farm), pesticide application prior to the visit and the concentration of pesticide in dust. Dust sample results have been reported previously (Curwin *et al*., 2005a). In order to have sufficient amounts of collected dust for analysis, dust samples from some households were tested for atrazine, chlorpyrifos and metolachlor (20 farm and 19 non-farm) while dust samples from the remaining households were tested for glyphosate (5 farm and 6 non-farm). Consequently, in analyses involving pesticide levels in dust, the sample size was reduced accordingly. A summary measure of the amount of pesticide in household dust was obtained by averaging the natural log transformed dust concentrations over all of the rooms tested. Farm size, amount of pesticide applied, number of acres applied and the number of days since the pesticide was last applied were considered in models of pesticide levels in urine from farm subjects. When modeling pesticide levels in urine from farm children, additional covariates included indicator variables for playing in crop fields, participation in farm chores, contact with treated fields and handling or applying pesticides. Results are presented as adjusted geometric means for comparative purposes. The significance level was set at 5%.

## RESULTS

The number of children per household and their age distributions were similar for farm and non-farm households. Among farm children, 12% (8 out of 66) reported playing in crop fields, 47% (31 out of 66) reported completing farm chores, 8% (5 out of 66) reported working in treated fields and 8% (5 out of 66) reported handling or applying pesticides. None of the 52 non-farm children in the study reported working in treated fields or handling or applying pesticides, but one non-farm child and two non-farm children reported playing in crop fields and completing farm chores, respectively.

### *Urine samples*

A majority of the urine voids detected the metabolites of chlorpyrifos, metolachlor and glyphosate above the LOD (Table 1). For atrazine, only ∼23% of the voids were detected above the LOD; however, when values below the LOD reported by the laboratory were considered, nearly 80% of the voids had an analytical level. Creatinine concentrations ranged

**Table 2** Urinary pesticide metabolite concentration, by household type

| Pesticide subject | Household type | Range[a] (μg/l) | ML estimate[b] GM (μg/l) | Mixed-effect model estimate[c] | | |
|---|---|---|---|---|---|---|
| | | | | GM (μg/l) | 95% CI | *P*-value[d] |
| Atrazine | | | | | | |
| Father | Non-farm | 0.00062–3.8 | 0.46 | 0.067 | 0.021–0.21 | <0.0001 |
| | Farm | 0.046–68 | 0.84 | 1.1 | 0.60–2.0 | |
| Mother | Non-farm | 0.0013–2.8 | 0.42 | 0.031 | 0.010–0.096 | <0.0001 |
| | Farm | 0.024–4.9 | 0.75 | 0.65 | 0.41–1.0 | |
| Child | Non-farm | 0.0028–2.2 | 0.46 | 0.054 | 0.020–0.15 | <0.0001 |
| | Farm | 0.037–3.6 | 0.71 | 0.6 | 0.38–0.93 | |
| Chlorpyrifos | | | | | | |
| Father | Non-farm | 3.8–47 | 12 | 13 | 11–15 | 0.018 |
| | Farm | 6.5–58 | 17 | 17 | 15–20 | |
| Mother | Non-farm | 1.8–35 | 11 | 11 | 9.6–14 | 0.052 |
| | Farm | 5.6–52 | 14 | 14 | 12–17 | |
| Child | Non-farm | 5.4–54 | 16 | 15 | 13–18 | 0.27 |
| | Farm | 6.1–87 | 16 | 17 | 15–19 | |
| Metolachlor | | | | | | |
| Father | Non-farm | 0.012–1.4 | 0.32 | 0.17 | 0.095–0.30 | 0.087 |
| | Farm | 0.0075–170 | 0.46 | 0.41 | 0.17–0.98 | |
| Mother | Non-farm | 0.0075–2.6 | 0.28 | 0.17 | 0.090–0.34 | 0.68 |
| | Farm | 0.010–9.7 | 0.24 | 0.21 | 0.11–0.41 | |
| Child | Non-farm | 0.010–4.2 | 0.4 | 0.24 | 0.14–0.40 | 0.17 |
| | Farm | 0.0075–64 | 0.45 | 0.39 | 0.24–0.65 | |
| Glyphosate | | | | | | |
| Father | Non-farm | 0.13–5.4 | 1.4 | 1.5 | 1.2–2.0 | 0.74 |
| | Farm | 0.020–18 | 1.9 | 1.6 | 1.1–2.4 | |
| Mother | Non-farm | 0.062–5.0 | 1.2 | 1.2 | 0.91–1.6 | 0.73 |
| | Farm | 0.10–11 | 1.5 | 1.1 | 0.71–1.8 | |
| Child | Non-farm | 0.10–9.4 | 2.7 | 2.5 | 2.1–3.1 | 0.082 |
| | Farm | 0.022–18 | 2 | 1.9 | 1.3–2.5 | |

[a]Range excludes values reported as non-detect.
[b]Geometric mean (GM) estimated using maximum likelihood methods via the LIFEREG procedure in SAS. Values below the limit of detection were left-censored at the limit of detection and the log-normal distribution was specified as a model option. Estimates for fathers and mothers were adjusted for urinary creatinine. Estimates for children were adjusted for age, sex and urinary creatinine.
[c]Geometric mean (GM) estimated using mixed-effects modeling via the MIXED procedure in SAS. Values below the laboratory limit of detection were used if reported and non-detects were replaced with one-half the minimum reported level. Values were natural log transformed prior to modeling. Estimates for fathers and mothers were adjusted for urinary creatinine. Estimates for children were adjusted for age, sex and urinary creatinine.
[d]*P*-value is for farm geometric mean versus non-farm geometric mean based on the mixed-effects model.

from 18.7–418 mg $dl^{-1}$ (median 95 mg $dl^{-1}$) and a majority of the concentrations were in the 30–300 mg $dl^{-1}$ range (733 of 785, or 93%). Urinary pesticide results based on a gas chromatograph (GC) method of analysis have already been reported for fathers (Curwin *et al.*, 2005b). Here we present analyses for fathers, mothers and children based on an immunoassay method of analysis.

Estimated urinary metabolite geometric means (GM) based on the maximum likelihood estimation method and adjusted for urinary creatinine are presented in Table 2 for fathers, mothers and children stratified by household type. Estimated GM levels based on the mixed-effects model and adjusted for urinary creatinine are also provided in Table 2. Estimates for chlorpyrifos are similar for the two methods, which was expected since nearly all samples detected chlorpyrifos above the LOD. For the remaining pesticides, both estimates require cautious interpretation due to high levels of censoring, particularly for atrazine. Based on the mixed-effects models, adjusted GM levels of the metabolite of atrazine were significantly higher in fathers, mothers and children from farm households compared with non-farm households ($P < 0.0001$). Metabolites of chlorpyrifos were higher in farm fathers ($P = 0.018$) and marginally higher in farm mothers ($P = 0.052$) when compared with non-farm fathers and mothers, but

metolachlor and glyphosate levels were similar between the two groups. GM levels of the metabolites of chlorpyrifos and metolachlor were not significantly different between farm and non-farm children. The GM glyphosate level for non-farm children was marginally significantly higher than the GM level for farm children.

### *Application status*

At the farm households, each pesticide was either not applied prior to the visit or if it had been applied it may have been applied by either a custom applicator or the farm father. Estimated geometric means (GM) based on the mixed-effects model and adjusted for urinary creatinine are presented in Table 3 for urinary levels of the metabolites of atrazine, chlorpyrifos, metolachlor and glyphosate for farm fathers, mothers and children stratified by application status. In most cases no application had taken place; however, when the pesticide had been applied, it was more often than not applied by the father.

Farm fathers who self-applied atrazine or metolachlor had significantly higher levels of urinary atrazine and metolachlor than fathers from farms where atrazine and metolachlor had not been applied prior to the visit (GM 2.5 versus 0.75 μg $l^{-1}$, $P = 0.023$ and GM 4.5 versus 0.31 μg $l^{-1}$, $P = 0.0041$, respectively). Chlorpyrifos and glyphosate urinary metabolite levels did not differ by application status among the farm fathers.

Urinary metabolite levels did not differ by application status among the farm mothers. Metabolites of atrazine were highest among farm children whose father applied atrazine prior to the visit (GM 0.96 μg $l^{-1}$), followed by children from farms where a custom applicator applied atrazine prior to the visit (GM 0.64 μg $l^{-1}$) and then by children from farms where atrazine was not applied prior to the visit (GM 0.34 μg $l^{-1}$). The only significant difference, however, was between children from farms where atrazine was not applied and children from farms where atrazine was applied by the father ($P = 0.026$). Metabolites of chlorpyrifos were higher among farm children whose father applied chlorpyrifos prior to the visit compared with children from farms where chlorpyrifos was not applied prior to the visit (GM 26 versus 16 μg $l^{-1}$, $P = 0.025$). Metolachlor and glyphosate urinary metabolite levels did not differ by application status among the farm children.

### *Household dust*

Pesticide levels in dust samples obtained from the households have been previously described (Curwin *et al*., 2005a). Here, we examined potential associations between urinary levels and levels in household dust for each pesticide. Table 4 shows the associations of pesticide urinary levels with pesticide dust concentrations and the percentage of the urinary pesticide variability that was explained by the dust concentrations. For farm fathers the pesticide level in urine was positively associated with household dust pesticide level for all pesticides except glyphosate, but was significant only for atrazine ($P = 0.01$) and

**Table 3** Urinary pesticide metabolite concentrations for farm family members, by application status

| Pesticide | Application group[a] | Farm fathers | | | Farm mothers | | | Farm children | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | *n*[b] | GM[c] | 95% CI | *n* | GM | 95% CI | *n* | GM | 95% CI |
| Atrazine | No application | 24 | 0.75 | 0.38–1.5 | 25 | 0.68 | 0.42–1.1 | 56 | 0.34 | 0.19–0.60 |
| | Custom application | 9 | 0.99 | 0.35–2.8 | 8 | 0.6 | 0.32–1.1 | 25 | 0.64 | 0.26–1.6 |
| | Father application | 15 | 2.5 [d] | 1.0–6.0 | 15 | 0.73 | 0.41–1.3 | 41 | 0.96 [d] | 0.47–2.0 |
| Chlorpyrifos | No application | 46 | 17 | 15–20 | 46 | 15 | 13–17 | 116 | 16 | 14–19 |
| | Custom application | 0 | — | — | 0 | — | — | 0 | — | — |
| | Father application | 2 | 21 | 13–35 | 2 | 16 | 8.8–30 | 6 | 26[d] | 17–39 |
| Metolachlor | No application | 40 | 0.31 | 0.14–0.66 | 41 | 0.18 | 0.096–0.34 | 102 | 0.33 | 0.20–0.54 |
| | Custom application | 3 | 0.43 | 0.062–3.0 | 2 | 0.3 | 0.029–3.1 | 7 | 0.8 | 0.22–2.9 |
| | Father application | 5 | 4.5[e] | 0.79–26 | 5 | 0.76 | 0.15–3.7 | 13 | 0.79 | 0.26–2.4 |
| Glyphosate | No application | 27 | 1.5 | 0.97–2.3 | 27 | 1.3 | 0.76–2.3 | 70 | 1.9 | 1.3–2.7 |
| | Custom application | 10 | 1.9 | 1.1–3.3 | 10 | 0.82 | 0.37–1.8 | 23 | 1.3 | 0.79–2.1 |
| | Father application | 11 | 2 | 1.1–3.5 | 11 | 1.1 | 0.47–2.6 | 29 | 2.1 | 1.3–3.5 |

[a]Application group indicates whether the pesticide was not applied, custom applied or applied by the farm father prior to the visit.
[b]*n* is the number of subject-visits.
[c]Geometric mean (GM, μg/l) and confidence interval (CI) estimated using mixed-effects modeling via the MIXED procedure in SAS. Values below the laboratory limit of detection were used if reported and non-detects were replaced with one-half the minimum reported level. Values were natural log transformed prior to modeling. Estimates for fathers and mothers were adjusted for urinary creatinine. Estimates for children were adjusted for age, sex and urinary creatinine.
[d]Significantly greater than the 'No application' geometric mean ($P < 0.05$).
[e]Significantly greater than the 'No application' geometric mean ($P < 0.01$).

**Table 4** Estimates of total variance for models with and without dust concentration, the percentage of variance explained by dust, and slope estimates, by family member and household type[a]

| Pesticide subject | Household type | *n* | $\hat{\sigma}^2_{\text{without dust}}$ | $\hat{\sigma}^2_{\text{with dust}}$ | % | $\hat{\beta}$ | *P*-value |
|---|---|---|---|---|---|---|---|
| Atrazine | | | | | | | |
| Father | Non-farm | 34 | 10.17 | 10.04 | 1.3 | 0.31 | 0.43 |
| | Farm | 40 | 1.36 | 1.11 | 18.4 | 0.22 | 0.01 |
| Mother | Non-farm | 35 | 15.93 | 16.45 | 0 | 0.01 | 0.98 |
| | Farm | 38 | 1.31 | 1.34 | 0 | −0.02 | 0.75 |
| Child | Non-farm | 79 | 10.28 | 9.26 | 9.9 | 0.64 | 0.03 |
| | Farm | 102 | 3.18 | 3.22 | 0 | 0.09 | 0.43 |
| Chlorpyrifos | | | | | | | |
| Father | Non-farm | 34 | 0.21 | 0.18 | 14.3 | 0.09 | 0.05 |
| | Farm | 40 | 0.18 | 0.15 | 18.6 | 0.1 | 0.005 |
| Mother | Non-farm | 35 | 0.28 | 0.27 | 1.5 | 0.05 | 0.31 |
| | Farm | 38 | 0.23 | 0.21 | 8.5 | 0.07 | 0.1 |
| Child | Non-farm | 79 | 0.18 | 0.16 | 11.5 | 0.06 | 0.08 |
| | Farm | 102 | 0.2 | 0.16 | 19.2 | 0.09 | 0.004 |
| Metolachlor | | | | | | | |
| Father | Non-farm | 34 | 2.53 | 2.33 | 8 | 0.3 | 0.03 |
| | Farm | 40 | 4.96 | 4.33 | 12.7 | 0.27 | 0.18 |
| Mother | Non-farm | 35 | 3.37 | 2.76 | 18.1 | 0.41 | 0.009 |
| | Farm | 38 | 2.97 | 2.8 | 5.8 | 0.19 | 0.26 |
| Child | Non-farm | 79 | 2.06 | 1.76 | 14.6 | 0.29 | 0.008 |
| | Farm | 102 | 2.57 | 2.48 | 3.3 | 0.17 | 0.13 |
| Glyphosate | | | | | | | |
| Father | Non-farm | 12 | 0.9 | 0.41 | 54 | 0.21 | 0.01 |
| | Farm | 8 | 2.42 | 3.02 | 0 | −0.28 | 0.79 |
| Mother | Non-farm | 12 | 0.73 | 0.77 | 0 | 0.02 | 0.88 |
| | Farm | 10 | 1.09 | 1.2 | 0 | 0.24 | 0.61 |
| Child | Non-farm | 17 | 1.06 | 1.11 | 0 | 0.04 | 0.76 |
| | Farm | 20 | 0.57 | 0.67 | 0 | 0.14 | 0.68 |

n is the number of observations used in the model; $\hat{\sigma}^2_{\text{without dust}}$ is the estimated total variance without dust as a fixed effect in the model; $\hat{\sigma}^2_{\text{with dust}}$ is the estimated total variance with dust as a fixed effect in the model; % is the percent of the urinary pesticide variance accounted for by dust in the model; $\hat{\beta}$ is the estimated coefficient (i.e., slope) of the relationship between the natural log transformed urinary concentration and the dust concentration, after adjusting for other fixed effects in the model; and P-value is for the association between the urinary concentrations and the dust concentrations.

[a]The associations between urinary pesticide levels and pesticide levels in household dust were obtained using the MIXED procedure in SAS to model the natural log transformed urinary pesticide level. The fixed effects in the model included urinary creatinine for fathers and mothers, and age, sex and urinary creatinine for children. Random effects included home and, for models of children's concentration, child within home. Models specified a compound symmetric covariance structure.

chlorpyrifos ($P = 0.005$). Urinary pesticide levels for non-farm fathers were positively associated with household dust pesticide levels for all pesticides but was significant only for chlorpyrifos ($P = 0.05$), metolachlor ($P = 0.03$) and glyphosate ($P = 0.01$).

For farm mothers all the pesticide urinary levels except atrazine were positively associated with household dust concentrations; however, none of the associations was statistically significant. The associations among non-farm mothers were positive for all pesticides, but only significantly for metolachlor ($P = 0.009$).

For farm children the pesticide levels in urine were positively associated with household dust pesticide level for all pesticides, but only significantly for chlorpyrifos ($P = 0.004$). For non-farm children the associations were positive for all pesticides, with atrazine ($P = 0.03$), and metolachlor ($P = 0.008$) being statistically significant.

It should be noted that the numbers of observations used in the models were relatively low for glyphosate. This was because glyphosate was only analyzed in the dust samples collected from five farm and six non-farm households. As a result the quality of these models was considered poor and results should be interpreted with caution.

### *Additional covariates*

Among farm fathers and mothers, urinary pesticide levels were not associated with farm size, number of acres applied, amount of pesticide applied or the number of days since the pesticide was last applied, with

**Table 5** Estimated correlations for urinary pesticide concentration among family members at the same visit[a]

| Pesticide | Non-farm households | | | | Farm households | | | |
|---|---|---|---|---|---|---|---|---|
| Atrazine | | Child | Father | Mother | | Child | Father | Mother |
| | Child | 1 | 0.54 | 0.55 | Child | 1 | 0.36 | 0.28 |
| | Father | | 1 | 0.70 | Father | | 1 | 0.43 |
| | Mother | | | 1 | Mother | | | 1 |
| Chlorpyrifos | | Child | Father | Mother | | Child | Father | Mother |
| | Child | 1 | 0.25 | 0.25 | Child | 1 | 0.62 | 0.54 |
| | Father | | 1 | 0.62 | Father | | 1 | 0.61 |
| | Mother | | | 1 | Mother | | | 1 |
| Metolachlor | | Child | Father | Mother | | Child | Father | Mother |
| | Child | 1 | 0.56 | 0.68 | Child | 1 | 0.63 | 0.54 |
| | Father | | 1 | 0.55 | Father | | 1 | 0.66 |
| | Mother | | | 1 | Mother | | | 1 |
| Glyphosate | | Child | Father | Mother | | Child | Father | Mother |
| | Child | 1 | 0.34 | 0.27 | Child | 1 | 0.62 | 0.55 |
| | Father | | 1 | 0.37 | Father | | 1 | 0.59 |
| | Mother | | | 1 | Mother | | | 1 |

[a]Correlation coefficients estimated using the MIXED procedure in SAS to model the natural log transformed urinary pesticide level. Fixed effects included group (farm, non-farm) and urinary creatinine. The model specified an unstructured covariance structure within the visit and a constant covariance between visits and fit separate parameters for farm and non-farm households. To simplify the calculations, child values for all the children in a household were averaged within the visit prior to computing the correlation estimates.

the exception of a marginally significant positive association observed between the level of atrazine in urine obtained from fathers and farm size ($P = 0.084$). It was difficult to assess these associations for chlorpyrifos, which was only applied to crops prior to two visits.

Among farm children, after adjusting for age, sex and urinary creatinine, urinary pesticide levels were not associated with farm size, number of acres applied, amount of pesticide applied, number of days since the pesticide was last applied, playing in crop fields, doing farm chores, working in treated fields, or handling or applying pesticides. Children's urinary concentrations were negatively associated with age for all pesticides after adjusting for creatinine excretion; however, none of the associations was significant.

### *Correlations*

Estimated correlation coefficients among the family members at the same visit are presented in Table 5. In general, for most of the pesticides, the urinary metabolite levels were fairly correlated among the family members. In the non-farm homes, higher correlations for urinary pesticide metabolite levels were generally observed between fathers and mothers than between children and fathers or between children and mothers; however, for metolachlor the highest correlation was between children and mothers. In the farm homes, the father's urinary metabolite levels were fairly correlated with both the child's and mother's urinary levels.

### *Variance components*

Estimated variance components, correlation coefficients, inter- and intra-individual variability and attenuation ratios for associations with urinary pesticide levels within farm family members are presented in Table 6. The within-subject (intra-individual) variability was more often higher than the between-subject (inter-individual) variability. However, for all the pesticides except atrazine, the father's urinary concentrations were more correlated, had less intra-individual variability compared to inter-individual variability and therefore had less exposure-response attenuation than the other family members. Conversely, the children's urinary pesticide levels were generally less correlated, had relatively higher intra-individual variability and greater attenuation.

## DISCUSSION

Farm family members generally had higher urinary pesticide levels for atrazine, metolachlor and chlorpyrifos than non-farm family members, but not higher levels of glyphosate. Among the children, only atrazine was significantly higher and glyphosate levels were actually higher among the non-farm children. Glyphosate is used agriculturally and residentially, which may explain why non-farm families had similar exposures to farm families. It is possible that glyphosate could have been applied residentially to the non-farm homes. Chlorpyrifos historically was used

**Table 6** Estimated variance components for farm family members

| Pesticide subject | Estimated variance components[a] | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | $\hat{\sigma}^2_{bh}$ | $\hat{\sigma}^2_{bs}$ | $\hat{\sigma}^2_{ws}$ | $\hat{\rho}_{c1,c2}$ | $\hat{\rho}_{v1,v2}$ | ${}_b\hat{R}_{95}$ | ${}_w\hat{R}_{95}$ | $\hat{\lambda}$ | $AR_2$ |
| Atrazine | | | | | | | | | |
| Father | — | 0.88 | 1.54 | — | 0.36 | 39.5 | 130.5 | 3.30 | 0.38 |
| Mother | — | 1.03 | 0.25 | — | 0.80 | 52.9 | 7.2 | 0.14 | 0.94 |
| Child | 0.25 | 0 | 2.79 | 0.08 | 0.08 | 1.0 | 698.0 | 698.0 | 0.003 |
| Chlorpyrifos | | | | | | | | | |
| Father | — | 0.10 | 0.07 | — | 0.59 | 3.5 | 2.9 | 0.82 | 0.71 |
| Mother | — | 0.07 | 0.13 | — | 0.35 | 2.8 | 4.1 | 1.44 | 0.58 |
| Child | 0.09 | 0.01 | 0.09 | 0.47 | 0.54 | 1.6 | 3.2 | 2.00 | 0.50 |
| Metolachlor | | | | | | | | | |
| Father | — | 2.18 | 1.58 | — | 0.58 | 325.7 | 136.9 | 0.42 | 0.83 |
| Mother | — | 1.19 | 1.69 | — | 0.41 | 72.1 | 162.8 | 2.26 | 0.47 |
| Child | 0.75 | 0.35 | 1.53 | 0.29 | 0.42 | 10.0 | 129.0 | 12.85 | 0.13 |
| Glyphosate | | | | | | | | | |
| Father | — | 0.67 | 0.27 | — | 0.71 | 24.7 | 7.6 | 0.31 | 0.87 |
| Mother | — | 0.77 | 0.92 | — | 0.46 | 31.5 | 43.1 | 1.37 | 0.59 |
| Child | 0.34 | 0 | 0.90 | 0.27 | 0.27 | 1.0 | 41.0 | 41.0 | 0.047 |

$\hat{\sigma}^2_{bh}$ is the estimated between-household variance (defined only for children); $\hat{\sigma}^2_{bs}$ is the estimated between-subject variance; $\hat{\sigma}^2_{ws}$ is the estimated within-subject variance; $\hat{\rho}_{c1,c2}$ is the estimated correlation for different children at the same visit (defined only for children), $\hat{\sigma}^2_{bh}/(\hat{\sigma}^2_{bh}+\hat{\sigma}^2_{bs}+\hat{\sigma}^2_{ws})$; $\hat{\rho}_{y1,y2}$ is the estimated correlation for the same subject at different visits, $\hat{\sigma}^2_{bs}/(\hat{\sigma}^2_{bs}+\hat{\sigma}^2_{ws})$ for fathers and mothers, $(\hat{\sigma}^2_{bh}+\hat{\sigma}^2_{bs})/(\hat{\sigma}^2_{bh}+\hat{\sigma}^2_{bs}+\hat{\sigma}^2_{ws})$ for children; ${}_b\hat{R}_{35}$ is estimated inter-individual variability, $\exp[3.92 \times \hat{\sigma}_{bs}]$; ${}_w\hat{R}_{95}$ is estimated intra-individual variability, $\exp[3.92 \times \hat{\sigma}_{ws}]$; $\hat{\lambda}$ is the ratio of the intra-individual to the inter-individual variability, ${}_w\hat{R}_{95}/{}_b\hat{R}_{95}$; and $AR_2$ is the attenuation ratio for an association based on $n = 2$ repeated measurements of exposure per individual, $\hat{\beta}/\beta = 1/(1 + \hat{\lambda}/n)$.
[a]Variance components estimated using the MIXED procedure in SAS to model the natural log transformed urinary pesticide levels among farm family members. Fixed effects included application status and urinary creatinine, and for models of children's concentrations, age and sex. Random effects included home and, for models of children's concentration, children within home. Models specified a compound symmetric covariance structure.

in residential applications but all residential uses were virtually eliminated in 2000 (USEPA, 2002b). However, other results have shown that chlorpyrifos still appeared to be ubiquitous in household environments (CDC, 2002; Fenske *et al*. 2002; Curwin *et al*. 2005a). Practically every urine sample collected in the present study had chlorpyrifos metabolite levels above the LOD. The biggest differences in urinary pesticide metabolite levels were seen among the fathers. This would be expected as the farm fathers were the principal farmer of each farm home and would therefore have had opportunity for greater pesticide exposure compared with non-farm fathers.

The atrazine data suffered from high rates of censoring at the limit of detection. The laboratory provided estimates of the concentrations below the LOD for a majority of the censored values; however, the high proportion of values below the LOD (either non-detect or positive) hindered the estimation of the geometric mean. A categorical analysis found that the proportion of atrazine levels above the LOD was higher for farm subjects compared with non-farm subjects for fathers and children ($P = 0.02$ and $P = 0.04$, respectively) and marginally higher for mothers ($P = 0.06$). Estimated geometric means for atrazine were higher for farm family members than non-family members in both the maximum likelihood and mixed-effects models; however, the differences were not as great in the latter analysis. The data suggests that there are differences between the farm and non-farm households but that the actual GM estimates, especially for the non-farm family members, are uncertain.

Estimated GMs for atrazine based on the mixed-effects model are also suspect due to the use of one-half the minimum reported value (0.0003 μg $l^{-1}$) for non-detectable values. However, regardless of the choice used to replace the non-detects, the non-farm GM would be affected more since non-farm family members had more non-detected urine samples than farm family members (~35% and 7% of the voids did not detect atrazine for non-farm and farm family members, respectively). Substituting 0.116 μg $l^{-1}$ (one-tenth the LOD for atrazine) for the non-detected values in the mixed-effects model produces similar farm GMs, but different non-farm GMs; however, the differences between farm and non-farm family members remain significant ($P < 0.0001$, $P = 0.0017$ and $P < 0.0001$ for fathers, mothers and children, respectively).

The estimates for chlorpyrifos appear to be higher than in reported literature. In the National Health and Nutrition Examination Survey (NHANES), adult males, adult females and children (aged 6–11 years)

had reported GM levels of 2.0, 1.5 and 2.8 μg $l^{-1}$, respectively (CDC, 2005). Fenske *et al*. (2002) reported mean levels of 4.9 and 4.6 μg $l^{-1}$ among children, 6 years old or younger, of agricultural workers and reference families, respectively. These values are three to seven times lower than the estimates presented in Table 2. The differences could be due to geography. NHANES is a national study, Fenske *et al*. was conducted in central Washington State while this study was conducted in eastern central Iowa State. However, the immunoassay analytical method used to measure TCP in the study may also be responsible. Duplicate urine samples from the fathers in the study were also analyzed with high performance liquid chromatography (HPLC). The TCP level in fathers' urine when analyzed with HPLC was 3.9 and 3.3 μg $l^{-1}$ for farmers and non-farmers, respectively (Curwin *et al*., 2005b), which was three to four times lower than the fathers' TCP level from the immunoassay method of analysis. However, the HPLC LOD was six times lower (0.5 ug $l^{-1}$) than that of the immunoassay method used here, which may explain the discrepancy. In describing and validating the HPLC method the authors noted that the HPLC method LOD was substantially lower than other methods (Olsen *et al*., 2004). The method paper describing the TCP immunoassay technique (MacKenzie *et al*., 2000) reported an $R^2$ correlation of 0.958 for the TCP Immunoassay with GCMS, suggesting that immunoassay is a reliable method for TCP analysis in urine.

The results suggest that a take-home pathway for pesticide exposure is possible, but are far from conclusive. Correlation coefficients for urinary metabolite levels between father and child were higher for the farm families for all the pesticides except atrazine and were higher between father and mother for farm families for metolachlor and glyphosate. Curl *et al*. (2002) in Washington State also found an association between adult and child urinary pesticide metabolite levels in families with agricultural workers.

In further support of the take-home pathway, the application of a pesticide by the father appears to influence exposure among the farm family members. Urinary atrazine and chlorpyrifos levels for farm children were significantly higher when these pesticides were applied by the father prior to the visit. Farm fathers had significantly higher atrazine and metolachlor metabolites in urine when they applied these pesticides prior to a visit. However, application of a pesticide prior to the visit did not influence the urinary metabolite levels of the farm mothers. Intuitively, one would expect the application of a pesticide prior to urine sample collection to influence the urinary metabolite levels of that pesticide. In previous work we demonstrated that the application of pesticides to crops by the farmer prior to collecting house dust samples resulted in higher levels of that pesticide in the dust (Curwin *et al*., 2005a). However, while generally there was a positive or slightly positive association between the mothers' urinary pesticide levels and pesticide dust concentrations, the association was only significant for metolachlor among non-farm mothers.

Similar to the mothers, the fathers' and children's urinary metabolite levels were also generally positively associated with dust concentrations; however, the associations were not always significant. When the associations were significant they tended to be within the non-farm families. Other sources of exposure are most likely present. In the case of the farms, family members may have other opportunities for exposure to pesticides than house dust (e.g. yard dirt), whereas within non-farm households dust may be contributing more proportionately to pesticide exposure. In contrast to our results, Curl *et al*. (2002) observed a significant positive association with azinphos-methyl concentration in house dust and urinary azinphos-methyl metabolite concentrations in children.

Several other covariates (e.g. farm size, amount of pesticide applied, playing in treated fields, and farm chores) were examined for their relationship with urinary pesticide levels but no associations were observed. This may be due in part to the large variability inherent in pesticide exposures and the small sample sizes; in some cases the covariate lacked sufficient variability to perform the analysis. The lack of an association with time since application may suggest that once pesticides have entered the home exposure may be more continuous and not dependant on a specific application event outside the home. Pesticides may persist longer in the indoor environment since they are not exposed to typical degradation products such as sun light, rain and soil bacteria. As a result, the timing of the collection of urine sample may not be critical, provided it is collected after pesticides have entered the home.

Measurement error in exposure estimation is a probable explanation for the inconsistent or lack of associations of urinary pesticide levels and environmental or behavioral factors. Kromhout and Heederik (2005) state that due to the complex pattern of agricultural exposures measurement error in agricultural exposures can be substantial and conclude that associations with exposure can go unnoticed as a result of enormous variability in exposure concentrations coupled with logistical difficulties in obtaining large numbers of measurements. In our study, the ratio between the intra- and inter-individual variability for the urine samples was often relatively large resulting in substantial attenuation in exposure associations. For example, given the variability we observed, any real association with child urinary atrazine levels would be attenuated by 99.7%, rendering it virtually impossible to detect.

Another possible explanation for the lack of associations found is that other sources of exposure may be involved. For example, dietary exposure, which may be an important pathway of exposure, was not evaluated and may account for some of the variability seen. Not only is food a source of parent pesticide exposure, but food has been demonstrated as a source of exposure for 3,5,6-trichloro-2-pyridinol, the metabolite of chlorpyrifos (Morgan *et al*., 2005). If food is a significant source of pesticide exposure in our study, then our results may be obscured making it difficult to detect any determinants of exposure or to find significant differences.

There are several limitations to the analyses. Other sources of exposure to the pesticides, such as diet and soil, were not evaluated. Chlorpyrifos was only applied on two occasions prior to a visit, so it is difficult to draw conclusions about the application effect for chlorpyrifos. In the glyphosate dust analysis, dust was collected from only five farm and six non-farm homes. Lack of variability among some of the other covariates precluded any meaningful analysis for these covariates. Statistical analyses needed to address two issues: the correlated nature of the data and the high proportion of data below the limit of detection. Unfortunately, methods are not readily available for dealing with both of these issues at the same time. We considered the use of maximum likelihood methods for estimating the geometric mean of left-censored data, but this analysis did not take into account dependencies among the repeated measures. We considered mixed-effects models, which are useful for modeling both the mean and covariance of the data, but this analysis used values reported below the LOD and substituted the minimum value divided by two for the non-detects. We presented estimates from both analyses, however, because we believe that the results are more informative than if we had merely analyzed whether or not the samples detected the pesticide. Finally, all models assumed a log-normal distribution for the data, a distribution that might not be appropriate, especially for the non-farm family members.

## CONCLUSION

In general, farm families had greater pesticide exposure than non-farm families and it appeared that the exposure may be occurring as a result of the take-home pathway; however, the results are inconclusive. The pesticide exposure varied widely and this fact coupled with the small sample sizes requires caution in interpreting the results. Often, the father's urinary pesticide metabolite levels were more correlated with their family members in farm families than in non-farm families. Further, when a farm father applied a pesticide, his children's urinary levels for that pesticide were often higher than those from farm children whose fathers did not apply the pesticide. Others have also found evidence for the take-home pathway of exposure but the significance of this source of exposure on total pesticide exposure among farm families has not been determined. Further study is needed to determine the proportion of total exposure that can be attributed to the take-home pathway.

*Acknowledgements—Disclaimer*: The findings and conclusions in this paper are those of the authors and do not necessarily represent the views of the National Institute for Occupational Safety and Health.

## REFERENCES

Alavanja MC, Hoppin JA, Kamel F. (2004a) Health effects of chronic pesticide exposure: cancer and neurotoxicity. Annu Rev Public Health; 25: 155–97.

Alavanja MC, Dosemici M, Samanic C *et al*. (2004b) Pesticides and lung cancer risk in the agricultural health study cohort. Am J Epidemiol; 160: 876–85.

ASTM. (2000) Standard practice for the collection of floor dust for chemical analysis. Standard Practice D5438–00. Philadelphia, PA: American Society for Testing and Materials.

Barr DB, Wilder LC, Caudill SP *et al*. (2005) Urinary creatinine concentrations in the U.S. population: implications for urinary biologic monitoring measurements. Environ Health Perspect; 113: 192–200.

Bearer CF. (1995) How are children different from adults? Environ Health Persp; 103: 7–12.

Biagini RE, Tolos W, Sanderson WT *et al*. (1995) Urinary Biomonitoring for Alachlor Exposure in Commercial Pesticide Applicators by Immunoassay. Bull Environ Contam Toxicol; 54: 245–250.

Biagini RE, Smith JP, Sammons DL *et al*. (2004) Development of a sensitivity enhanced multiplexed fluorescent covalent microbead immunosorbant assay (FCMIA) for the measurement of glyphosate, atrazine and metolachlor mercapturate in water and urine. Anal Bioanal Chem; 379: 368–74.

Bradman MA, Harnly ME, Draper W *et al*. (1997) Pesticide exposure to children from California's Central Valley: Results of a pilot study. J Exp Anal Environ Epidem; 7(2): 217–234.

CDC. (2002) Pesticide exposure in children living in agricultural areas along the United States-Mexico boarder, Yuma County, Arizona. Atlanta, GA: US Centers for Disease Control, National Center for Environmental Health, Health Studies Branch.

CDC. (2005) Third national report on human exposure to environmental chemicals. Atlanta, GA: U.S. Centers for Disease Control and Prevention, National Center for Environmental Health, Division of Laboratory Sciences.

Curl CL, Fenske RA, Kissel JC *et al*. (2002) Evaluation of take-home organophosphorus pesticide exposure among agricultural workers and their children. Environ Health Perspect; 110: A787–92.

Curwin BD, Sanderson WT, Reynolds SJ *et al*. (2002). Pesticide use and practices in an Iowa farm family pesticide exposure study. J Agr Safety Health; 8: 423–33.

Curwin BD, Hein MJ, Sanderson WT *et al*. (2005a) Pesticide contamination inside farm and non-farm homes. J Occup Environ Hyg; 2: 357–67.

Curwin BD, Hein MJ, Sanderson WT *et al*. (2005b) Urinary and hand wipe pesticide levels among farmers and non-farmers in Iowa. J Exp Anal Environ Epidemiol; 15: 500–8.

Daniels JL, Olshan AF, Savitz DA. (1997) Pesticides and childhood cancers. Environ Health Perspect; 105: 1068–77.
Dich J, Zahm SH, Hanberg A *et al*. (1997) Pesticides and cancer. Cancer Causes Control; 8: 420–43.
Fenske RA, Lu C, Barr D *et al*. (2002) Children's exposure to chlorpyrifos and parathion in an agricultural community in central Washington State. Environ Health Perspect; 110: 549–53.
Flower KB, Hoppin JA, Lynch CF *et al*. (2004) Cancer risk and parental pesticide application in children of Agricultural Health Study participants. Environ Health Perspect; 112: 631–5.
Garry VF. (2004) Pesticides and children. Toxicol Applied Pharmacol; 198: 152–63.
Garry VF, Harkins ME, Erickson LL *et al.* (2002) Birth defects, season of conception, and sex of children born to pesticide applicators living in the Red River Valley of Minnesota, USA. Environ Health Perspect; 110 (Suppl 3): 441–9.
Health Council of the Netherlands. (2004) Pesticides in food: assessing the risk to children. Health Council of the Netherlands, Publication No. 2004/11E. The Hague, Netherlands.
Helsel DR. (2005) More than obvious: better methods for interpreting nondetect data. Environ Sci Technol; 39: 419A–23A.
Hines CJ, Deddens JA, Striley CAF *et al.* (2003) Biological Monitoring for Selected Herbicide Biomarkers in the Urine of Exposed Custom Applicators: Application of Mixed-Effect Models. Ann Occup Hyg; 47(6): 503–517.
Kirkhorn SR, Schenker MB. (2002) Current health effects of agricultural work: respiratory disease, cancer, reproductive effects, musculoskeletal injuries, and pesticide-related illnesses. J Agric Saf Health; 8: 199–214.
Kromhout H, Heederik D. (2005) Effects of errors in the measurement of agricultural exposures. Scand J Work Environ Health; 31 (Suppl 1): 33–8.
Lewis RG, Fortune CR, Blanchard FT *et al.* (2001) Movement and deposition of two organophosphorus pesticides within a residence after interior and exterior applications. J Air Waste Manage Assoc; 51: 339–51.
Liu K, Stamler J, Dyer A. (1978) Statistical methods to assess and minimize the role of intra-individual variability in obscuring the relationship between dietary lipids and serum cholesterol. J Chronic Dis; 31: 399–418.
Loewenherz C, Fenske RA, Simcox NJ *et al*. (1997) Biological monitoring of organophosphorus pesticide exposure among children of agricultural workers in central Washington State. Environ Health Persp; 105(12): 1344–1353.
Lu C, Fenske RA, Simcox NJ, Kalman D. (2000) Pesticide exposure of children in an agricultural community: Evidence of household proximity to farmland and take-home exposure pathways. Environ Research; 84: 290–302.
Ma X, Buffler PA, Gunier RB *et al*. (2002) Critical windows of exposure to household pesticides and risk of childhood leukemia. Environ Health Perspect; 110: 955–60.
MacKenzie BA, Striley CA, Biagini RE. (2000) Improved rapid analytical method for the urinary determination of 3,5,6 trichloro-2-pyridinol, a metabolite of chlorpyrifos. Bull Environ Contam Tox; 65: 1–7.
Maroni M, Fait A. (1993) Health effects in man from long-term exposure to pesticides: a review of the 1975–1991 literature. Toxicology; 78: 1–180.
McCauley LA, Michaels S, Rothlein J *et al*. (2003) Pesticide exposure and self reported home hygiene: Practices in agricultural families. Amer Assoc Occup Health Nurses J; 51(3): 113–119.
Morgan MK, Sheldon LS, Croghan CW *et al*. (2005) Exposure of preschool children to chlorpyrifos and its degradation product 2,3,6-trichloro-2-pyridinol in their everyday environments. J Exp Anal Environ Epidemiol; 15: 297–309.
Munger R, Isacson P, Hu S *et al*. (1997) Intrauterine growth retardation in Iowa communities with herbicide-contaminated drinking water supplies. Environ Health Perspect; 105: 308–14.
National Academy of Sciences. (1993) Pesticides in the diets of infants and children. Washington, DC: National Academic Press.
NIOSH. (1995) Report to congress on workers' home contamination study conducted under the Workers' Family Protection Act (29 U.S.C. 671a). U.S. Department of Health and Human Services, Public Health Service, Centers for Disease Control and Prevention, National Institute for Occupational Safety and Health, Cincinnati, OH. DHHS (NIOSH) Publication No 95–123.
Nishioka MG, Burkholder HM, Brinkman MC *et al*. (1999). Distribution of 2,4-dichlorophenoxyacetic acid in floor dust throughout homes following homeowner and commercial lawn applications: quantitative effects of children, pets and shoes. Environ Sci Technol; 33: 1359–65.
Nishioka MG, Lewis RG, Brinkman MC *et al*. (2001) Distribution of 2,4–D in air and on surfaces inside residences after lawn applications: comparing exposure estimates from various media for young children. Environ Health Persp; 109: 1185–91.
NRDC. (1998) NRDC reports: Trouble on the farm—Growing up with pesticides in agricultural communities. New York, NY: Natural Resources Defense Council, Inc.
Olsson AO, Baker SE, Nguyen JV *et al*. (2004) A liquid chromatography tandem mass spectrometry multiresidue method for quantification of specific metabolites of organophosphorus pesticides, synthetic pyrethroids, selected herbicides and DEET in human urine. Anal Chem; 76(9): 2453–61.
Renwick AG. (1998) Toxicokinetics in infants and children in relation to the ADI- and TDI. Food Addit Contam; 15 (Suppl): 17–35.
Richter ED, Chlamtac N. (2002) Ames, pesticides and cancer revisted. J Occup Environ Health; 8: 63–72.
Rubio F, Veldhuis LJ, Clegg BS *et al*. (2003) Comparison of a direct ELISA and an HPLC method for glyphosate determinations in water. J Agr Food Chem; 51: 691–696.
SAS Institute Inc. (2004) SAS/STAT® 9.1 User's Guide. Cary, NC: SAS Institute Inc.
Simcox NJ, Fenske RA, Wolz SA *et al*. (1995) Pesticides in household dust and soil: Exposure pathways for children of agricultural families. Environ Health Persp; 103(12): 1126–1134.
Striley CAF, Biagini RE, Mastin JP *et al*. (1999) Development and validation of an ELISA for metolachlor mercapturate in urine. Analytical Chimica Acta; 399: 109
USEPA. (2002a) Pesticide industry sales and usage: 1998 and 1999 market estimates. Washington, DC: Biological and Economic Analysis Division, Office of Pesticide Programs, Office of Prevention, Pesticides and Toxic Substances, U.S. Environmental Protection Agency.
USEPA. (2002b) Interim Reregistration Eigibility Decision for Chlorpyrifos (Case No. 0100) EPA 738-R-01-007. Washington, DC: Health Effects Division, Office of Pesticide Programs, U.S. Environmental Protection Agency.
Zahm SH, Ward MH. (1998) Pesticides and childhood cancer. Environ Health Perspect; 106 (Suppl 3): 893–908.
Zahm SH, Ward MH, Blair A. (1997) Pesticides and cancer. Occup Med; 12: 269–89.