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Volume 76, No. 6 January/February 2014

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Volume 76, No. 6 January/February 2014

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Editor's Note: This special issue features *Journal* articles that were prepublished online in 2013 and made available to NEHA members through My NEHA and the online Store.

ABOUT THE COVER

This special compendium issue of the *JEH* encompasses articles from all over the world. From the effects of asbestos-containing materials after a tidal surge in Australia, to the correlation between dengue and bushfires in Brazil, to the risks associated with heavy metals in shellfish in Vietnam, this issue demonstrates that environmental health is truly an international concern. The *JEH* is pleased to present this compendium of articles with a global scope.

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ADVERTISERS INDEX

American Public University 17
CDC's National Center
for Environmental Health 85
Comark Instruments 41
Custom Data Processing.................................. 65
Decade Software 195
HealthSpace USA 196
Industrial Test Systems, Inc. 121
Inspect2GO 5
Mycometer 85
OnlineRME, LLC.. 75
Ozark River/Integrity Distribution................. 101
Shat-R-Shield.. 75
Sweeps Software, Inc. 17
Taylor Technologies, Inc. 101
Temple University 179
Underwriters Laboratories 2
University of Illinois Springfield 41

ADVANCEMENT OF THE SCIENCE

continued on page 4

Editor's Note: An installment of the *Direct From CDC* column is not included in this issue as a result of the U.S. government shutdown that occurred October 1–16, 2013, coinciding with copy deadlines for this issue. Due to space limitations in this issue, an installment of the *Demystifying the Future* column will not be included. Both columns will return in the March 2014 issue.

Furthermore, the *Journal* is pleased to announce that new columns from the Centers for Disease Control and Prevention's Environmental Public Health Tracking Branch and the Association of Environmental Health Academic Programs will be published in select 2014 issues.

Journal of Environmental Health Volume 76, No. 6 January/February 2014
continued from page 3

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Official Publication

NATIONAL ENVIRONMENTAL HEALTH ASSOCIATION

Journal of Environmental Health
(ISSN 0022-0892)

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Published monthly (except bimonthly in January/February and July/August) by the National Environmental Health Association, 720 S. Colorado Blvd., Suite 1000-N, Denver, CO 80246-1926. Phone: (303) 756-9090; Fax: (303) 691-9490; Internet: www.neha.org. E-mail: kruby@neha.org. Volume 76, Number 6. Subscription rates in U.S.: $135 per year and $250 for two years. International subscription rates: $160 per year and $300 for two years (airmail postage included). Single copies: $12, if available. Reprint and advertising rates available at www.neha.org/JEH. CPM Sales Agreement Number 40045946.

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Full text of this journal is available through Bigchalk.com at www.bigchalk.com and from ProQuest Information and Learning, (800) 521-0600, ext. 3781; (734) 973-7007; or www.umi.com/proquest. The *Journal of Environmental Health* is indexed by Current Awareness in Biological Sciences, EBSCO, and Applied Science & Technology Index. It is abstracted by Wilson Applied Science & Technology Abstracts and EMBASE/Excerpta Medica.

All technical manuscripts submitted for publication are subject to peer review. Contact the content editor for Instructions for Authors, or visit www.neha.org/JEH.

To submit a manuscript, visit http://jeh.msubmit.net. Direct all questions to Kristen Ruby, content editor, kruby@neha.org.

Periodicals postage paid at Denver, Colorado, and additional mailing offices. POSTMASTER: Send address changes to *Journal of Environmental Health*, 720 S. Colorado Blvd., Suite 1000-N, Denver, CO 80246-1926.

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PRESIDENT'S MESSAGE

Alicia Enriquez Collins, REHS

Limbo: Straddling the Present and Future

We are about to witness a tectonic shift in our profession. Within the next couple of years, the baby boomers employed in the environmental health field will be outnumbered by Generation X and Generation Y (also known as "millennials"). This realization should prompt us to examine the evolution of our profession, our mission, and how our professional association is meeting the needs of the millennials who are about to dominate the environmental health workforce.

This shift causes me to reflect upon how our personal and professional lives have been impacted by technological advances of the past and present and how our paradigms will be affected by the changes the future will surely bring. To remain current in our profession, we find ourselves having to adapt rapidly to the changes around us—something most of us are already accustomed to doing. For instance, depending on your generation, you may remember a time before

- debit cards;
- laptop and tablet computers;
- thermocouples;
- e-mail, conference calls, and video conferences;
- cellular phones, text messaging, and smart phones; and
- social media.

The changes happening in our workforce and professional associations were recently well articulated by this year's Florida Environmental Health Association (FEHA) President Robert Maglievaz in his presidential address at FEHA's annual conference. Like NEHA and many other organizations, FEHA finds its organization in a period of transition. Many of their members are retiring and the incoming generations are not replacing the outgoing members. Attracting younger professionals to join our environmental health associations and become part of our cause is our responsibility. If we do not take action to preserve our professional associations that support our field of practice and our continuing education, we are at risk of being witness to the fading of more programs and lack of support for our environmental health workforce. How will this impact our association world and our environmental health functions within our communities? The generational transition is very real and is impacting our affiliate organizations. The FEHA president's efforts to outline their current struggles and rally the membership are to be commended. I hope you will also appreciate and relate to Mr. Maglievaz's presentation excerpted below.

Attracting younger professionals to join our environmental health associations and become part of our cause is our responsibility.

Connecting the Future One Dot at a Time

by Robert Maglievaz, 2013–2014 FEHA president (Presented at the 2013 FEHA annual conference, September 2013)

Autumn is a time for reflection upon the change happening around us and there has been no shortage of it within our organization. A combination of economics, demographics, societal shifts, and technological change has forced our organization to alter the way we have always done things … and change is never easy. So we reflect upon the organization we once were and doors that have closed behind us. We lament over the things we left behind in the process and worry about what the future might bring.

Alexander Graham Bell once said, "We often look so long and so regretfully upon the closed door that we do not see the one which has opened for us." I ask our members to consider the possibility that Alexander Graham Bell may have been right … that we need to stop focusing on the closed ones and instead notice the doors opening up around us, the doors leading us to new opportunities that could make our association better, stronger, and take us to a more relevant place than we find ourselves today.

Connecting Dots

Getting to that place is not always obvious. It sometimes requires a "connecting of the dots" before the true picture of the future emerges. The problem is, as Steve Jobs once said, "You can't connect the dots looking forward; you can only connect them looking backwards. So you have to trust that the dots will somehow connect in the future." I think that was the secret to his success … when a door opened to a new opportunity, he somehow knew how to read all those dots and piece them together to form the "big picture." I think this analogy applies to the future of our association. These newly opened doors have revealed a lot of dots to us … dots that we must somehow start to connect in order to form the picture of our future. I'd like to share my perspectives and discuss the dots and how I see them connecting for our future.

A Dot in Time—January 1, 2011

One of the first dots that I see is January 1, 2011. Why is this date important? On this day, the largest turnover of human capital in American history commenced when members of the baby boom generation reached 65. Each day afterward, another 10,000 baby boomers reached retirement age. I believe it is no coincidence that our membership began dropping around this date.

Young People Just Don't Care?

Consider the statement, "Millennials only want to work eight hours whilte they look for a better opportunity." If this is true, then why are thousands of young people appearing at events such as Occupy Wall Street and how did somebody convince all these millennials to care? Clearly there must be a disconnect here, because the research says young people do care … and the ones going into the environmental health profession really do care about environmental health issues and they really do want to be involved. The research tells us the disconnect is how we expect them to plug in. We expect them to be just like baby boomers and they are not.

So the next dot connection we must figure out is how to build the plug and where to put the outlets in order for our new generation of environmental health professionals to plug into our organization and get them involved. Limited leave, tight economic times, and increased workloads also make it difficult for them to attend an annual educational conference on their own time and dollar.

According to the research, millennials really want to become part of a community that supports causes they can identify with, a community that offers connections to others within the profession, and a virtual infrastructure through which they can rapidly plug into to make these connections from the comfort of their home or office. So building an infrastructure that can support that virtual environmental health community are the next dots we must connect.

We Need a Cause

What better cause could we adopt than fighting for the respect and salaries of these new people coming into our profession and supporting them as they move through their careers? Building that respect, supporting our own, and promoting environmental health in our communities are all causes that will attract millennials to our ranks.

A Better Future

One hundred and fifty years ago, for every person who died of old age, eight more died from environmental-associated diseases. It was through the commitment and diligence of those who came before us that we now all enjoy the possibility of living to old age.

This is no time to rest on the laurels of our past. This is no time to wallow in the trial and tribulations of our current circumstances or to allow pessimism and pragmatism to keep us on the plateau of the present. I have come to you to remind you of the urgency of NOW and why an organization like FEHA is important to us all. This is the time for a new beginning, a beginning that could eventually unite the professionals in our environmental health programs and make them stronger and more effective. I challenge you to rediscover the spirit of urgency and purpose that was present when the environmental health profession was created. Now is the time for us to make the commitment to be involved and take the journey together.

Moving Toward the Future

The challenge is identifying and connecting with younger professionals using different approaches and technologies. NEHA is addressing this challenge by expanding our student mentorship program. We have the opportunity to hear directly from graduating environmental health students now entering the profession. NEHA has also been working diligently to incorporate practices that are more appealing and conducive to learning for our incoming workforce. Some examples of recent changes within NEHA include 1) e-Learning opportunities online, 2) learning laboratory formats for our conferences, 3) virtual conferences where participants can tune in remotely either live or at a later time, 4) E-News publications one to two times per month, 5) release of an E-*Journal* last month, 6) e-mail blasts for quick dissemination of information, 7) development of an electronic communication tool for information exchange between affiliate presidents, 8) e-commerce, 9) creation of environmental health blogs, and 10) community outreach efforts at annual conferences.

At our upcoming 2014 Annual Educational Conference & Exhibition in Las Vegas, Nevada, our meeting with affiliate presidents will focus on the technological and generational transition. We will strive to do more, are open to your ideas, and will engage in further exchange with our affiliates. Please feel free to contact me by e-mail or contact your NEHA regional vice president (see listing on page 186) if you have any thoughts on taking proactive measures during this time of rapid transition.

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Asthma Prevalence and Risk Factor Assessment of an Underserved and Primarily Latino Child Population in Colorado

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Abstract **Asthma is a substantial public health burden among children. Disease and risk-factor discrepancies have been identified among racial, ethnic, and socioeconomic groups. At a rural health clinic (Salud Family Health Center) with primarily underserved and Latino patients in Colorado, the authors evaluated 250 medical records and administered 57 parental surveys to describe this population with respect to asthma diagnosis, asthma-like symptoms, and environmental/occupational risk factors among children. Wheeze and asthma were indicated in 9.7% and 8.9% of medical records, respectively. Twenty parents (35.7%) reported in a questionnaire that their child had experienced wheezing or whistling in the chest. Parents reported that children play in farming fields (21.8%) and feed livestock/animals (10.9%). Additionally, 13.2% and 9.4% of children have a household member who works around livestock or around grain, feed, or dust, respectively. Information from the Salud population can be used to develop larger-scale research and public health initiatives to eliminate health and risk factor disparities among underserved children.**

Introduction

In the U.S., the percentage of children who are Hispanic has grown faster than any other racial or ethnic group (Federal Interagency Forum on Child and Family Statistics, 2009). In 2008, 22% of U.S. children were Hispanic, an increase from 9% in 1980 (Federal Interagency Forum on Child and Family Statistics, 2009). In 2004, Latinos represented 19.2% of the Colorado population (U.S. Census Bureau, 2004a), increasing from the 17.1% identified in the 2000 Census (U.S. Census Bureau, 2004b).

While Latinos represent the largest demographic group among U.S. children, the group is not ethnically homogeneous (Choudhry et al., 2007). Genetic, socioeconomic, educational, and demographic variation within and among Latino ethnic groups provides a unique opportunity to examine the impacts of race, genetics, culture, and environment on complex diseases, such as asthma (Choudhry et al., 2007). Asthma, a chronic inflammatory disorder of the airways characterized by episodic and reversible airflow obstruction and airway hyperresponsiveness, is experienced disproportionately among certain racial, ethnic, and socioeconomic groups. For example, asthma prevalence was 120% higher in Puerto Rican children, 60% higher in African-American children, and 25% higher in Native American children as compared to non-Latino white children (Gold & Wright, 2005). Additionally, those below the federal poverty level had higher asthma rates (10.3%) compared with those at or above the poverty level (6.4% to 7.9%) (Moorman et al., 2007).

Analysis of national data reveals that Latino asthma prevalence has increased over time (Flores & Committee on Pediatric Research, 2010). Other asthma-related disparities experienced disproportionately by Latinos include emergency department and urgent care visits; hospitalizations; activity limitations; a higher potential asthma burden (diagnosed plus undiagnosed asthma); lower use of asthma medications; and among Mexican-Americans, higher odds of cockroach and dust mite allergen sensitivity (Flores & Committee on Pediatric Research, 2010). The asthma discrepancies among different Latino ethnic groups, as well as the lack of research conducted among these populations, has led the American Academy of Pediatrics to identify asthma among Latinos as an urgent priority for future research (Choudhry et al., 2007).

Challenges remain in understanding chronic health conditions potentially associated with disparities involving occupation, environment, economics, education,

TABLE 1

Height and Weight by Age Category as Recorded on the Well-Child Checkup Forms in the Salud Family Health Center Medical Charts

Category[a]	#	Mean	Standard Deviation	Minimum	Median	Maximum
3-year well-child checkup						
Height	118	40.5	2.3	35.0	40.8	46.0
Weight	119	36.7	5.3	25.0	36.8	51.0
5-year well-child checkup						
Height	90	44.9	2.8	36.0	45.0	51.5
Weight	95	46.7	9.5	31.0	45.0	82.0
7-year well-child checkup						
Height	51	51.1	4.0	44.1	51.0	60.0
Weight	52	67.5	21.3	42.0	63.0	141.0
11-year well-child checkup						
Height	21	58.2	5.2	47.5	58.0	65.5
Weight	24	99.8	28.2	50.0	99.0	156.0

Note. Same child could be represented in multiple well-child checkups.
[a]Height in inches and weight in pounds.

culture, language, and immigration status (Flores et al., 2002). In the U.S., farm workers continue to be one of the most impoverished and underserved populations (Heuer, Hess, & Batson, 2006) and greater attention has recently been focused on agricultural-related illness among children living and working on farms (Park et al., 2003). The majority of jobs in rural areas of Colorado are agricultural. Eliminating disparities will require the development of longitudinal epidemiologic research that can be translated into culturally appropriate public health initiatives.

For our pilot study, we partnered with the Salud Family Health Center (Salud), an organization providing health care to underserved and primarily Latino populations (approximately 64% of Salud clients are Latino), in order to evaluate two different methods of data acquisition. Using medical record abstraction, we characterized the Salud child population with respect to asthma and risk factors thought to adversely affect Latino populations. In addition, we assessed, via parental questionnaire, the prevalence of environmental and occupational risk factors also disproportionately affecting Latino populations in this area of Colorado. Latinos are frequently not included in child health research (Flores et al., 2002). Therefore it is critical to identify research questions for future larger-scale research initiatives aimed at evaluating potential environmental and occupational risk factors (either directly to the children themselves or indirectly via exposures from the parents) and eliminating health disparities among underserved children in Colorado.

Methods and Data Analysis

Study Population

The study population consisted of child patients at the Salud clinic in Fort Lupton, Colorado. Fort Lupton has an estimated population of 7,620 and is located in a rural area of Colorado, approximately 20–30 miles northeast of Denver. About 6% of Salud clients have private insurance, 4% qualify for Medicare, and 25% receive Medicaid; the remaining 65% have no significant third-payer source. Salud serves all community members; however, low-income medically underserved populations and migrant/seasonal farmworker populations are the priority clientele. In addition, Salud does not refuse patients based on insurance coverage or ability to pay.

A random sample of 250 records for children aged 5–12 years was created from a database of billing records from September 2007 through August 2008 in order to perform the medical record abstraction (described below). A total of 7,013 visits (2,671 unique patients) occurred during this time period for children aged 0–19 years. There were 2,099 total visits (984 unique patients) among children aged 5–12 years. In addition to the medical record abstraction, a convenience sample of 57 parents of children visiting the clinic from January through May 2009 was identified to participate in a respiratory health and environmental exposures questionnaire (described below). All study procedures were approved by Colorado State University institutional review board and by Salud.

Medical Record Abstraction

Medical records were reviewed in order to ascertain the types of data available to researchers for larger scale studies in this population. Based on this review, a database for the medical record abstraction, designed to describe the population in terms of demographics, anthropometrics, and health was created. Additional surveys were located within a subset of the medical records and included information on potential indicators of environmental or occupational-related exposures. Records included files from the entirety of the patient's tenure at Salud; therefore, visits occurring during younger ages were also assessed. Variables abstracted included address, work in agricultural fields (child and/or family), occupancy in public housing, migrant status, ethnicity, family size, primary language, annual family income, country of birth, sex, date of birth, insurance provider, term birth status, presence of an asthma diagnosis, any indication of a wheezing symptom, diabetes, environmental tobacco smoke, height/weight (from routine well-child reports), and dates for diseases/symptoms. A random sample of 250 charts among the 5–12 year age group was abstracted.

Questionnaire Development and Administration

The questionnaire was developed to incorporate two previously validated surveys: the

International Study of Asthma and Allergies in Childhood (Asher et al., 1995) and the Keokuk County Rural Health Study, which evaluated exposures in rural Iowa children (Merchant et al., 2005; Park et al., 2003); additional questions appropriate for this population were identified through meetings with Salud personnel. The questionnaire was designed to obtain information about the prevalence of diagnosed asthma ("Has your child ever had asthma?") and undiagnosed asthma (via symptoms assessment that included a detailed module on wheeze); prevalence of potential risk factors for childhood asthma, such as occupational and environmental exposures to pollutants (including occupation, parental occupation, rural versus urban residence, truck traffic, and questions regarding housing quality), and obesity; as well as age, gender, race, ethnicity, and education level of the mother and father. Questionnaires were adapted and translated into Spanish. Salud staff administered questionnaires in person in English or Spanish (depending on participant preference) from January 15, 2009, through May 28, 2009. All respondents provided informed consent. A limitation of these methods is that the sample of 57 parents surveyed represents a convenience sample selected based on Salud staff and patient availability. Because of the timing and logistical constraints of attempting to administer a survey between patient check-in and contact with the physician, many surveys were not fully completed.

Data Analysis

Descriptive frequencies and prevalence of asthma, adverse respiratory symptoms, and demographic, environmental, and occupational risk factors were calculated for the sample population (from medical records and questionnaire data). Analyses to compare asthma and risk factor prevalence across data source types were deemed inappropriate due to the logistical difficulties of administering the complete parental survey to a representative population. We obtained only a small number of participants with both a completed questionnaire and a reviewed chart (n = 13; among the 13 children, only one had an indication of asthma diagnosis and this was indicated on both the questionnaire and the chart review).

TABLE **2**

Demographic Frequencies as Recorded in the Salud Family Health Center Medical Charts

Characteristic	#	%
"In the past 24 months have you or a family member worked in the agricultural fields?"		
Yes	12	10.8
No	99	89.2
"Do you currently live in public housing?"		
Yes	5	8.3
No	55	91.7
"What is your current status?"		
Migrant	10	5.2
Seasonal	12	6.3
Other	169	88.5
Ethnicity		
Hispanic	152	81.3
White	35	18.7
Language		
English	61	33.2
Spanish	123	66.9
Country of birth		
U.S.	101	78.3
Mexico	28	21.7
Sex		
Male	122	49.6
Female	124	50.4

Note. Medical charts obtained from September 2007 through August 2008 (surveys were completed by the adult accompanying the child).

Results

Medical Records

Demographic characteristics collected and maintained in the charts are presented in Tables 1 and 2. The Salud clinic maintained charts for well-child checkups for infants and children aged 3, 5, 7, and 11 years. Height and weight were abstracted (Table 1). Means are not independent across age categories as the same child was likely represented in several age categories while a patient at the Salud clinic. Hispanic ethnicity was reported by 81.3% of the population (Table 2). Twelve (10.8%) parents reported having a family member who worked in the agricultural fields in the past 24 months, and migrant or seasonal status was reported by 5.2% and 6.3% of the population, respectively (Table 2). Only 11 charts included a reference to environmental tobacco smoke (seven with reported exposure and four without); therefore, the assessment of this factor via medical chart is likely not useful. Reported annual incomes ranged from \$0–\$54,852. For a family of three (n = 29), the mean income was \$14,459 (SD = \$8,240) and for a family of seven (n = 10), the mean income was \$27,978 (SD = \$9,598). Wheeze and asthma were indicated in 9.7% and 8.9% of the medical records, respectively (percentage represents the number of charts with indication of the health endpoint divided by the total number of charts abstracted). Term birth status was only identified on 60 of the abstracted charts (24.0%); therefore, it is likely not a useful variable to obtain solely via medical record. Of those charts with information on birth status, 6 were listed as preterm and 54

TABLE 3

Demographic Frequencies Ascertained via Questionnaires Administered to Parents of Children Visiting the Salud Family Health Center

Characteristic	#	%
Sex		
Male	36	65.5
Female	19	34.6
Ethnicity		
Hispanic/Latino	52	96.3
Not Hispanic/Latino	2	3.7
Race		
White	31	86.1
Native American or Alaskan Native	1	2.8
Other (did not specify)	2	5.6
Do not know	1	2.8
Decline to answer	1	2.8
Mother years of education		
Less than 9 years	3	10.3
9–12 years	16	55.2
Greater than 12 years	10	34.5
Father years of education		
Less than 9 years	5	20.0
9–12 years	13	52.0
Greater than 12 years	7	28.0

were listed as term babies. Of those charts with information on the length of gestation; 45 (75% of the 60 charts) also listed information on weight for age at gestation. The majority were identified as average-for-gestational age (40/45 charts) and five were listed as large-for-gestational age (including one that was also identified as preterm). No reference to diabetes occurred in any of the reviewed charts.

Questionnaire

Fifty-seven in-person questionnaires were administered in either English (n = 25) or Spanish (n = 32), depending on the preference of the participant. The surveys obtained information on children representing the following age ranges: 2–3 years, n = 7; 4–5 years, n = 7; 6–7 years, n = 9; 8–9 years, n = 16; 10–11 years, n = 10; and 12–13 years, n = 5. Demographic characteristics and parental education ascertained via questionnaire are presented in Table 3. Illnesses and symptoms are presented in Table 4. Twenty parents (35.7%) reported that their child has had "wheezing or whistling in the chest at any time in the past." Seven parents (12.7%) reported that their child had asthma and 14 parents (25.0%) reported their child having had "a dry cough at night, apart from a cough associated with a cold or chest infection within the last 12 months." Frequencies of selected exposures that may influence asthma risk are presented in Table 5; 79.2% and 73% of the population reported having fitted carpets in their child's bedroom currently and during the child's first year of life, respectively. When asked about the surroundings of their child's home, 30.2% of parents reported "rural, open spaces or fields nearby"; 14.0% reported "suburban, with many parks or gardens"; 44.2% reported "suburban, with few parks or gardens"; and 11.6% reported "urban, with no parks or gardens" (Table 5). Fifty-two parents answered the question, "How often do trucks pass through the street where you live, on weekdays?" with 11.5% responding "never"; 42.3% responding "seldom"; 26.9% responding "frequently through the day"; and 19.2% responding "almost the whole day" (Table 5). Fifty-two parents also answered the question, "Outside school hours, how often does your child usually exercise so much that he/she gets out of breath or sweats?" with 34.6% responding "every day"; 15.4% responding "4–6 times a week"; 21.2% responding "2–3 times a week"; 11.5% responding "once a week"; 5.8% responding "once a month"; and 11.5% responding "less than once a month." Agricultural exposures thought to be disproportionately experienced by rural dwellers are presented in Table 6: 21.8% of children play in farming fields, 16.4% eat fruits and vegetables without washing, and 10.9% feed livestock or other animals. In addition, 13.2% and 9.4% have someone in their household who works around livestock or around grain, feed, or dust, respectively.

Discussion

Reducing health disparities among population groups has become a priority among many of the leading health organizations in the U.S. In addition, Healthy People 2010 identified 10 leading health indicators, including obesity and environmental quality, as top health priorities (U.S. Department of Health and Human Services, 2000). The majority of these indicators, chosen because they are preventable threats to health, are disproportionately experienced by underserved or minority populations, such as African-American, Native American, and Latino children (Chowdhury, Balluz, Okoro, & Strine, 2006).

In the U.S., the burden of asthma falls disproportionately and increasingly on racial/ethnic minorities and poor children (Centers for Disease Control and Prevention, 2000; Halfon & Newacheck, 1993). African-American, Native American, and Latino children have higher prevalence rates for asthma as compared to non-Latino white children (Gold & Wright, 2005; Perrin, Bloom, & Gortmaker, 2007). Additionally, less use of preventive asthma medications was reported among minority children than white children within the same managed Medicaid population (Lieu et al., 2002). Previous studies suggest that financial access and use of managed care does not eliminate racial/ethnic variation in asthma-related care (Flores, Bauchner, Feinstein, & Nguyen, 1999; Krishnan et al., 2001). Disproportionate increases in childhood obesity may partly explain disparities in asthma prevalence (Rodriguez, Winkleby, Ahn, Sundquist, & Kraemer, 2002; Schwartz, Gold, Dockery, Weiss, & Speizer, 1990).

Small sample sizes limit the ability to draw strong conclusions about the Salud population in regards to asthma prevalence; however, 20 parents (35.7%) reported that their child had experienced wheezing or whistling in the chest at any time in the past and an asthma diagnosis was reported among 12.7% (*n* = 7) of the population. In addition, a "dry cough at night, apart from a cough associated with a cold or chest infection, in the last 12 months," was reported among 25.0% (*n* = 14) of the population. For comparison, among children aged 2–16 years participating in the Third National Health and Nutrition Examination Survey (NHANES III), 6.6% had self-reported current asthma and 7.4% had self-reported wheezing (wheezing or whistling in the chest at any time in the past 12 months and the child's chest sounding wheezy or whistled when the child did not have a cold) (Romieu, Mannino, Redd, & McGeehin, 2004). In 2007, the prevalence of lifetime asthma among Colorado residents 1–14 years old was 11.9%; white non-Hispanic children had a nonsignificantly higher current and lifetime prevalence (8.6% and 11.6%, respectively) than white Hispanic children (6.2% and 9.9%, respectively) (Colorado Department of Public Health & Environment, 2008). A limitation with interpreting the Salud population percentages and making comparisons to the general population is that the denominator is the population actually visiting the clinic and not necessarily the entire target population for the clinic. It is possible that children live there who do not receive health care attention who would otherwise be recommended to this clinic. If these children are healthier than those who attend the clinic, then the percentages we present may over-represent illnesses among the Salud clinic's target population.

TABLE **4**

Reported Health Symptoms Ascertained via Questionnaire Administered by Salud Family Health Center Staff to Parents With Children Visiting the Clinic

Symptom	#	%
Has your child ever had wheezing or whistling in the chest at any time in the past?		
Yes	20	35.7
No	36	64.3
Has your child had wheezing or whistling in the chest *in the last 12 months*?		
Yes	12	26.7
No	33	73.3
How many attacks of wheezing has your child had *in the last 12 months*?		
None	6	35.3
1 to 3	8	47.1
4 to 12	3	17.7
More than 12	0	0
In the last 12 months, how often, on average, has your child's sleep been disturbed due to wheezing?		
Never woken with wheezing	8	50.0
Less than one night per week	5	31.3
One or more nights per week	3	18.8
In the last 12 months, has wheezing ever been severe enough to limit your child's speech to only one or two words at a time between breaths?		
Yes	3	17.7
No	14	82.4
Has your child *ever* had asthma?		
Yes	7	12.7
No	48	87.3
In the last 12 months, has your child's chest sounded wheezy during or after exercise?		
Yes	6	11.1
No	48	88.9
In the last 12 months, has your child had a dry cough at night, apart from a cough associated with a cold or chest infection?		
Yes	14	25.0
No	42	75.0
Has your child *ever* had an itchy rash that was coming and going for at least six months?		
Yes	7	12.7
No	48	87.3

Asthma disparities may partly be explained by environmental exposures as Latino children have disproportionately greater exposures to environmental toxins, including ambient and indoor air pollutants and pesticides (Mott, 1995; Wernette & Nieves, 1992). Ambient air pollution is likely a significant factor affecting asthma among urban children and the little research performed among rural U.S. children indicates that asthma prevalence is high (16% in a rural Iowa population and 20% among farm children) (Merchant et al., 2002) and that the prevalence of children with severe symptoms consistent with asthma (but without a doctor diagnosis) may be even higher (Chrischilles et al., 2004). More research on the differential distribution of air pollution health effects on racial/ethnic minorities is needed to gain a better understanding of this issue (O'Neill et al., 2003).

TABLE **5**

Selected Responses From the Exposure Module Ascertained via Questionnaire Administered by Salud Family Health Center Staff to Parents With Children Visiting the Clinic

Exposure	# Reporting Yes	%
Does or did your child's mother smoke:		
At present? (*n* = 54)	2	3.7
During the child's first year of life? (*n* = 39)	1	2.6
During pregnancy with your child? (*n* = 39)	1	2.6
Does anybody, *at present,* smoke inside your child's home? (*n* = 55)	4	7.3
Does your child smoke? (*n* = 53)	0	0
Does or did your child's home have damp spots on the walls or ceiling?		
At present (*n* = 53)	0	0
During the child's first year of life (*n* = 42)	3	7.1
Does or did your child's home have visible molds or fungus on the walls or ceiling?		
At present (*n* = 53)	4	7.6
During the child's first year of life (*n* = 40)	3	7.5
What kind of floor covering is or was there in your child's bedroom?		
At present (*n* = 48)		
Fitted carpets	38	79.2
Loose carpets	2	4.2
Bare floors	8	16.7
During the child's first year of life (*n* = 37)		
Fitted carpets	27	73.0
Loose carpets	2	5.4
Bare floors	8	21.6
Are there any insect problems (for example, cockroaches, spiders, ants, etc.) in the child's home? (*n* = 53)	6	11.3
Are there any problems with mice or rats in the child's home? (*n* = 55)	4	7.3
In the last 12 months, have pesticides been used in or around your house or on your lawn or garden? (*n* = 54)	4	7.4
How would you describe the surroundings of your child's home at present? (*n* = 43)		
Rural, open spaces or fields nearby	13	30.2
Suburban, with many parks or gardens	6	14.0
Suburban, with few parks or gardens	19	44.2
Urban, with no parks or gardens	5	11.6
How would you describe the surroundings of your child's home *during the child's first year of life*? (*n* = 30)		
Rural, open spaces or fields nearby	7	23.3
Suburban, with many parks or gardens	4	13.3
Suburban, with few parks or gardens	12	40.0
Urban, with no parks or gardens	7	23.3
How often do trucks pass through the street where you live, on weekdays? (*n* = 52)		
Never	6	11.5
Seldom	22	42.3
Frequently through the day	14	26.9
Almost the whole day	10	19.2

Among children, traffic-related air pollution has been associated with respiratory health (Gauderman et al., 2004, 2007), including asthma (Gauderman et al., 2005; Jerrett et al., 2008); and nearly one-fifth of our study population reported truck traffic at their home as occurring "almost the whole day." Children as young as five often participate in farm chores and children in rural communities are often exposed to organic dusts, agricultural chemicals, animal allergens, and grain dust mites that are brought into the home on work clothing (Mayo, Richman, & Harris, 1990; Merchant, 1987; Park et al., 2003) and several of these types of exposures (playing in farming fields and in dirt near fields and feeding livestock/animals) were reported among the Salud population. More research describing the relationships between environmental exposures and increased asthma risk is needed as the contribution of these types of exposures to childhood disease is believed to be substantial (Flores et al., 2002).

A main limitation of the population risk factor (environmental and occupational) proportions we report is that information was collected from a convenience sample of parents with ample time between check-in and physician examination. Although we do not have reason to believe that parents surveyed would respond in a systematically different manner than those not surveyed, identifying a random sample of participants and conducting the survey at a more convenient and feasible time (e.g., after the physician examination) may allow for a more representative estimate of risks.

According to the Centers for Disease Control and Prevention (CDC), in body mass index (BMI) growth charts, the transition from healthy weight to overweight occurs around 17 kg/m^2 for children aged 3–5 years. The weight distinctions are based on national percentiles according to age and sex (above the 85th percentile is considered overweight). Using data ascertained from the three-year well-child checkups in the medical charts, the population of male children aged 3–5 years visiting the Salud clinic had an average BMI of 15.7 kg/m^2 (*SD* = 1.2 kg/m^2; *n* = 60). The transition from healthy weight to overweight occurred around the 87th percentile in this population of boys, indicating that the Salud population at the 3–5 year age range is likely similar to that of the general U.S. According to the data abstracted from the five-year well-child checkup, the average BMI for boys aged 5–7 years is 16.4 kg/m^2 (SD = 1.8 kg/m^2; n = 44). For the general U.S. population at this age group, the transition from healthy weight to overweight also occurs around 17 kg/m^2. In the Salud population, only about 70% of the population of boys has a BMI less than 17 kg/m^2. Although the small sample size is a limitation, this suggests that in the 5–7 year age group, the Salud population of boys is more overweight than the general U.S. population and may, therefore, be at increased risk for overweight and obesity-related health endpoints. In addition, half of the Salud population interviewed reported that their child exercised (defined as getting out of breath or sweating outside of school hours) 2–3 times per week or less. Among 8–16 year olds completing the physical activity questions in the NHANES III survey, 19.4% (weighted percentage adjusted to represent the U.S. population) reported vigorous physical activity less than three times per week (Dowda, Ainsworth, Addy, Saunders, & Riner, 2001). Given the overlap in categories of activity, this comparison is not direct; however, it does indicate that the Salud population of children may be less active than the general U.S. population. Only 34.6% of the Salud children exercise every day, which is the CDC-recommended level (60 minutes per day) for all children. Physical activity disparities by race and ethnicity have been reported among U.S. children (Eaton et al., 2008) and eliminating these types of disparities was also a focus of Healthy People 2010.

TABLE **6**

Child and Family Member Job and Activity Exposures Ascertained via Questionnaire Administered by Salud Family Health Center Staff to Parents With Children Visiting the Clinic

Exposure	# Reporting Yes	%
Does your child have a job outside of the house? (n = 53)	0	0
Does your child participate in any of the following activities?		
Plays in farming fields (n = 55)	12	21.8
Plays in dirt near fields (n = 55)	16	29.1
Plays in livestock buildings (n = 55)	6	10.9
Plays in grain or feed storage or handling facilities (n = 55)	3	5.5
Swims in irrigation channels (n = 55)	3	5.5
Is outside near farming fields while the fields are sprayed with pesticides (n = 55)	3	5.5
Eats fruits and vegetables without washing (n = 55)	9	16.4
Picks crops in the fields (n = 54)	1	1.9
Drives or rides in tractors to cut wheat and corn or to pick up trash (n = 55)	3	5.5
Sprays weeds and insects (n = 55)	1	1.8
Helps to feed or move cattle (n = 55)	3	5.5
Cleans livestock/animal buildings (n = 55)	3	5.5
Feeds livestock/animals (n = 55)	6	10.9
Cleans grain bins (n = 55)	3	5.5
Does someone from your household participate in any of the following jobs?		
Works around livestock (n = 53)	7	13.2
Picks crops such as lettuce, beans, or cabbage in the field (n = 53)	2	3.8
Picks crops such as wheat, corn, or potatoes in the field (n = 53)	2	3.8
Works at a sale barn (n = 53)	4	7.6
Works at a slaughterhouse (n = 53)	1	1.9
Works around grain, feed, or dust (n = 53)	5	9.4
Works with or applies pesticides (n = 53)	2	3.8
Works somewhere where pesticides are applied (n = 53)	3	5.7

Conclusion

General epidemiologic characteristics including prevalence and severity of asthma and environmental and occupational risk factors have not been extensively evaluated within the Latino populations in Colorado. Although a more analytic approach to evaluate the relationship between risk factors and adverse health was beyond the scope of this project, we have demonstrated that abstracting medical records and administering questionnaires to parents of children visiting the clinic are both feasible options for obtaining these types of data. Administering questionnaires allows for the collection of more directed responses concerning asthma-like symptoms as well as detailed exposure information as compared to information generally recorded in medical records; however, identifying a more appropriate time to administer the surveys should benefit future studies. From the results of this hypothesis-generating pilot project, we were able to demonstrate the elevated occurrence of asthma-like symptoms as well as multiple occupational and environmental risk factors in this unique population, which is critical for developing future hypothesis-testing research.

Acknowledgements: This project was supported by the National Institute for Occupational Safety and Health Mountain and Plains Education and Research Center funding mechanism to support short-term pilot research and community-based projects in occupational and environmental health. We are grateful to Elva Garcia and the staff at the Fort Lupton Salud clinic for data acquisition and logistical support. In addition, we thank the participating parents.

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continued on page 16

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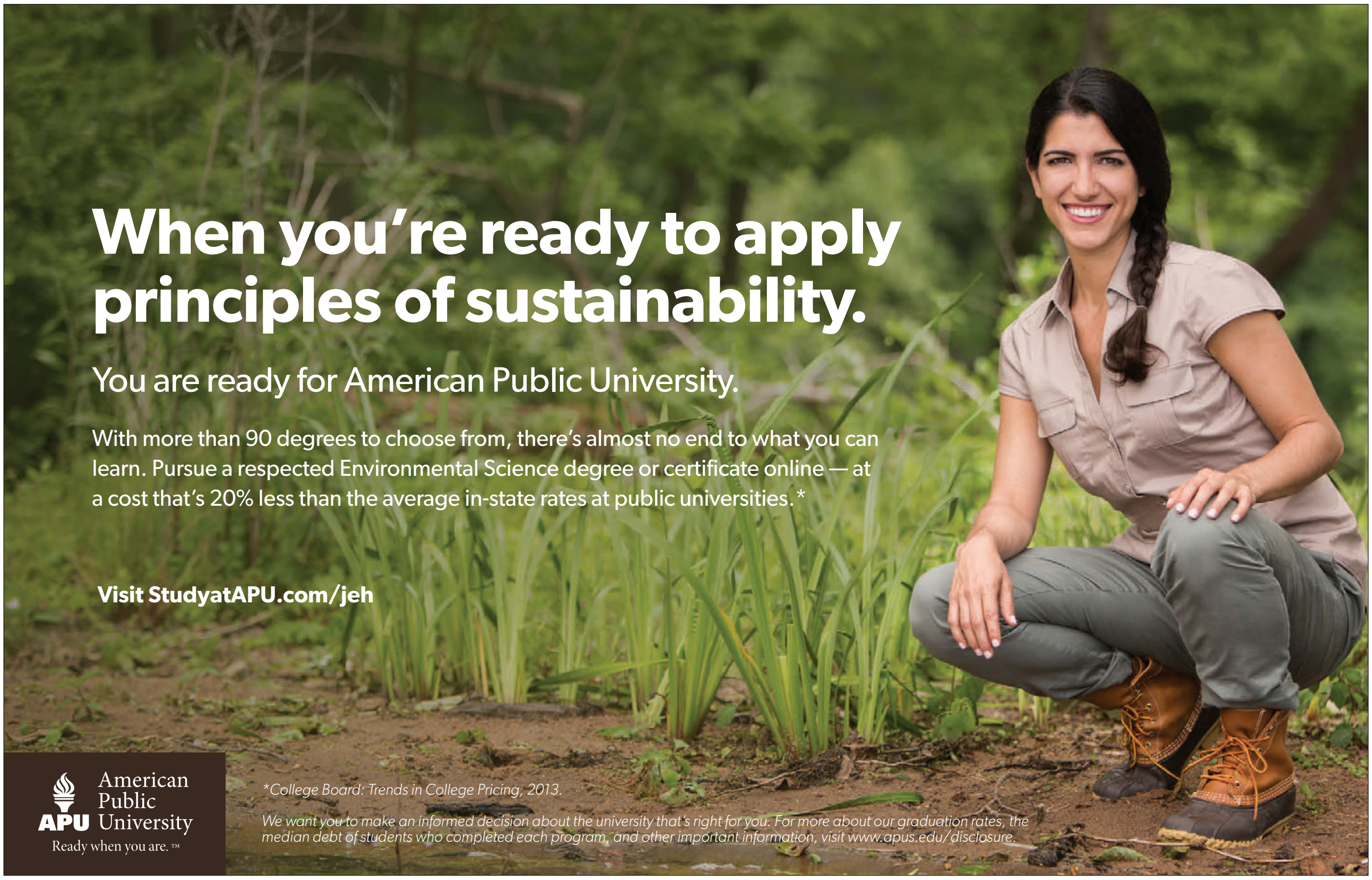
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Effects of Centralized and Onsite Wastewater Treatment on the Occurrence of Traditional and Emerging Contaminants in Streams

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Abstract **The authors conducted a survey of small streams to evaluate the effects of centralized and onsite wastewater treatment on the occurrence of selected traditional and emerging contaminants in small streams in the upper Neuse River basin, North Carolina. An undeveloped site was included to assess effects of residential land use activities on stream quality. Concentrations of nutrients and ions were higher in samples from streams in residential sites than from the stream in an undeveloped area. Overall, streams draining residential areas showed relatively small differences with respect to type of wastewater treatment. Two sites, however—one in an area of centralized wastewater treatment apparently near a suspected sewer line leak, and the second in an area of onsite wastewater treatment—showed effects of wastewater. Organic wastewater compounds were detected more frequently in samples from these two sites than from the other sites. Optical brighteners levels were correlated (r^2 = .88) with the number of organic wastewater and pharmaceutical compounds detected at the residential sites and could potentially serve as a screening method to assess wastewater effects on small streams.**

Introduction

Historically, investigations of the effects of wastewater on streams have addressed dissolved oxygen, nutrients, and pathogens. In recent years, investigations have included the effects of pharmaceuticals, hormonally active compounds, and personal care products. Advances in analytical capabilities have enabled measurement of these compounds, referred to as emerging contaminants, at environmentally relevant concentrations (parts per billion to parts per trillion). Prior to several European studies (Aherne & Briggs, 1989; Heberer & Stan, 1997; Ternes, 1998), the environmental occurrence of pharmaceutical and hormonally active compounds was relatively unknown. In a survey of U.S. surface waters during 1999–2000, pharmaceutical compounds, hormones, and other organic compounds considered indicative of wastewater were detected in 80% of 139 streams sampled (Kolpin et al., 2002). Although the ecological and human health effects of environmental exposures to most emerging contaminants are largely unknown, concern exists about the potential for adverse effects at environmentally relevant concentrations (Christensen, 1998; Jones, Voulvoulis, & Lester, 2004; Webb, 2001).

Pharmaceutical compounds as well as endogenous hormones are excreted in human wastes and are present in domestic wastewater along with household and personal care products. Despite vigorous oxidative treatment processes such as chlorination and ozonation employed by conventional wastewater treatment plants (WWTPs), many pharmaceutical and hormonally active compounds are not removed (Buser, Poiger, & Müller, 1999; Glassmeyer & Shoemaker, 2005). As a result, effluent from centralized WWTPs, commonly discharged directly into streams, is a major source of pharmaceutical and hormonally active compounds in surface waters (Daughton & Ternes, 1999).

Centralized wastewater treatment systems can also have detrimental effects on water quality because of leaking or overflowing sewer lines. Nationwide, an estimated 3–10 billion gallons of untreated sewage are discharged annually through leaking or overflowing sewer lines (U.S. Environmental Protection Agency, 2002). Rates of leakage from gravity flow sewer lines are highly variable, and leaks may close as particulate materials in sewage are deposited within the lines (Blackwood, Ellis, Revitt, & Gilmour, 2005; Ellis, Revitt, Lister, Willgress, & Buckley, 2003).

TABLE **1**

Characteristics of Study Sites in the Upper Neuse River Basin

Site #	USGS[a] Site ID #	Stream Location	Wastewater Treatment Category	Major Soil Series†	Percolation‡	Drainage Area (km^2)	Housing Density[a] (Houses/km^2)
1	0208503422	Rhodes Creek tributary above NC Highway 751 near Durham, NC	Undeveloped	Helena, Georgeville, Iredell	Slow	0.49	0
2	0208503920	Nancy Rhodes Creek tributary at Cole Mill Road near Durham, NC	Centralized	Appling, Georgeville, Wedowee	Slow	1.04	250
3	02085067	Black Meadow Run at Argonne Drive near Durham, NC	Centralized	Iredell, White Store, Herndon	Slow	1.11	470
4	0208503990	Eno River tributary below Clover Hill Place near Durham, NC	Centralized	Herndon, Georgeville	Slow	0.13	590
5	0208503945	Sevenmile Creek tributary at Inverness Drive near Durham, NC	Onsite	Georgeville, Nason, Herndon	Slow	0.34	140
6	0208505880	Crooked Creek tributary at Greenbay Drive near Durham, NC	Onsite	Herndon, Georgeville, Lignum	Slow	1.68	120
7	0208525095	Cabin Branch tributary at Paragon Circle near Durham, NC	Onsite	Herndon, Lignum, Goldston	Slow	0.41	130

[a]USGS = U.S. Geological Survey; Household densities were calculated from online GIS files of the City of Durham (2005).
†Soil series from U.S. Department of Agriculture (2006a, 2006b).
‡Kirby (1976).

The effects of emerging contaminants in wastewater from decentralized, or onsite, wastewater treatment systems on streams are not well known. Movement of treated wastewater leaving onsite system drainfields has lateral and downward components, rates of which depend on properties of the underlying soil and distance to the water table. Upon reaching the saturated zone, treated wastewater is eventually discharged to streams. Wastewater from onsite systems primarily is treated by the processes of microbial degradation and sorption (Dawes & Goonetilleke, 2003; Hallahan, 2002). Studies investigating the degradation of surfactants (Doi et al., 2002; McAvoy, DeCarvalho, Nielson, & Cano, 2002) and other detergent components (Lee, McAvoy, Szydlik, & Schnoor, 1998) by onsite systems showed varying rates of removal. A variety of surfactant metabolites, disinfectants, caffeine, fecal sterols, and pharmaceutical compounds including antibiotics were identified in residential septic tank effluent (Conn, Barber, Brown, & Siegrist, 2006). Percolation through subsurface soils resulted in removal rates exceeding 90% for most of these compounds (Conn, Siegrist, Barber, & Meyer, 2010). Removal efficiencies may vary depending on soil type and thickness.

Onsite wastewater treatment systems are used by about 30% of the U.S. population (U.S. Census Bureau, 2001) and are common in rural and low-density residential areas. Properly functioning onsite systems remove nutrients and many other compounds from domestic wastewater. The estimated effective lifespan of onsite systems typically ranges from 11 to over 30 years (Siegrist, Tyler, & Jennsen, 2001) and about half of U.S. septic systems are more than 30 years old (U.S. Census Bureau, 2001). The goal of our study was to compare the water quality in small streams in areas where domestic wastewater was primarily treated by either centralized or onsite systems.

Study Area

The study area was in the upper Neuse River basin in Orange and Durham counties, North Carolina, an area where population growth has resulted in rapid residential expansion into areas lacking the infrastructure required for centralized wastewater treatment. As a result, increasing numbers of onsite systems have been installed in the basin. An estimated 30% and 72% of the residences in Durham and Orange counties, respectively, use onsite wastewater treatment (North Carolina Department of Environment and Natural Resources, Division of Environmental Health, 2003). Major soil groups in the catchments of the study sites are characterized as moderately to severely limited for onsite drainfields because of slow percolation (Kirby, 1976). Streams in the study area flow into the Falls Lake reservoir, the water supply for the city of Raleigh.

Methods

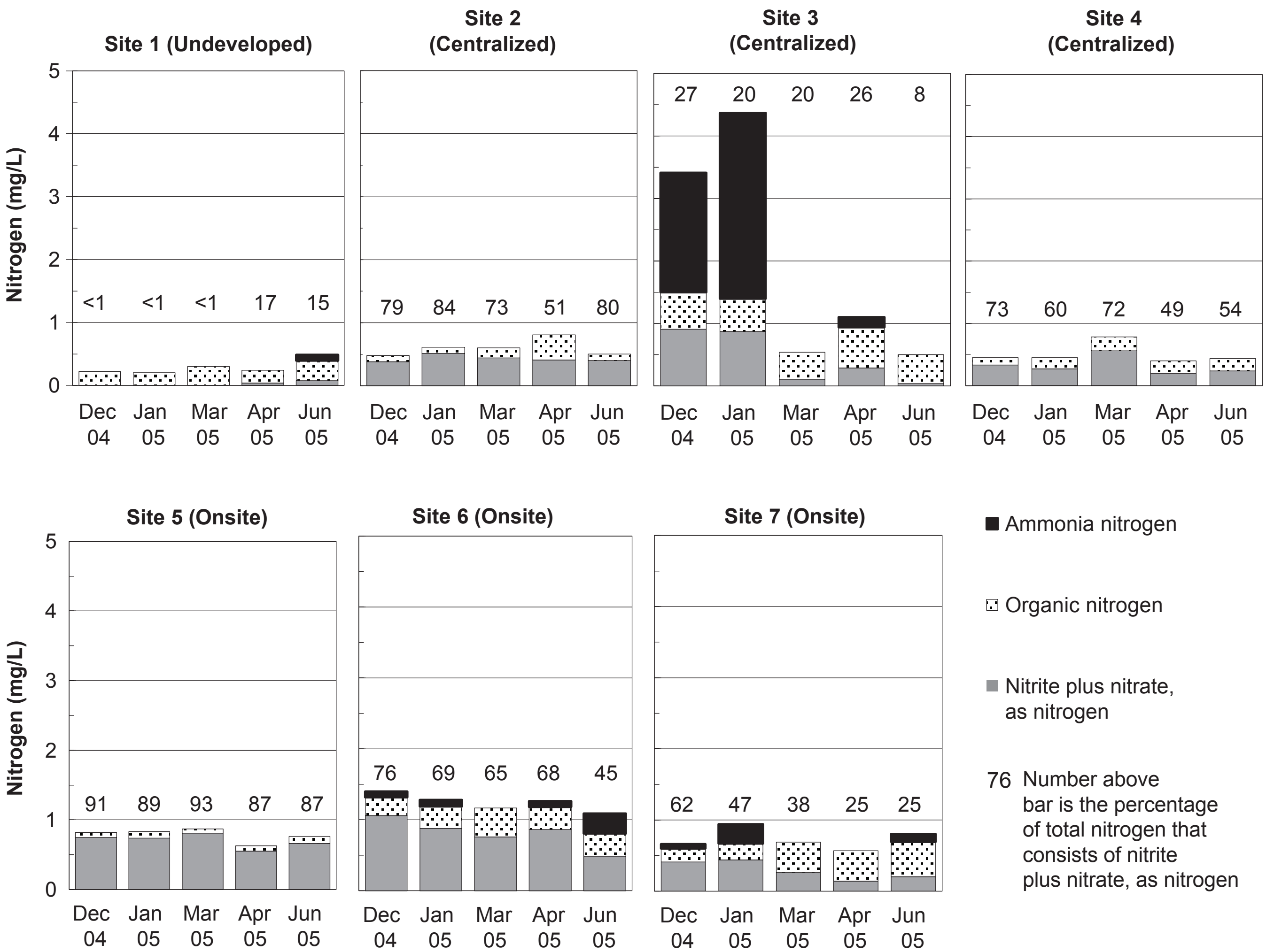
Site Selection and Characterization

Sites are in areas underlain by metamorphic and igneous rock where topographic relief and shallow depth to groundwater are anticipated to provide a short flow path for movement of treated wastewater to streams.

A sampling network (sites 1–7) was established to compare the effects of centralized and onsite wastewater treatment on stream quality in small catchments and includes seven sites with drainage areas ranging from 0.13 km^2 to 1.68 km^2 (Table 1).

FIGURE **1**

Temporal Variation in Concentrations of Dissolved Ammonia, Organic Nitrogen, and Nitrite Plus Nitrate



In water samples from streams in areas that are undeveloped, residential with centralized wastewater treatment, and residential with onsite wastewater treatment.

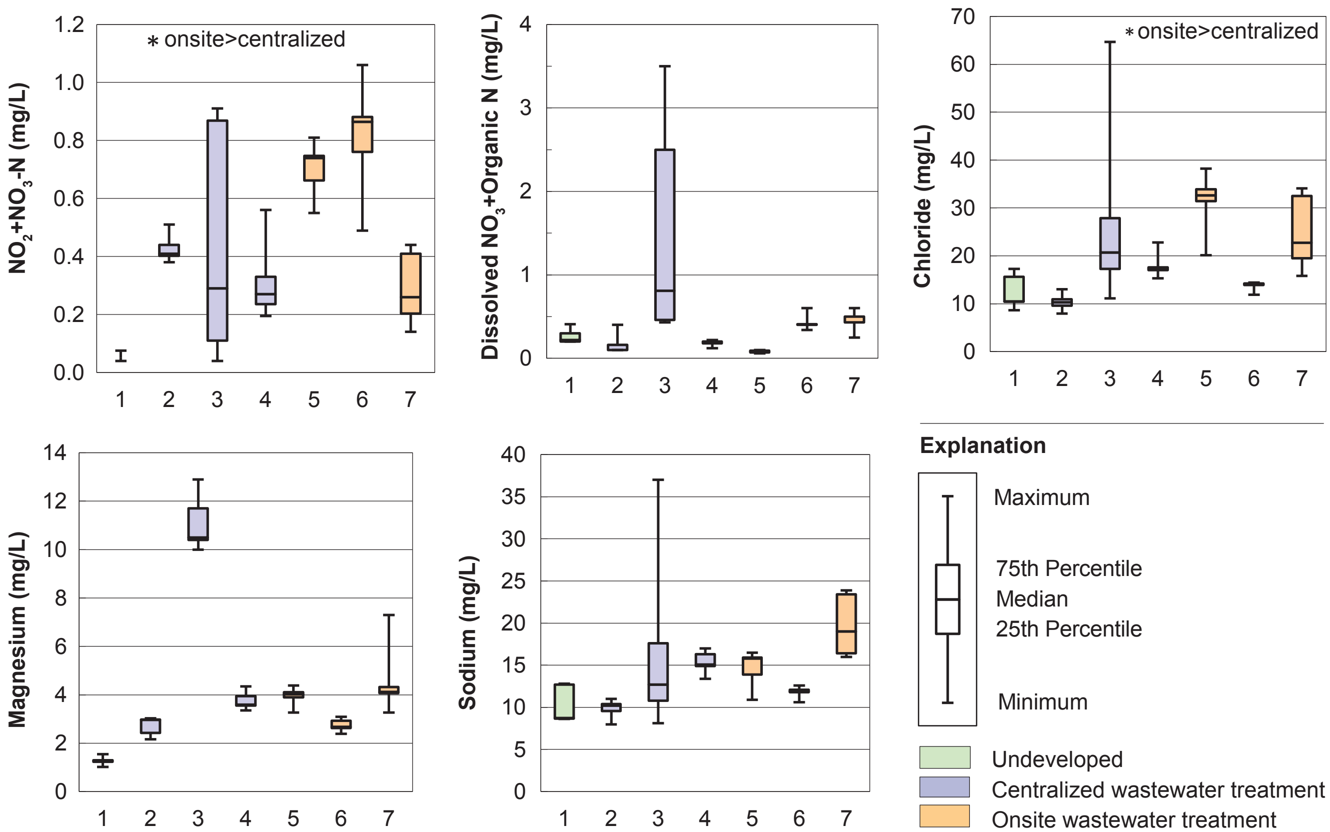
Six of the sites are in primarily residential areas and one (site 1), considered to be representative of background conditions, is in an undeveloped catchment managed for timber production. Three of the residential catchments (sites 2–4) are in areas served by a municipal sewer system and three (sites 5–7) are in areas where wastewater is treated on site. Household densities in the basins with centralized wastewater treatment are about 2–5 times greater than in the basins with onsite wastewater treatment. The majority of the residences in these catchments were constructed during 1960–1985.

Sample Collection and Analysis

Water samples were collected and processed according to guidelines of the U.S. Geological Survey (USGS, Various Dates). Upon completion of processing, samples were placed on ice and transported by overnight carrier to the analyzing laboratory. Samples were collected from December 2004 to June 2005 under baseflow conditions except for the samples collected on June 16, 2005, some of which were affected by runoff from thunderstorms. Equipment blanks and replicate samples were obtained for about 10% of the samples. Water samples were analyzed by the USGS National Water Quality Laboratory in Denver, Colorado. Samples were analyzed for nutrients, ions, and metals using methods described by Fishman and Friedman (1989) and for a suite of 59 organic wastewater compounds (OWWCs) using capillary column gas chromatography/mass spectroscopy methods described by Zaugg and co-authors

FIGURE **2**

Concentrations of Selected Dissolved Ions and Nutrients in Stream Water Samples



Site 1 is in an undeveloped area; sites 2–4 are in areas served by a centralized wastewater treatment system; sites 5–7 are in areas of onsite wastewater treatment; * denotes statistically significant difference between sites in areas of centralized and onsite wastewater treatment, $p = .05$.

(2002). Thirty-three pharmaceutical compounds, including antibiotics, carbamazepine, and ibuprofen were analyzed by the USGS Organic Geochemistry Laboratory in Lawrence, Kansas, using the online solid phase extraction and liquid chromatography/tandem mass spectrometry with electrospray ionization method described by Meyer and co-authors (2007).

Water samples collected for analysis of fecal bacteria were analyzed at the Duke University Marine Laboratory in Beaufort, North Carolina. Samples were analyzed for *E. coli* and fecal coliform bacteria using the five-tube dilution method with A-1 media according to Eaton and co-authors (1995). Water samples were analyzed for optical brighteners (OBs) at Virginia Tech in Blacksburg, Virginia, using techniques described by Dickerson and co-authors (2007). To prevent photodegradation, samples were stored in darkness until analyzed. Samples for OBs were collected in conjunction with samples analyzed for *E. coli* and fecal coliforms on June 1 and 16, 2005, to determine if OB fluorescence was correlated with fecal bacteria.

Nonparametric statistical comparisons of water quality data were made using the Mann-Whitney-Wilcoxon test (Conover, 1980) to determine differences between samples collected from streams draining areas with centralized and onsite wastewater treatment. Samples were ranked by collection period and tests were performed on ranked values.

Results and Discussion

Nutrients and Ions in Water Samples

Concentrations of nutrients in samples from the undeveloped catchment were lower than those in samples from the residential catchments and indicate the lack of human activities and the greater nutrient uptake associated with forested lands. Nutrient concentrations generally were similar among samples from the residential stream sites (sites 2–7) with the exception of site 3, which had higher concentrations of total nitrogen and ammonia concentrations, a smaller proportion of nitrogen in the form of NO_2+NO_3 (nitrite plus nitrate), and greater overall variability

TABLE **2**

Summary of Organic Wastewater Compounds Detected in Streamwater Samples

Analyte†	Primary Use or Source	Reporting Level (µg/L)	Site Type						
			Undeveloped	Residential Basins					
				Centralized Wastewater Treatment			Onsite Wastewater Treatment		
			Site 1 (*n* = 5)	Site 2 (*n* = 5)	Site 3 (*n* = 5)	Site 4 (*n* = 5)	Site 5 (*n* = 5)	Site 6 (*n* = 5)	Site 7 (*n* = 5)
Anthraquinone	Manufacturing	0.5	ND[a]	ND	(2)	ND	ND	ND	(1)
Isophorone	Solvent	0.5	(1)	ND	(1)	ND	ND	(1)	(2)
p-Cresol	Wood preservative and auto exhaust	1	(2)	(2)	(2)	(1)	(1)	(3)	(2)
5-Methyl-1H-benzotriazole	De-icing agent	2	ND	ND	(1)	ND	ND	ND	ND
4-t-Octylphenol diethoxylate	Detergent metabolites	**1**	**(1)**	**ND**	**ND**	**ND**	**ND**	**ND**	**ND**
4-t-Octylphenol monoethoxylate		**1**	**ND**	**(1)**	**(1)**	**ND**	**ND**	**(1)**	**(1)**
4-Nonylphenol		**5**	**(1)**	**(1)**	**(1)**	**ND**	**ND**	**(1)**	**(1)**
4-Nonylphenol diethoxylate		**5**	**(1)**	**ND**	**(1)**	**ND**	**ND**	**(1)**	**(1)**
Tris(2-chloroethyl)phosphate	Flame retardants and plasticizers	**0.5**	**ND**	**(1)**	**(2)**	**ND**	**ND**	**ND**	**ND**
Tris(2-butoxyethyl) phosphate		**0.5**	**(1)**	**ND**	**2(3)**	**ND**	**ND**	**(1)**	**1(1)**
Tris(dichloroisopropyl)phosphate		**0.5**	**ND**	**(1)**	**(2)**	**ND**	**ND**	**ND**	**(1)**
Triphenyl phosphate		**0.5**	**ND**	**ND**	**2**	**ND**	**ND**	**ND**	**ND**
3-Methyl-1H-indole (skatole)	Fragrances (many have natural sources)	1	(1)	(1)	(2)	ND	ND	(2)	(1)
3-b-Coprostanol		2	(1)	ND	(1)	ND	(1)	ND	ND
Acetyl hexamethyl tetrahydronaphthalene		0.5	ND	(1)	(1)	ND	ND	(4)	(4)
Hexahydrohexamethyl cyclopentabenzopyran		0.5	ND	ND	ND	ND	ND	(2)	(4)
Indole		0.5	(1)	ND	(2)	ND	ND	(2)	(2)
Isoborneol		0.5	ND	ND	(1)	ND	ND	ND	ND
Menthol		0.5	ND	ND	(2)	ND	ND	(1)	ND
Methyl salicylate		0.5	(2)	ND	(2)	(1)	ND	(1)	(1)
Benzophenone		**0.5**	**(1)**	**ND**	**ND**	**ND**	**ND**	**ND**	**ND**
Metolachlor	Pesticides	0.5	ND	(1)	(1)	ND	ND	ND	ND
Prometon		0.5	ND	ND	ND	(1)	ND	ND	ND
Carbaryl		1	ND	ND	(1)	(1)	ND	ND	(1)
Benzo[a]pyrene	PAH[a]	0.5	ND	ND	ND	ND	ND	ND	(1)
Naphthalene		0.5	(1)	ND	(3)	ND	ND	ND	ND
Fluoranthene		0.5	ND	ND	(1)	ND	ND	ND	ND
Phenanthrene		0.5	(1)	ND	(1)	ND	ND	ND	ND
Pyrene		0.5	ND	ND	(1)	ND	ND	ND	ND
b-Stigmastanol	Plant and animal steroids	2	ND	(1)	(2)	ND	(1)	ND	(1)
b-Sitosterol		2	(1)	(2)	(2)	ND	(1)	(1)	ND
Cholesterol		2	(1)	(1)	1(2)	(1)	(1)	(1)	(2)
Tributyl phosphate	Plasticizers	0.5	(3)	1(1)	1(1)	ND	(1)	1	1
Triethyl citrate		0.5	ND	ND	ND	ND	ND	(1)	ND
1-Methylnaphthalene	Coal tar, gasoline	0.5	(1)	(1)	(1)	(1)	(1)	(1)	(1)
2-Methylnaphthalene		0.5	(1)	(1)	(1)	(1)	(1)	(1)	(1)
Isoquinoline	Coal tar	0.5	ND	ND	1	ND	ND	1	ND
Carbazole	Pigment, coal tar	0.5	ND	ND	(1)	ND	ND	ND	ND
Triclosan	Disinfectant	**1**	**(1)**	**ND**	**(1)**	**ND**	**ND**	**(1)**	**ND**
Camphor	Flavoring agent	0.5	ND	(3)	(3)	(3)	(4)	(2)	(2)
Cotinine	Nicotine metabolite	1	ND	(1)	(1)	ND	ND	ND	(1)
Caffeine	Stimulant	0.5	ND	(1)	1(2)	(1)	(1)	(3)	(3)
Number of analytes detected†			19	17	36	9	11	22	23

Note. Qualitative detections (presence confirmed but not quantifiable) are shown in parentheses. Analytes suspected of hormonal activity are shown in bold.

[a]ND = not detected; PAH = polycyclic aromatic hydrocarbon.

†Anthracene, chlorpyrifos, diazinon, isopropylbenzene, 2,6-dimethylnaphthalene, 1,4-dichlorobenzene, bromacil, 3-t-butyl-4-hydroxyanisole, isopropylbenzene, 4-cumylphenol, and 4-n-octylphenol were not detected in samples.

TABLE **3**

Detections of Antibiotic and Pharmaceutical Compounds in Streamwater Samples From Study Sites in the Upper Neuse River Basin

Analyte (Reporting Limit in µg/L)	Category	Use[a]	Site Type						
			Undeveloped	Residential Basins					
				Centralized Wastewater Treatment			Onsite Wastewater Treatment		
			Site 1 (*n* = 5)	Site 2 (*n* = 5)	Site 3 (*n* = 5)	Site 4 (*n* = 5)	Site 5 (*n* = 5)	Site 6 (*n* = 5)	Site 7 (*n* = 5)
Azithromycin (0.005)	Antibiotic	H	ND[a]	ND	ND	1 (0.011)	1 (0.008)	ND	ND
Carbamazepine (0.005)	Anticonvulsant	H	ND	2 (0.007)	ND	1 (0.011)	1 (0.004)	4 (0.023)	2 (0.01)
Erythromycin (0.008)	Antibiotic	HV	ND	ND	ND	1 (0.009)	ND	ND	ND
Anhydro-erythromycin (0.008)	Antibiotic metabolite	–	ND	ND	1 (0.017)	1 (0.014)	ND	ND	ND
Ibuprofen (0.050)	Anti-inflammatory	H	ND	ND	ND	ND	ND	ND	1 (0.084)
Lincomycin (0.005)	Antibiotic	V	ND	ND	ND	1 (0.027)	ND	ND	ND
Tylosin (0.005)	Antibiotic	V	ND	ND	2 (0.007)	1 (0.019)	1 (0.025)	ND	ND
Total number of detections			0	2	3	6	3	4	3
Number of compounds detected			0	1	2	6	3	1	2

Note. Value shown in parentheses is maximum concentration. Ciprofloxacin, chlortetracycline, doxyclocyline, enrofloxacin, epi-chlortetracycline, epi-oxytetracycline, epi-tetracycline, iso-chlortetracycline, lomefloxacin, norfloxacin, ofloxacin, ormetoprim, oxytetracycline, roxityromycin, sarafloxacin, sulfachloropyridazine, sulfadiazine, sulfadimethoxine, sulfamethazine, sulfamethoxazole sulfathiazole, tetracycline, trimethoprim, and virginiamycin were not detected in any of the samples.

[a]ND = not detected; H = human; V = veterinary; – = not applicable.

(Figures 1 and 2). The NH_3-N (ammonia, as N) concentrations in samples collected in December 2004 and January 2005 from site 3 located in a catchment served by centralized wastewater treatment were about 2 and 3 mg/L, respectively. These high concentrations of NH_3-N indicate the presence of wastewater, the source of which is likely a leaking sewer line in an area with centralized wastewater treatment. With the exception of one sample from site 6 (onsite wastewater treatment) that had a concentration of 1.06 mg/L, concentrations of NO_2+NO_3-N were less than 1 mg/L.

Overall, NO_2+NO_3-N concentrations were statistically higher (p = .05) in samples from sites in areas of onsite wastewater treatment than from sites in areas of centralized wastewater treatment with average concentrations of 0.60 mg/L and 0.40 mg/L, respectively. Interestingly, the site with the lowest average concentration of NO_2+NO_3-N, 0.29 mg/L (site 7), is in an area where onsite wastewater treatment is used. Orthophosphate concentrations were near or less than the reporting limit of 0.02 mg/L at all sites. Onsite treatment processes typically remove 85% to 95% of the influent phosphorus, whereas removal rates for nitrogen range from 10% to 40% (Sikora & Corey, 1976).

Variations in nutrient concentrations were small and likely occurred in response to seasonal and hydrologic conditions with the exception of site 3 (Figures 1 and 2). Concentrations of ammonia in the samples collected after January 2005 at site 3 decreased to levels near the detection limit. The large variation in concentrations of nitrogen compounds in samples where a sewer line leak was suspected at site 3 could be due to changes in rates of leakage or to decreased microbial transformation and biological uptake associated with colder temperatures during winter months. Dry conditions during May and June also could have slowed subsurface flow and enabled increased assimilation of nitrate. Deposition of particulate materials within the sewer lines has been shown to decrease rates of leakage (Ellis et al., 2003). Fertilizer applications to lawns are an additional source of nitrogen in the residential catchments.

Concentrations of ions generally were lowest in the samples from the undeveloped site (site 1). Concentrations of chloride were statistically higher in samples from streams draining areas served by onsite wastewater treatment in comparison to areas served by centralized wastewater treatment facilities, whereas concentrations of potassium and sulfate were lower. Concentrations of calcium, potassium, and sulfate varied more in samples from the site with the suspected sewer line leak (site 3) than in the other sites (Figure 2). Differences in ion concentrations among sites are likely related to differences in soils and geology of the study sites.

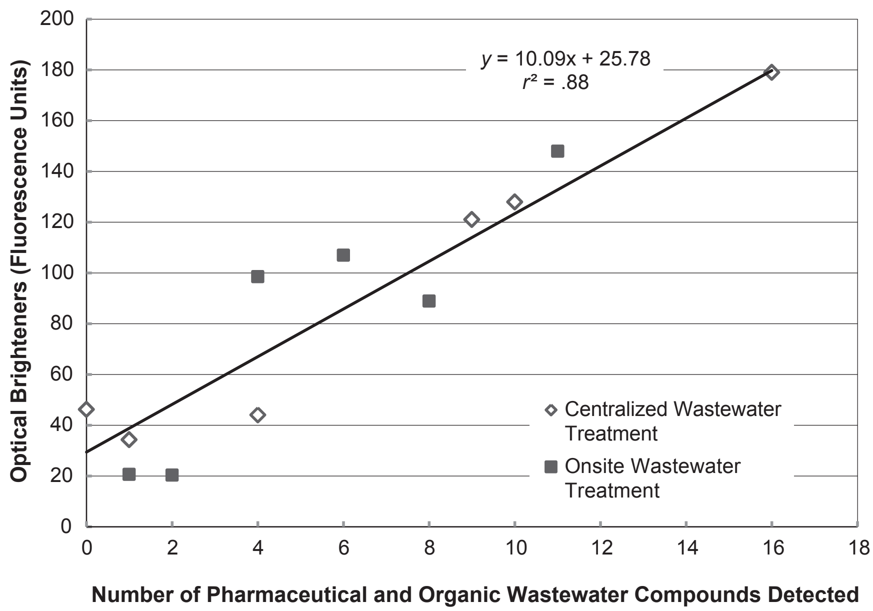
OWWCs and Pharmaceuticals

Detections of OWWCs and pharmaceuticals are summarized in Tables 2 and 3, respectively. Concentrations of OWWCs generally were detected at trace levels (less than reporting limits) and did not exceed maximum contaminant levels established by the U.S. Environmental Protection Agency. Standards have not been established, however, for most of these compounds.

The most OWWCs (36) were detected in samples from the site with the suspected sewer line leak (site 3). The second largest number of detections (23) was from site 7 (onsite wastewater treatment). OWWCs detected at the background site (site 1) included cholesterol, likely derived from wildlife, detergent metabolites, a plasticizer, and fragrance components that occur naturally at trace levels. Detections of pharmaceutical compounds did not correspond to detections of OWWCs. The greatest number of pharmaceuticals (six), four of which are used for veterinary purposes, was detected in samples from site 4, a site with centralized wastewater treatment and the highest housing density of the residential sites (590 houses/km^2; Table 1). The number and frequency of detection of pharmaceutical compounds were similar among samples at the remainder of the residential sites with no detections in samples from the undeveloped site (site 1). Carbamazepine was detected in samples from all but one of the residential stream sites. Previous investigations have found carbamazepine to be widely distributed in water and wastewater because of its resistance to degradation and widespread use (Clara, Strenn, & Kreuzinger, 2004).

FIGURE **3**

Relationship Between Optical Brighteners Fluorescence and the Number of Pharmaceutical and Organic Wastewater Compounds



Detected in water samples from small streams in selected residential areas. Organic wastewater compounds detected in associated laboratory and field blanks were omitted from the regression analysis.

OBs and Fecal Coliform Bacteria

Samples for OBs were analyzed in conjunction with samples analyzed for OWWCs and pharmaceuticals during January and April 2005 and with samples analyzed for fecal coliforms and *E. coli* collected on June 1 and 16, 2005. OBs are detergent additives that fluoresce under the visible spectrum to enhance the appearance of white fabrics and have been used as indicators of wastewater (Dickerson et al., 2007; Stoll & Giger, 1998). OBs are photochemically degraded (Stoll, Ulrich, & Giger, 1998) and exhibit fluorescence patterns similar to some naturally occurring organic compounds. Consequently, the use of OBs as wastewater indicators is limited to streams with low concentrations of interfering compounds and to tracking wastewater over only short time periods because of photodegradation. Aqueous extracts of loblolly pine needles fluoresce at the same wavelengths as optical brighteners (unpublished data). Compounds in pine foliage are likely the source of the high OB values in samples from the undeveloped site (site 1), which is located in a pine forest. Levels of OBs in the streams draining residential areas, which have primarily hardwood trees, were highly correlated (r^2 = .88) with the number of OWWCs and pharmaceuticals detected (Figure 3).

Fecal coliform samples were comprised of *E. coli* (Table 4) and densities did not correlate with levels of OBs or type of wastewater treatment. Coliform densities were lower in the June 1 samples than in the June 16 samples. Samples collected on June 16 were affected by runoff from thunderstorms that contributed to increased bacterial densities in some of the samples. The highest coliform densities for both sampling periods were in samples from site 6, which is located in an area of onsite wastewater treatment and may indicate the presence of poorly functioning onsite treatment systems within the catchment. Properly functioning onsite wastewater treatment systems are capable of removing more than 99% of the fecal bacteria in domestic wastewater (Gerba, Wallis, & Melnick, 1975). Similarly, fecal coliform bacteria from leaking sewer lines are removed by sorption to soils within short distances of the source of the leak (Hua, An, Winter, & Gallert, 2003).

Two sites, one in an area of centralized wastewater treatment and the other in an area of onsite wastewater treatment, showed greater effects of wastewater than the other sites. Site 3 (centralized wastewater treatment) had the greatest number of detections of OWWCs and the highest concentrations of NH_3-N. Site 6 (onsite wastewater treatment) had the highest concentrations of NO_2+NO_3-N and the largest densities of fecal coliform bacteria. Water quality for the remaining sites showed little difference with respect to type of wastewater treatment.

Conclusion

Nutrient and ion concentrations generally were larger in samples from the residential areas than the undeveloped

TABLE 4

Optical Brighteners and Fecal Bacteria in Streamwater Samples From Study Sites in the Upper Neuse River Basin

Site Type	Site #	Optical Brighteners (Fluorescence Units)				Fecal Coliform Density MPN[a] (col/100 mL[a])[†]	*E. coli* Density MPN (col/100 mL)[†]	% *E. coli*	Fecal Coliform Density MPN (col/100 mL)[†]	*E. coli* Density MPN (col/100 mL)[†]	% *E. coli*
		January 2005	April 2005	June 1, 2005	June 16, 2005	June 1, 2005			June 16, 2005		
Undeveloped	1	99	105	71.8	81.5	23 (9–86)	23 (9–86)	100	140 (60–340)	140 (60–340)	100
Centralized	2	34.3	121	38.3	45.9	70 (30–210)	70 (30–210)	100	240 (100–940)	240 (100–940)	100
Centralized	3	128	179	164	128	11 (4–29)	11 (4–29)	100	130 (50–390)	130 (50–390)	100
Centralized	4	46.2	44.1	30.8	42.6	23 (9–86)	23 (9–86)	100	130 (50–390)	130 (50–390)	100
Offsite 2 drain	–[a]	–	–	191	310	–	–	–	130 (50–390)	49 (20–170)	38
Onsite	5	20.6	20.4	20.2	14.1	7 (2–20)	7 (2–20)	100	170 (70–480)	170 (70–480)	100
Onsite	6	98.5	88.9	82.8	110	350 (160–820)	350 (160–820)	100	540 (200–2,000)	540 (200–2,000)	100
Onsite	7	107	148	191	136	13 (5–38)	13 (5–38)	100	240 (1–940)	240 (1–940)	100

[a]MPN = most probable number; col/100 mL = colonies per 100 milliliters; – = not analyzed.
[†]Value shown in parentheses is the 95% confidence interval for the MPN.

area and likely reflect human activities. Effects of onsite treatment on streams were not obvious. Two of the sites, site 3, in an area of centralized wastewater treatment near a suspected sewer line leak, and site 6, in an area of onsite wastewater treatment, appeared to show effects of wastewater, whereas effects were less obvious for the remaining sites. Concentrations of NO_2+NO_3-N and chloride in samples from sites in areas of onsite wastewater treatment were statistically larger (p = .05) than those from sites in areas of centralized wastewater treatment. The site with the lowest NO_2+NO_3-N concentrations was in an area of onsite wastewater treatment, which indicates onsite systems can effectively treat wastewater.

Concentrations of OWWCs and pharmaceuticals were near detection limits. The largest number of these compounds was detected in samples from the site with the suspected sewer line leak. Differences in water quality were as large within as between the residential wastewater treatment groups (centralized and onsite). OB levels were strongly correlated (r^2 = .88) with the presence of OWWCs and pharmaceuticals in streams draining residential areas. Although OWWCs and pharmaceuticals do not have the same fluorescence spectra as OBs, they co-occur with OBs in wastewater. Because of this association, measurement of OBs may provide an economical means of screening for wastewater in small streams. Methods of decreasing background interferences from naturally occurring organic matter are needed, however. Samples from site 3 (centralized wastewater treatment) and site 6 (onsite wastewater treatment) generally had higher levels of nutrients, OBs, and numbers of detected OWWCs than the other sites, which suggests less effective treatment of wastewater at these locations.

The results of our study indicate that properly functioning onsite wastewater treatment systems had little effect on stream quality. Our study also illustrates that use of centralized wastewater treatment does not eliminate sources of wastewater and the impact of a leaking sewer line can be as significant as that associated with poorly functioning onsite wastewater treatment systems. Comparison of the effects of wastewater treatment practices on stream quality was limited, however, by the small numbers of sites and samples. Because streams in residential areas provide a potential route of human exposure to emerging contaminants, further investigation is needed. Analytical data collected as part of our study are available online through the USGS National Water Information System at http://waterdata.usgs.gov/nc/nwis/sw.

continued on page 26

continued from page 25

Acknowledgements: Funding for this study was provided by a grant to the North Carolina Department of Environment and Natural Resources, Division of Environmental Health, through the U.S. EPA's Clean Water Act Section 319 program and the USGS Cooperative Funding program. Assistance provided by the city and county of Durham, North Carolina, in delineating areas served by onsite and centralized wastewater treatment was greatly appreciated.

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Fate and Transport of Phosphate From an Onsite Wastewater System in Beaufort County, North Carolina

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Abstract **The objectives for the study described in this article were to evaluate the fate and transport of onsite wastewater system (OWS)–derived phosphate from a residential system in Beaufort County, North Carolina, and to determine if current OWS setback regulations are sufficient to prevent elevated phosphate discharge to surface waters. Piezometers were installed in nests at different depths adjacent to drainfield trenches and up- and down-gradient of a residential OWS. Groundwater and septic effluent phosphate concentrations, temperature, pH, dissolved oxygen, and electrical conductivity were monitored every two months from February 2011 to October 2011 (five times). The mean groundwater phosphate concentration beneath the OWS (3.05 ± 0.74 mg/L) was not significantly different than septic effluent (2.97 ± 0.76 mg/L) and was elevated relative to background groundwater (0.14 ± 0.12 mg/L). Groundwater phosphate concentrations were inversely related (r^2 = .83) to distance from the system. Onsite system setback regulations may have to be increased (>30 m) in some areas to ensure groundwater phosphate concentrations are reduced to background concentrations before discharge to surface waters.**

Introduction

Water Quality Issues Related To Excess Phosphorus

Eutrophic conditions and fish kills continue to be problematic in the Tar-Pamlico watershed in North Carolina (North Carolina Division of Water Quality, 2010). Nutrient-sensitive water management strategies were implemented in 2000 to help reduce nitrogen loads to the watershed and cap discharges of phosphorus from various point and nonpoint sources of nutrient pollution. Despite these efforts, elevated nutrient problems still exist in the river and estuary. More scrutiny is now being placed on potential sources of pollution such as onsite wastewater systems (OWS) that were not addressed in previous regulations.

OWS and Phosphorus Attenuation

OWS process wastewater containing elevated phosphorus concentrations and are used by approximately 48% of residences in the nutrient-sensitive Tar-Pamlico River basin (Pradhan, Hoover, Austin, & Devine, 2007). Therefore, it is important to determine the nutrient reduction efficiency of OWS. While some phosphorus is retained in the septic tank with the settled solids, effluent leaving the tank is still enriched with total phosphorus (5–15 mg/L) and phosphate (1.2–12.1 mg/L) (Corbett, Dillon, Burnett, & Schaefer, 2002; Humphrey & O'Driscoll, 2011; Robertson, Schiff, & Ptacek, 1998; U.S. Environmental Protection Agency, 2002). The soil beneath the drainfield trenches and down-gradient from the system may provide an environment for phosphate attenuation. In the soil beneath drainfield trenches, the dominant phosphate removal processes are mineral precipitation and adsorption (Robertson et al., 1998). Mineral precipitation occurs when phosphate combines with other elements to form solids and thereby limits transport of phosphate away from the OWS. Soil capacity to adsorb significant amounts of phosphate onto reactive surfaces such as clay minerals and iron oxide coatings diminishes over time and can be reversed (desorption), thus releasing phosphate to groundwater. When phosphate adsorption surfaces are completely occupied, phosphate leaching and transport can occur. Thus, mineral precipitation is often considered a more stable sink for phosphate reduction (Harmon, Robertson, Cherry, & Zanini, 1996).

Mineral Precipitation

Mineral precipitation beneath OWS is dictated by the pH and redox status of the soils and the availability of other elements such as calcium, iron, or aluminum that combine with phosphate to form various minerals (Zanini, Robertson, Ptacek, Schiff, & Mayer, 1998). For example, the mineral variscite ($AlPO_{4*}2H_2O$) may precipitate when soil pH is below 5.5, anaerobic conditions are present, and aluminum and phosphate are available. When aerobic conditions are present with sufficient aluminum and phosphate and soil pH is below 8, variscite may also precipitate (Robertson et al., 1998). Strengite ($FePO_{4*}2H_2O$) precipitation can occur when soil or groundwater pH is between 6 and 8, aerobic conditions are present, and phosphate and iron are available (Robertson et al., 1998).

Adsorption

Several studies have indicated that soil texture beneath OWS influences phosphate treatment with fine-textured soils providing more phosphate reduction than coarse-textured soils. The increased surface area of silt-rich and clay-rich soils provides more opportunity for adsorption and also increases the residence time of wastewater in the unsaturated zone, enhancing phosphate attenuation. Robertson and co-authors (1998) reported in a literature review of 10 OWS in Canada that OWS installed in more fine-textured soils such as silt had much shorter phosphate plume lengths than medium- and coarse-grained sands. Karathanasis and co-authors (2006) evaluated the phosphorus removal efficiency of different soil types by leaching wastewater through soil monoliths. They found that in general, clay-textured soils provided better total phosphorus treatment than sandy soils. A recent study in coastal North Carolina (Humphrey & O'Driscoll, 2011) documented significant groundwater phosphate concentrations beneath OWS in sandy, slightly acidic soils (mean: 2.46 mg/L), but much lower groundwater phosphate concentrations beneath systems in more fine-textured sandy clay loams (mean: 0.04 mg/L).

Phosphate Transport

Shallow groundwater phosphate concentrations adjacent to OWS can exceed 2 mg/L in some coastal areas (Humphrey & O'Driscoll, 2011; Ptacek, 1998; Robertson, Cherry, & Sudicky, 1991). Because phosphate concentrations two orders of magnitude lower (~0.03 mg/L phosphate) may stimulate algal blooms in some surface waters, OWS near coastal waters must be efficient at reducing phosphate transport or surface water degradation may occur (Ptacek, 1998). Research conducted in coastal sandy areas of Rhode Island (Postma, Gold, & Loomis, 1992) and in coastal Virginia (Reay, 2004) indicated minimal groundwater phosphate transport away from conventional OWS. Environmental conditions conducive to phosphate attenuation via mineral precipitation and adsorption must have been present at those sites. Research conducted at three permanent residences on St. George Island, Florida, however, indicated significant groundwater phosphate transport in the sandy surficial aquifer more than 30 m from three different types of OWS (a conventional, an aerobic, and an elevated OWS) (Corbett et al., 2002). Humphrey and O'Driscoll (2011) reported elevated groundwater phosphate concentrations beneath OWS in coastal sandy soils of North Carolina. Therefore, potential exists in some environments for significant phosphate transport down-gradient from OWS. Many coastal North Carolina water resources are impaired because of excess nutrients, so more research is needed to evaluate phosphate transport from OWS in these settings.

Our study objectives were to evaluate the fate and transport of OWS-derived phosphate from a residential system in Beaufort County, North Carolina, and to determine if current OWS setback regulations are sufficient to prevent elevated phosphate discharge to surface waters.

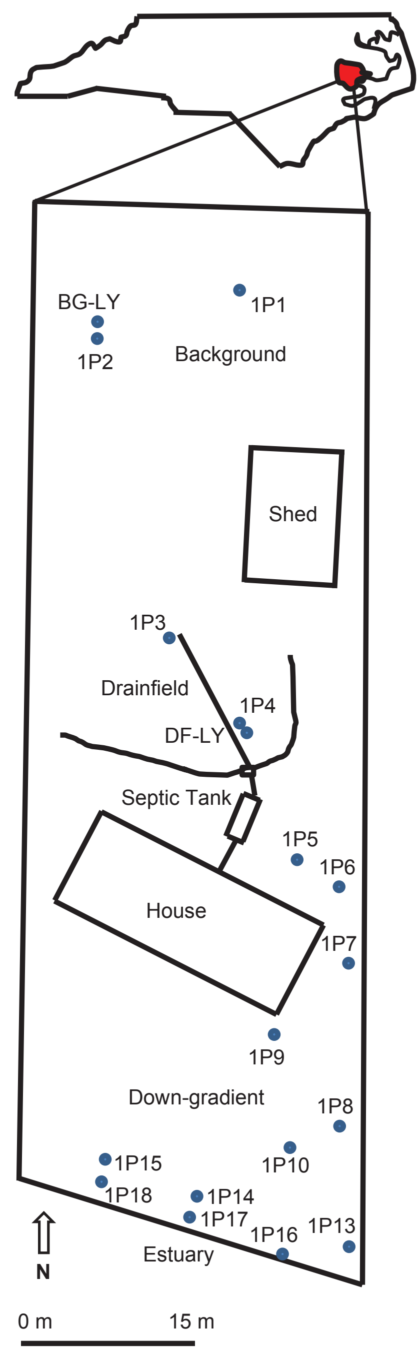
Methods

Site and System Location

A residential site in coastal Beaufort County, North Carolina, was chosen for our study because it bordered the nutrient-sensitive waters of the Tar-Pamlico estuary (Figure 1). The system components (3,780 L tank, distribution box, and 3–15 m long drainfield trenches) were located on site using the OWS permit information and a tile drain probe rod. The system was installed in the early 1980s and was utilized by two

FIGURE 1

Research Site in Beaufort County, North Carolina

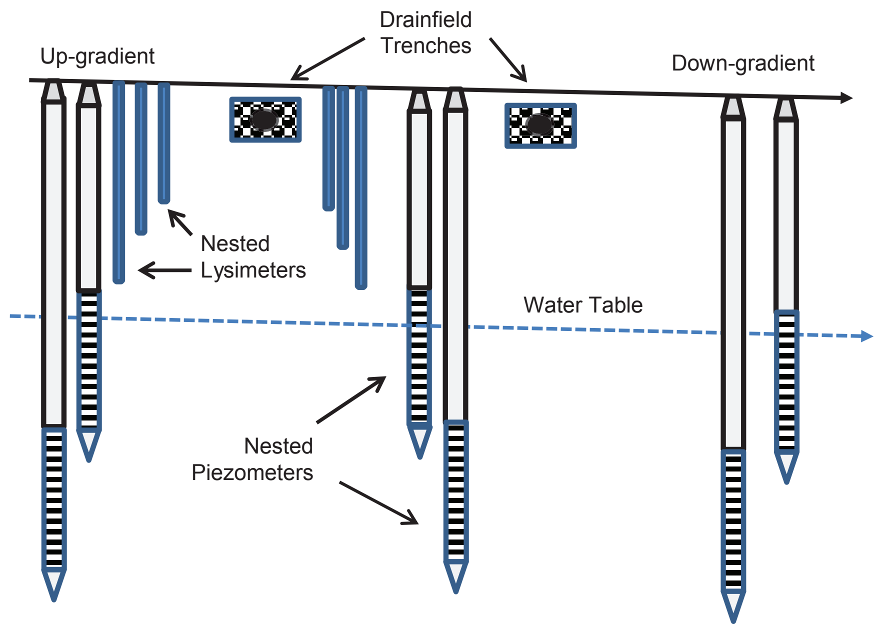


Piezometer and lysimeter nests are shown as blue circles. Piezometer nests for groundwater monitoring are numbered 1p1 to 1p18. Lysimeter nests are located up-gradient from the system (BG-LY) and between the drainfield trenches (DF-LY).

people during our study. Groundwater flow direction and the orientation of the septic plumes were estimated using electrical resistivity surveying and three-point contouring, with some data from exploratory groundwater piezometers (Heath, 1998; Humphrey, Deal, O'Driscoll, & Lindbo, 2010). Because wastewater contains elevated concentrations of salts relative to fresh groundwater, groundwater affected by wastewater exhibits elevated electrical conductivity. Resistivity surveys allow production of 3-D images that reveal variations in the conductive properties of the subsurface and thus help delineate wastewater plumes (Humphrey et al., 2010). The orientation of the plume image and the three-point water level contouring method (Heath, 1998) were used to determine the groundwater flow direction and to decide the location of groundwater and soil water monitoring points.

FIGURE **2**

Soil Water and Groundwater Monitoring Network



Network includes piezometers and lysimeters nested at different depths up-gradient and between drainfield trenches. Piezometers were also nested down-gradient from the system.

Soil Characteristics

The onsite system was installed in soils similar in characteristics to the Seabrook (Mixed, thermic Aquic Udipsamments) and Tarboro (Mixed, thermic Typic Udipsamments) soil series. Seabrook soils are characterized as very sandy and moderately well drained with rapid permeability (15–50 cm/hour), low cation exchange capacity (<3 cmol/kg), and acidic conditions (pH between 4.5 and 6.5) (U.S. Department of Agriculture [USDA], 1995). Tarboro series are somewhat excessively drained, coarse-textured soils with rapid permeability (15–50 cm/hour) and acidic conditions (pH 5.1–6.5) (USDA, 1995).

Groundwater and Soil Water Monitoring Network

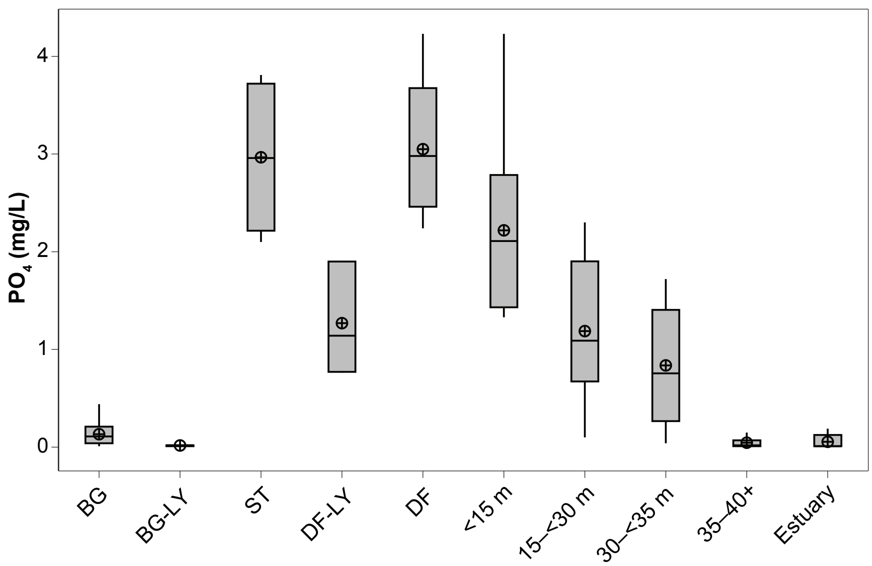
Multidepth, nested piezometers constructed of 5–10 cm diameter PVC pipe with 60 cm screen intervals were installed between drainfield trenches, and up- and down-gradient of the OWS flow paths (Figure 2). Hand augers were used to create boreholes to the desired depth, typically 1 m or so below the water table for the "deep" nested piezometer, and 0.4 m below the water table for the "shallow" nested piezometer. Piezometers were cut to the appropriate length and driven into boreholes. Sand was poured into the annular space around each piezometer until the screened section was covered. The remaining annular space was filled all the way to the surface with a mixture of sand and bentonite. Soil samples at depths beneath the drainfield trenches were collected and sent to the North Carolina Agronomic Services Division for descriptive analysis including pH and effective cation exchange capacity. Water table depths were measured manually every two months from the piezometers (five times overall) with a temperature level and conductivity (TLC) meter. The TLC meter consists of a water sensor at the end of a tape measure connected to a reel that allows for quick, manual determination of water table depths. The meter was calibrated before each use. Lysimeters were installed in nests in the unsaturated zone (above water table) at 90, 120, and 150 cm below the surface near background and drainfield piezometers (Figure 2). Lysimeters were used to collect soil water samples.

Sampling Procedures

Septic tank, lysimeters, piezometers, and the estuary were sampled every two months during February–October 2011 (five times). A hand vacuum pump was used to pull soil water from the tension lysimeters and then into sample bottles. A new disposable bailer was used to collect groundwater samples from each piezometer. Septic tank samples were collected using a peristaltic pump connected to rigid tubing inserted into the clear zone of the tank through access manholes. A sample bottle was lowered in the estuary to collect the surface water samples. Groundwater, estuary, and septic tank samples were analyzed for pH, electrical conductivity, dissolved oxygen (DO), and temperature using a multimeter. The multimeter allowed for quick field analysis of most environmental parameters and was calibrated before each sampling event. Water and septic tank samples were kept on ice in a cooler and transported to the East

FIGURE 3

Mean Phosphate Concentrations at the Research Site



BG = background groundwater; BG-LY = background soil water; ST = septic tank effluent; DF-LY = soil water between drainfield trenches; and DF = groundwater between drainfield trenches. The numbers 15, 30, 35, and 40+ refer to distances down-gradient from the system.

Carolina University Central Environmental Laboratory (CEL) within 12 hours for phosphate analyses. Graduate students assisted with sample collection, transport, and laboratory analysis to maintain the chain of custody. Water samples were filtered at the CEL. Samples were analyzed for phosphate using the U.S. Environmental Protection Agency–approved Smart Chem 200 method.

Statistical Comparison Groups

Septic tank effluent phosphate concentrations were compared to groundwater phosphate between the OWS trenches to assess the effectiveness of OWS in reducing phosphate concentrations before discharge to groundwater. Groundwater phosphate concentrations between the drainfield trenches were compared to background groundwater to evaluate the effects of OWS on shallow groundwater. Soil water phosphate concentrations from background lysimeters were compared to samples collected between drainfield trenches to determine OWS influence on soil water. North Carolina regulations dictate that OWS must be located at least 15–30 m+ from surface waters depending on the surface water classification (15A NCAC 18A .1950d). Down-gradient groundwater phosphate concentrations ≤15 m from the OWS were compared to those between 15 and 30 m. Down-gradient groundwater samples 30–35 m from the system were compared to groundwater samples collected at 35–40 m+ and the estuary. Mann-Whitney tests (Davis, 2002) were performed using Minitab 16 statistical software to determine if significant differences in phosphate concentrations existed between comparison groups. A linear regression of groundwater phosphate concentrations plotted against distance down-gradient from the OWS was used to determine the relationship between setback distance and groundwater phosphate concentrations.

Results

Physical and Chemical Parameters of Water

Groundwater levels beneath the OWS fluctuated by nearly 1 m during our study. The lowest level observed during sampling was in early August 2011 (1.98 m below surface), while the highest water levels occurred in October 2011 (0.98 m below surface). Groundwater flow direction was predominantly toward the south east (1p4 towards 1p16) (Figure 1). Groundwater samples collected from piezometers 1p4 to 1p10, 1p13, and 1p16 were considered to be within the OWS plume and were used to determine concentration reductions with distance from the system.

Mean water sample temperatures, electrical conductivity, and pH levels were highest for septic effluent (20.4 ± 5.9°C, 1,314 ± 95 µS/cm, and 7.3 ± 0.3), and were elevated in groundwater between drainfield trenches (17.1 ± 3.8°C, 809 ± 213 µS/cm, and 6.6 ± 0.4) relative to adjacent background groundwater values (16.4 ± 4.9°C, 86 ± 213 µS/cm, and 6.3 ± 0.8). These data confirm that wastewater was influencing the physical and chemical properties of groundwater adjacent to the OWS. The highest mean DO concentrations (3.3 ± 1.1 mg/L) were in the background groundwater piezometers. Mean DO concentrations beneath the drainfield trenches were 3.2 ± 0.4 mg/L, indicating aerobic conditions. Septic effluent had the lowest DO levels (0.3 ± 0.1 mg/L). The soil pH beneath the drainfield trenches (6.9) was lower than septic effluent pH (7.3), but higher than groundwater between disposal field trenches (6.7), indicating the influence of wastewater on soil chemical properties.

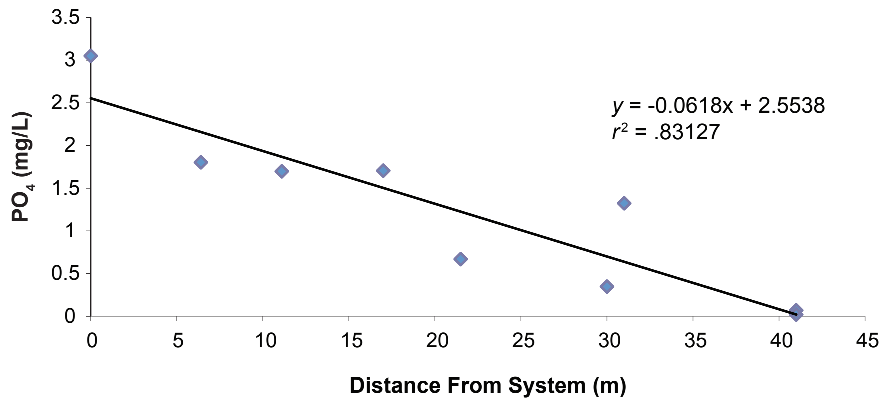
Phosphate Analysis

Average septic tank effluent phosphate concentrations (2.97 ± 0.76 mg/L) were within the range of effluent concentrations reported in other studies (Corbett et al., 2002; Humphrey & O'Driscoll, 2011; Robertson et al., 1998). Groundwater phosphate concentrations between the drainfield trenches (3.05 ± 0.74 mg/L) were not significantly different at $p < .05$ than septic effluent concentrations and were elevated relative to background conditions (0.14 ± 0.12 mg/L) (Figure 3). Soil water phosphate concentrations in background lysimeters (0.02 ± 0.01 mg/L) were significantly lower than soil water phosphate concentrations between drainfield trenches (1.27 ± 0.58 mg/L) as shown in Figure 3. Groundwater phosphate concentrations within the plume (1p4 towards 1p16) typically decreased with distance from the OWS. For example, groundwater within 15 m of the OWS had mean phosphate concentrations of 2.22 ± 0.85 mg/L, while mean groundwater phosphate between 15 and 30 m from the system was 1.19 ± 0.76 mg/L; groundwater between 30 and 35 m from the system contained 0.84 ± 0.59 mg/L phosphate; and groundwater more than 35 m from the system had 0.05 ± 0.05 mg/L phosphate (Figure 3). The estuary had mean phosphate concentrations of 0.06 ± 0.08 mg/L (Figure 3).

Groundwater phosphate concentrations showed an inverse relation (r^2 = .83) to distance from the OWS (Figure 4). Based on the regression equation ($y = -0.0618x + 2.5538$) and mean background phosphate concentration (0.14 mg/L), a 39 m setback distance would be required to reduce OWS-derived phosphate concentrations to background levels.

FIGURE **4**

Regression of Phosphate Concentrations and Distance From the System



Discussion

Onsite System Phosphate Attenuation

The dominant phosphate removal processes in soils beneath OWS are mineral precipitation and adsorption (Robertson et al., 1998). Precipitation of the minerals strengite or variscite would be possible at this site during the study period given the near-neutral pH of soil (6.9) and groundwater beneath the drainfield (6.7) as well as aerobic conditions (DO: mean 3.2 mg/L) (Robertson et al., 1998). Vivianite ($Fe_3[PO_4]_2{*}8H_2O$) is another iron-based mineral that could precipitate at this site. Given the soil and groundwater pH, vivianite formation would be possible if anaerobic conditions occurred between the drainfield and estuary and if phosphate and iron were available (Ptacek, 1998). Mean phosphate concentrations in groundwater beneath the OWS were not significantly different, however, than septic effluent and groundwater phosphate concentrations more than 30 m down-gradient from the system and were still elevated relative to background levels. These data indicate that phosphate attenuation processes were not sufficient to prevent groundwater enrichment of phosphate at distances greater than the minimum setbacks from OWS to surface waters (15–30 m depending on water classification).

The OWS in our study was installed in sandy, permeable, low-reactive surface area Tarboro and Seabrook soil series. The OWS had been in use for more than 25 years. It is possible that the phosphate adsorptive capacity of the sandy soils was exceeded over the past few decades allowing for significant phosphate transport more than 30 m down-gradient. It is also possible that phosphate minerals did not form because not enough iron was available for precipitation of variscite, strengite, or vivianite. The Seabrook soils at the research site were located near the front of the drainfield trenches and between the OWS and the estuary. The Seabrook series soils in Beaufort County, North Carolina, typically have a light gray color (chroma less than 2) starting at depths 1 m below the surface (USDA, 1995). The gray colors indicate a lack of iron-coated sand grains and potential iron deficiency at that depth (Richardson & Vepraskus, 2001). Even if phosphate is abundant and pH and redox potential are conducive to formation of variscite, strengite, or vivianite, precipitation is unlikely if iron is not present.

Onsite System Setback Distances

The OWS was contributing elevated concentrations of phosphate to soil water and shallow groundwater beneath and down-gradient from the system. While groundwater phosphate concentrations decreased with increasing distance from the system, elevated concentrations (0.84 ± 0.59) were still discovered more than 30 m down-gradient. Research conducted in various areas of Canada (Robertson et al., 1998) and St. George Island, Florida (Corbett et al., 2002) also found elevated phosphorous concentrations more than 30 m down-gradient from OWS installed in sandy, coastal soils. For our site, it was determined that a 39 m setback would be necessary to reduce phosphate concentrations to background levels. Thus, 30 m may not be a sufficient setback distance from OWS to surface waters in some settings.

Conclusion

Our study has shown that OWS installed in some sandy coastal environments can contribute significant phosphate to shallow groundwater more than 30 m down-gradient from the OWS. The OWS in our study was located more than 40 m from the estuary and thus phosphate concentrations were reduced to background levels before reaching surface waters. In eastern North Carolina and other coastal regions, setback distances from OWS may need to be increased, however, to protect nutrient-sensitive waters or more advanced technologies employed to reduce phosphate transport in some areas.

Acknowledgements: The authors would like to thank Dr. Max Zarate-Bermudez for facilitating the project, and the Centers for Disease Control and Prevention and NEHA for funding and assisting with the project. We would also like to acknowledge the efforts of several East Carolina University students including Shawn Thieme, Keaton Henry, Rob Howard, Sarah Hardison, Katie Supler, and others.

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Place-Based Exposure and Cataract Risk in the Beaver Dam Cohort

Prepublished online January/February 2013, National Environmental Health Association.

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Abstract **According to the Centers for Disease Control and Prevention, with an aging U.S. population an estimated 30 million people will be diagnosed with cataract by 2020. Several modifiable risk factors have been identified for nuclear cataract, cortical cataract, and posterior subcapsular cataract (PSC), including smoking, diabetes, and steroid medications. In the study described here, the authors evaluated residential location as a potential proxy of risk factors for cataracts in the Beaver Dam Eye Study cohort established in 1987. Cataract risk was calculated using general estimating equation modeling to account for correlation between eyes. Fifteen-year cumulative incidence rates were calculated for each type of cataract by eye. Of the 4,926 study participants, 3,253 seen at the baseline examination were included in the analyses. Compared to urban residents, the odds ratio (95% confidence interval) for rural participants' risk of cortical, nuclear, and PSC was 0.92 (0.73, 1.16), 0.85 (0.69, 1.06), and 0.71 (0.48, 1.05), respectively, adjusting for age, sex, educational status, and smoking status. The lowest cumulative incidences were for those living in rural areas, compared to edge or urban areas for all three types of cataracts.**

Introduction

Cataracts are common concomitants of aging, and restoration of vision after the effects of cataracts usually requires surgery. The cost of cataract surgery and related doctor visits to Medicare in 2010 was budgeted at $3.2 billion (Lane & Aggarwala, 2010). While cataract surgery is effective, complications of such surgery exist. As longevity increases, the number of persons having cataract surgery is likely to rise. Thus, diagnosis of cataracts imposes a health burden to the individuals involved and to health care costs for society.

Three types of age-related cataracts include nuclear, cortical, and posterior subcapsular. While the most important risk factor for each type of cataract is age, evidence suggests different factors may increase risk, and those factors appear to differ among the three types. For example, smoking is associated with increased risk of nuclear cataract, diabetes with cortical cataract, and steroid medications with posterior subcapsular cataract (PSC). Conversely, statin medications appear to decrease the risk of nuclear cataract (Klein, Klein, Lee, & Grady, 2006).

Several environmental exposures have been associated with eye diseases. For example, high-dose whole body radiation exposure is associated with cataracts (Blakely et al., 2010; Little, 2009), occupational exposure to ionizing radiation is associated with posterior lens changes (Ciraj-Bjelac et al., 2010), and the more conventional exposure to environmental UV-B in ambient light is associated with cortical cataracts (Cruickshanks, Klein, & Klein, 1992). Other environmental exposures, such as lead, gold, copper, and heavy metals have been hypothesized to increase the risk of developing cataracts (Ernst, Baltzan, Deschenes, & Suissa, 2006; Schaumberg et al., 2004).

Efforts to identify modifiable risk factors continue. In the Beaver Dam cohort, data were available for the established risk factors for cataracts as well as geocoded residential locations, which allowed us to characterize the cohort by location. In addition, exploratory spatial analyses utilized environmental monitoring data of rural well water to evaluate cataract risk. We hypothesized that rural living may be associated with the development of specific cataract types in older adults.

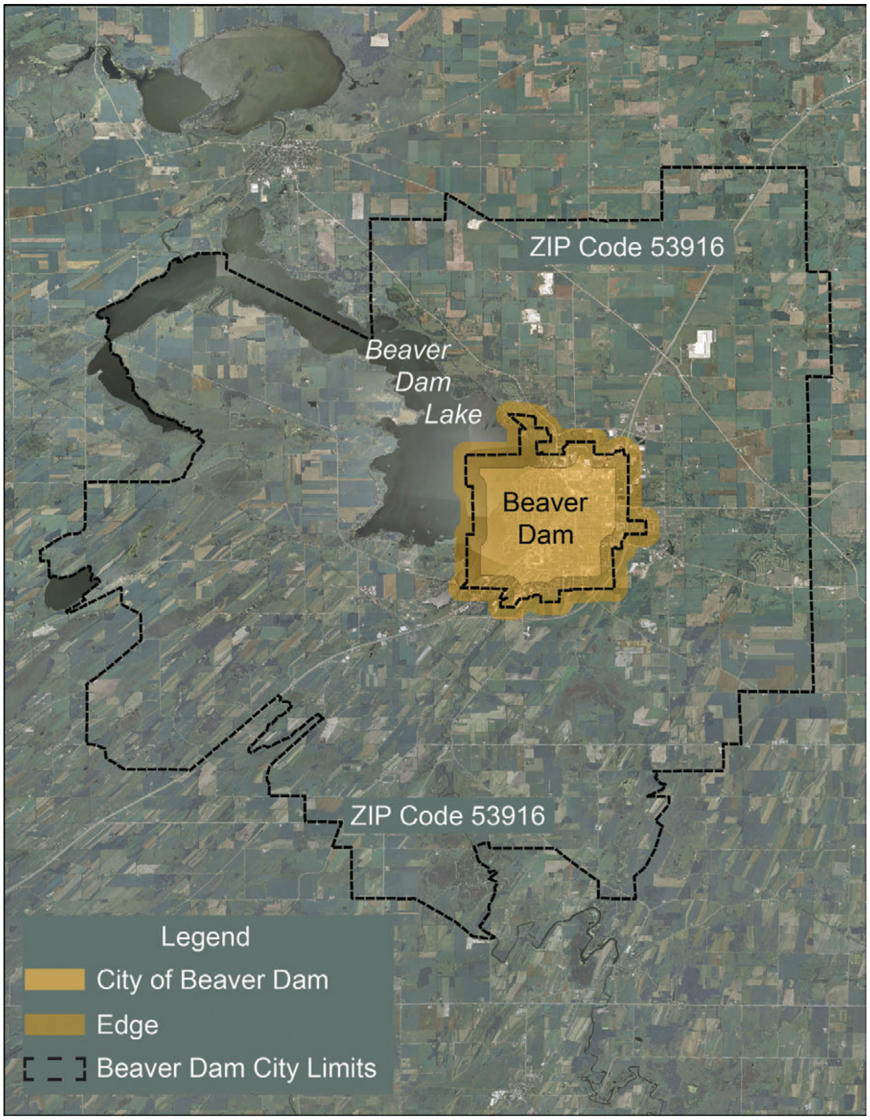
Methods

Population

Methods used to identify and describe the population have appeared in detail

FIGURE **1**

Orthophoto Showing Boundaries of Beaver Dam ZIP Code 53916



Also includes edge (1/4 mile inside and outside) of Beaver Dam city boundary and city of Beaver Dam.

in previous reports (Klein, Klein, & Lee, 1996; Klein, Klein, Lee, Cruickshanks, & Chappell, 2001; Klein, Klein, Lee, Cruickshanks, & Gangnon, 2006; Klein, Klein, Linton, & De Mets, 1991; Linton, Klein, & Klein, 1991). In brief, a private census of the population of Beaver Dam, Wisconsin (99% white), was performed from fall 1987 to spring 1988 (Linton et al., 1991). Of the 5,924 enumerated persons 43–84 years of age, 4,926 participated in the baseline examination in 1988 to 1990. In the three follow-up examinations, 3,722, 2,962, and 2,375 persons participated in the 5, 10, and 15-year follow-up examinations, respectively (Klein et al., 1996; Klein et al., 2001; Klein et al., 2006; Klein et al., 1991). Tenets of the Declaration of Helsinki were followed, institutional human experimentation committee approval from the University of Wisconsin was granted, and informed consent was obtained from all subjects in the form of a signature verifying they had read an explanation of procedures and policies and agreed to them. During the study visit, standard measurements were made and a codified questionnaire was administered.

Photographs of the lenses were taken with two different cameras: a slit-lamp camera and a retroillumination camera (Klein, Klein, Linton, Magli, & Neider, 1990). The pupil diameter at the time of the baseline photographs was recorded on the examination form for each of the two subsequent examinations. The grading procedures for the lens were based on detailed codified decision rules (Klein et al., 1990). Graders were masked to subject identity and personal characteristics. Scores for nuclear sclerosis were based on comparisons with standard photographs. The scale has five levels of severity based on opacity of the nucleus, with 1 indicating no opacity and 5 indicating a great amount of opacity. Levels 4 and 5 were considered to be cases of nuclear cataract in this and previous publications of prevalence data (Klein, Klein, & Linton, 1992). Scores for cortical cataract and PSC were based on weighted estimates of the degree of opacity of the lens area, as defined by a circular grid, divided into eight pie-wedge shaped peripheral areas and a central circular area overlaid on the photograph (Klein et al., 1990). Cases of cortical cataract were those with opacity of 5% or more of the lens surface. PSC opacity was defined as 5% or more of a grid segment. The classification of cataract types corresponded to a lens opacity of sufficient severity that a clinical ophthalmologist would label it as a cataract (Klein et al., 1990). The estimates of incidence were based on all persons having corresponding gradable subfields at all visits.

Geocoding Participants' Location

Participants' street mailing addresses at the baseline examination were assigned latitude and longitude coordinates (i.e., geocodes) to the address point location with an 80% spelling and 80% overall sensitivity score

TABLE **1**

Participant Characteristics by Urban/Edge/Rural Residential Status in the Beaver Dam Eye Study

Characteristics	Urban (n = 2263)	Edge (n = 540)	p-Value	p-Value[a]	Rural (n= 450)	p-Value	p-Value[a]
	Crude %	Crude %			Crude %		
Age (years)							
43–54	35.7	32.41	.42	.43	46.89	<.001	<.001
55–64	29.21	31.67			30.89		
65–74	26.38	27.59			17.78		
≥75	8.71	8.33			4.44		
Alcohol use							
No	12.46	12.41	.97	.9	11.78	.69	.9
Yes	87.54	87.59			88.22		
Comorbidities[b]							
No	71.91	71.7	.92	.97	76.92	.03	.38
Yes	28.09	28.3			23.08		
Hypertension[c]							
No	51.37	54.63	.17	.12	56.89	.03	.33
Yes	48.63	45.37			43.11		
Income ($/year)							
≤29,000	60.4	57.5	.22	.13	53.69	.009	.67
≥30,000	39.6	42.5			46.31		
Education							
<High school	20.88	31.3	<.001	<.001	25.33	.003	<.001
High school	46.84	44.63			47.78		
College	15.57	12.96			15.33		
>College	16.72	11.11			11.56		
Sex							
Female	56.12	57.04	.7	.74	51.11	.05	.09
Male	43.88	42.96			48.89		
Smoking status							
Never	43.94	44.81	.73	.88	50.22	.08	.002
Past	36.78	36.3			31.11		
Current	19.27	18.89			18.67		
Steroid use							
No	95.75	95.42	.74	.76	96.56	.44	.53
Yes	4.25	4.58			3.44		
Visual acuity[d]							
Better than 20/40	98.13	97.59	.47	.55	99.55	.03	.2
20/40–20/160	1.69	2.22			0.45		
20/200 and worse	0.18	0.19			0.00		
Sedentary lifestyle							
No	27.93	24.44	.1	.11	18.44	<.001	<.001
Yes	72.07	75.56			81.56		
Sunlight exposure							
Unexposed	74.46	75.93	.48	.48	79.15	.036	.036
Exposed	25.54	24.07			20.85		

[a]Adjusted for age and gender.
[b]Includes cancer, diabetes, and cardiovascular disease.
[c]Defined as systolic blood pressure ≥140 mmHg and/or diastolic blood pressure ≥90 mmHg or use of antihypertensive medication.
[d]Best corrected visual acuity in the better eye.

using ArcView GIS 3.2. For unmatched addresses using street address, the nine digit ZIP code line segment centroid was used as the geocode. For the remaining unmatched addresses, the 1990 ZIP code centroid was used.

To examine location as a risk factor, the authors assigned participants who lived in the ZIP code of 53916 (Beaver Dam, Wisconsin) at baseline an edge, urban, or rural classification. "Edge" was defined as living within a buffer zone, i.e., within a quarter mile of either side of the Beaver Dam incorporated boundary in 1990 (U.S. Census Bureau, 2005). "Urban" was defined as living within the Beaver Dam incorporated area in 1990 but not within the buffer zone. "Rural" was defined as not living in either the Beaver Dam incorporated area or the buffer zone in 1990 but within the ZIP code of 53916 (Figure 1). For exploratory analysis of rural participants' exposure to nitrate-nitrogen exposure from drinking private well water, the authors reclassified the geocoded locations as living outside of the incorporated area of Beaver Dam (rural) or not.

Nitrate-Nitrogen Well Water Data

We obtained publicly available data on nitrate-nitrogen contamination of groundwater from the Wisconsin Department of Agriculture, Trade and Consumer Protection (WDATCP). As part of the Atrazine Rule Evaluation Study, WDATCP randomly sampled 289 private wells using a stratified random sampling procedure to analyze the groundwater for various herbicides and nitrate-nitrogen in 1994 (Baldock, 1993; LeMasters & Baldock, 1997; Vanden Brook et al., 2002). These samples were analyzed using gas chromatography for nitrate-nitrogen by the WDATCP's Bureau of Laboratory Science. Wisconsin residents living outside of incorporated city or village boundaries rely on private wells located near their residences for their drinking water.

Estimation Approach

Natural neighbor interpolation (Sibson, 1981) was used to estimate nitrate-nitrogen levels in groundwater across the entire state. Natural neighbor interpolation uses a weighted moving average of concentrations of nitrate-nitrogen residues in residential drinking water in surrounding or neighboring observed wells. Neighboring points and the corresponding weights are based on the Voronoi diagram of the data points (Okabe, 2000). The Voronoi diagram of a set of points is a partitioning of the plane into regions associated with each point such that every point in a given partition is closer to the generating point than any other point. This interpolation was performed using ArcGIS 9.3.

Statistical Methods

Variables used in the analyses were age at examination; educational status (four categories: less than high school graduate, high school graduate or GED, some college or baccalaureate degree, and graduate or professional school [e.g., law, medicine]); income (dichotomous, defined as reported annual household income of ≤$29,000 or ≥$30,000); history of comorbitidities (cardiovascular disease, cancer, and diabetes; all were dichotomous); reported drinking alcohol at least one time in the previous year (dichotomous); steroid use (dichotomous; defined as currently taking any steroid medication); and smoking status (three categories: current, former, and never). A never smoker was defined as someone who had smoked less than 100 cigarettes in his or her lifetime; a former smoker had smoked at least 100 cigarettes in his or her lifetime but was not currently smoking every day or some days; a current smoker was defined as smoking at least 100 cigarettes in his or her lifetime and currently smoking every day or some days. Level of physical activity was also included in the analyses; a sedentary lifestyle was defined as engaging in physical activity that caused sweating fewer than three times per week, and an active lifestyle was defined as engaging in physical activity that caused sweating three or more times per week.

Incidence of cataract was calculated separately for each eye for each type of age-related cataract, taking into account the competing risk of death and cataract surgery. For the incidence of a particular cataract type, the population at risk included all eyes free of that cataract type and cataract surgery at baseline. The 15-year cumulative incidence was calculated for each eye separately, and was reported for the right eye only (Table 2) (Kaplan, 1958). Odds ratios (*OR*), confidence intervals (*CI*), and *p*-values were modeled by incorporating data from both eyes, using general estimating equation (GEE) techniques to account for correlation between the eyes (Table 2) (Hosmer & Lemeshow, 1989). SAS version 9 was used for all analyses (Klein, Klein, & Moss, 1997).

Eligibility Criteria

Those eyes whose lenses were removed due to trauma or in conjunction with ocular surgery unrelated to cataract were excluded. Other specific characteristics caused an eye to be excluded from the calculation of incidence of an age-related cataract; these included intraocular surgery, invasive intraocular trauma, confounding lens lesions in the photographs, absence of a photograph, or ungradable photograph at the baseline or follow-up examination. Analyses are based on those persons who lived within the ZIP code of 53916 (Beaver Dam) at the time of the baseline examination and had a geocode match to the address or nine-digit ZIP code.

Eligible Participants

Of the 3,684 people seen at baseline and then at follow-up, 48 were excluded for residing outside of the 53916 ZIP code and six more were excluded because they had geocodes of their ZIP code centroid. In other words, for these six participants, their specific location within Beaver Dam ZIP code could not be ascertained; they could have lived within the city limits, in the edge, or in the rural part of the ZIP code. Of the remaining 3,630 eligible participants, 268 people were excluded from all analyses for right eye cataract and 254 were excluded from all analyses for left eye cataract due to surgery or trauma. An additional 217 were excluded from right eye analyses and 204 were excluded from left eye analyses due to missing or ungradable photos (including those with confounding lesions). Overall, a total of 377 participants were excluded from all analyses for both eyes, and 3,253 participants contributed to GEE modeling for at least one outcome.

Results

Characteristics of all participants in the Beaver Dam Eye Study cohort were similar to the participants contributing to these study analyses (data not shown). Rural residents tended to be statistically younger, male, less educated, more sedentary, and never smokers compared to urban residents, and edge residents tended to be less educated than urban residents (Table 1).

The cumulative incidence of each cataract endpoint by urban/rural/edge home location indicates slight differences in cumulative incidence among the three cataract types (Table 2). Within each type of cataract, compared to those living in urban areas, the lowest incidences were for those living in rural areas and then those living in the edge areas. Compared to urban residents, the *OR* (95% *CI*) for rural participants' risk of cortical cataract, nuclear cataract, and PSC were 0.92 (0.73, 1.16), 0.85 (0.69, 1.06), and 0.71 (0.48, 1.05), respectively, controlling for age, sex, smoking status, and educational status (Table 2). Adding other important covariates listed in Table 1 as well as duration of residence from baseline by location did not alter the results. Little change occurred in *OR*s when we included the number of years living in the same location.

To further examine potential effects of place of residence on cataract risk, we evaluated rural residents ($n = 640$) who lived outside of the Beaver Dam city limits but within the Beaver Dam Zip code of 53916 and their relative exposure to nitrate-nitrogen in their drinking water. Compared to rural residents with nitrate-nitrogen exposure of less than 5 parts per million (ppm), rural residents whose well water had concentrations of nitrate-nitrogen of 10 ppm (*OR*; 95% *CI*) had higher odds of developing cortical cataract (1.37; 0.81, 2.31), nuclear cataract (0.97; 0.58, 1.61), and PSC (1.23; 0.50, 3.05), although the relationships were not statistically significant after adjusting for age, gender, educational status, and smoking (Table 3). No apparent effect occurred of quantity of water consumed or of use of a water filter in the home water system on these relationships (data not shown).

TABLE **2**

Cumulative Incidence in Right Eyes and Multivariable-Adjusted Odds Ratio of All Eyes for Cataract Type by Residential Location

	Right Eye Only		All Eligible Eyes		
Cataract Type	At Risk (*n*)	Cumulative Incidence (%)[a]	Adjusted[b] *OR*[c]	Adjusted[b] 95% *CI*[c]	Adjusted[b] *p*-Value
Cortical cataract					
Urban	1985	20.8	Ref		
Edge	472	20	0.94	0.77, 1.16	.59
Rural	414	16.3	0.92	0.73, 1.16	.47
Nuclear cataract					
Urban	2000	26.9	Ref		
Edge	462	27.1	0.97	0.80, 1.17	.74
Rural	409	19.1	0.85	0.69, 1.06	.16
PSC[c]					
Urban	2162	7.3	Ref		
Edge	514	7.6	1.05	0.78, 1.42	.74
Rural	432	4.6	0.71	0.48, 1.05	.09

[a]Accounts for competing risk of death.
[b]Adjusted for age, gender, education, and smoking status.
[c]*OR* = odds ratio; *CI* = confidence interval; PSC = posterior subcapsular cataract.

TABLE **3**

Cumulative Incidence in Right Eyes and Multivariable-Adjusted Odds Ratio of All Eyes for Cataract Type by Nitrate Levels[a] in Water Supply of Rural Residents

	Right Eye Only		All Eligible Eyes		
Cataract Type	At Risk (*n*)	Cumulative Incidence (%)[b]	Adjusted[c] *OR*[d]	Adjusted[c] 95% *CI*[d]	Adjusted[c] *p*-Value
Cortical cataract					
Low nitrate	261	15.3	Ref		
Mid nitrate	257	17.4	1.03	0.70, 1.51	.88
High nitrate	74	18.5	1.37	0.81, 2.31	.24
Nuclear cataract					
Low nitrate	256	20.7	Ref		
Mid nitrate	256	17.2	0.8	0.56, 1.15	.23
High nitrate	68	28.4	0.97	0.58, 1.61	.9
PSC[d]					
Low nitrate	271	3.7	Ref		
Mid nitrate	268	5.9	1.07	0.55, 2.08	.85
High nitrate	77	2.9	1.23	0.50, 3.05	.65

[a]Low nitrate level ≤4 parts per million (ppm); mid = 5–9 ppm; high ≥10 ppm.
[b]Accounts for competing risk of death.
[c]Adjusted for age, gender, education, and smoking status.
[d]*OR* = odds ratio; *CI* = confidence interval; PSC = posterior subcapsular cataract.

Discussion

In our study, the Beaver Dam cohort did not have an increased risk of cataracts if they lived in a rural environment, after adjusting for age, sex, educational status, and smoking status. Rather, they seem to have been at a reduced risk, though not statistically significant. A strength of our study is presence and type of cataract determined by objective grading of standard images from their cohort. Furthermore, data were prospectively collected on established risk factors for cataracts and adjusted for those that vary by urban, edge, or rural status.

Before we began these analyses, we hypothesized that differences might exist in cataract incidence related to location. Historically, the known disparity in health of individuals living in a rural environment compared to those living in an urban environment have documented differences in health care access and utilization, cost, and geographic variations in providers, specialists, and services (Hartley, 2004). With an increased use of multilevel modeling, environment-specific factors are also being studied to understand the differences in health outcomes between urban and rural residents (Verheij, 1996). Although virtually all support the concept that life in the country is different from life in town, the specifics related to these differences particularly as they pertain to environmental exposures have yet to be fully characterized. We could not find any literature that addressed variations in cataract risk within a small geographic area that related to rural and small city residents. In comparing cataract risk for those living in a rural setting to those living in a small city, we found a nonstatistically significant reduced risk for rural residents. We were unable to identify the factor or factors that led to this finding from data collected in our longitudinal study.

To evaluate location beyond categorizing residential location as urban, edge, or rural, we also explored the potential relationship of nitrate-nitrogen exposure from drinking water and cataract risk. The source of nitrate-nitrogen in drinking water is from applications of it along with other agricultural chemicals to agricultural fields (Vanden Brook et al., 2002). The Safe Water Drinking Act of 1996 sets the maximum allowable level of nitrate-nitrogen in public drinking water at 10 ppm. This level is based on an association of incidence of methemoglobinemia (blue baby syndrome) and exposure to nitrate-nitrogen in drinking water (Johnson & Kross, 1990). Unlike public drinking water, no regulations exist on private drinking water regarding testing or remediation of contaminants. In Wisconsin, the percentage of private wells with nitrate-nitrogen levels ≥10 ppm has remained steady at approximately 9% (Brandt et al., 2008). In the segment of the population that might have had significant exposure to agricultural chemicals (i.e., rural persons who use well water), measured by 10 ppm nitrate-nitrogen exposure in drinking water, the authors did not see a difference in relative risk of cataracts compared to those with low levels of nitrate-nitrogen exposure (<5 ppm). This finding provides some evidence that exposure to agricultural chemicals in drinking water may not influence cataract risk.

Inferences from our study should be drawn cautiously due to several limitations. Our study was performed in a relatively small city in the Midwestern U.S. Urban-rural differences in exposures in this setting are likely to differ from urban-rural differences in and around larger cities, so these data may not reflect differences elsewhere. The Beaver Dam cohort is virtually entirely of northern European ancestry; differences in ethnicity and genetic background may affect susceptibility to cataract risk factors and this may differ in urban and rural environments. We were not able to evaluate gene-environment interaction. Another limitation of our study was the relatively small sample size in which a small or modest effect size may not have been able to be detected. The nitrate-nitrogen exposure assessment used interpolated values from a relatively small number of randomly selected wells, which could not take into consideration any geological variation in water distribution. Finally, no data were collected on complete water consumption habits, including source of water. Therefore, only exposure to residential drinking water was evaluated.

Conclusion

These results suggest that further research of differences associated with environmental exposures in cataract incidence is necessary, as cataract is a common condition responsible for functional disability. In evaluating nitrate-nitrogen exposure through drinking water, results suggest that exposure to agricultural chemicals in rural drinking water was not a likely source of cataract risk. We were unable to identify any specific factor or factor(s) associated with a reduced cataract risk for rural residents compared to small city residents. Identifying modifiable risk factors may result in decreased incidence of this condition. This would provide a benefit to quality of life and to reduced medical costs.

Acknowledgements: We thank Stan Senger, environmental quality section supervisor at WDATCP for the private well water data. We also thank Michael Hardy, GIS specialist at the University of Missouri Center for Applied Research and Environmental Systems for his assistance with the orthophotograph of Beaver Dam and GIS. This work was supported by a grant from the National Eye Institute, National Institutes of Health (EY05694); and, in part, by Research to Prevent Blindness (R. Klein and B.E.K. Klein, Senior Scientific Investigator Awards). The content of this article is solely the responsibility of the authors and does not necessarily reflect the official views of the National Eye Institute or the National Institutes of Health.

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References on page 40

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Residential Radon Testing Intentions, Perceived Radon Severity, and Tobacco Use

Prepublished online January/February 2013, National Environmental Health Association.

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Abstract **Kentucky homeowners requesting radon test kits through the Kentucky Radon Program and randomly selected homeowners (*N* = 129) completed a survey assessing factors related to their radon testing intentions and perceived severity of radon exposure, including social influence, perceived susceptibility, synergistic risk perception, and tobacco use. Perceived severity, social influence, and current smoking were the strongest predictors of radon testing intentions. Those with higher perceived severity were nearly eight times more likely to plan to test. Perceived severity was highest among females and those rating combined radon and tobacco smoke exposure as much riskier than tobacco smoke alone. Knowing someone who had tested for radon was associated with seven times greater likelihood of planning to test for radon. Current smokers were over six times more likely to plan to test than nonsmokers. The findings have implications for targeting interventions to improve residential radon testing and decrease lung cancer risk.**

Introduction

Radon is one of four top environmental risks to public health (Reuben, 2010). Exposure to radon, a naturally occurring gas derived from the decomposition of uranium in the ground, is the second-leading cause of lung cancer and is associated with an estimated 15,400 to 21,800 cases of lung cancer in the U.S. each year (Committee on Health Risks of Exposure to Radon, 1999). Reducing the risk of radon exposure in homes with and without basements requires testing for radon and installing radon mitigation systems if radon levels are high (U.S. Environmental Protection Agency [U.S. EPA], 2012). Residential radon levels below 4 picocuries per liter (pCi/L), equivalent to 148 becquerels per cubic meter (Bq/m^3), are recommended by U.S. EPA (2012). The World Health Organization (WHO, 2009) recommends national radon reference levels of 2.7 pCi/L (100 Bq/m^3).

Public awareness of radon is high in the U.S. and Canada, with 52%–82% reporting having heard of radon, but only 8%–15% reporting testing or considering radon testing (Environics Research Group, 2007; Gregory & Jalbert, 2004; Halpern & Warner, 1994; Wang, Ju, Stark, & Teresi, 2000). The gap between radon awareness and testing presents a challenge to public health professionals.

Correlates of intentions to test for radon include perceived community radon risk, perceived susceptibility, perceived severity, perceived community concern, and number of known radon testers (Weinstein, Sandman, & Roberts, 1991). Other factors positively associated with radon testing include education (Wang et al., 2000); income (Halpern & Warner, 1994; Hill, Butterfield, & Larsson, 2006); female gender (Halpern & Warner, 1994); presence of children living in the home (DiPofi, LaTour, & Henthorne, 2001); home ownership (Hill et al., 2006); and younger age (Halpern & Warner, 1994; Wang et al., 2000).

Synergistic risk perception, one's assessment of risk from combined exposure to radon and tobacco smoke, may also influence radon risk reduction behavior (Hampson, Andrews, Barckley, Lichtenstein, & Lee, 2000; Lichtenstein et al., 2008). Although radon is a risk for both smokers and nonsmokers, those who have smoked and have had radon exposure have a higher risk of developing lung cancer because of multiplicative interaction between tobacco smoke and radon on lung cancer risk (Committee on Health Risks of Exposure to Radon, 1999). Despite smokers' increased radon-related risk, they may be less concerned about radon than nonsmokers

(Kennedy, Probart, & Dorman, 1991; Mainous & Hagen, 1993).

Reduction of radon-related lung cancer risk is particularly relevant in Kentucky, which leads the nation in lung cancer incidence (125.1 in Kentucky versus 79.5 per 100,000 nationally; U.S. Cancer Statistics Working Group, 2012); has historically been a national leader in smoking (24.8% versus 17.3% nationally; Centers for Disease Control and Prevention [CDC], 2010); and has many counties with residential radon levels above the U.S. EPA action level (Western Kentucky University, 2009). The Kentucky Radon Program partners with local health departments, the University of Kentucky Clean Indoor Air Partnership, and Western Kentucky University Environmental Health and Safety for radon awareness activities and the distribution of free radon test kits. Despite these educational efforts and high homeownership (69.9% versus 66.6% nationally; U.S. Census Bureau, 2012), fewer than 1% of Kentucky homes have been tested or mitigated (C. Hardwick, personal communication, June 29, 2012).

The purpose of our study was to investigate whether perceived severity, perceived susceptibility, synergistic risk perception, social influence, and smoking status are associated with intention to test for radon, controlling for age, gender, and education. A secondary purpose was to examine factors related to perceived radon severity. We hypothesized that those with higher levels of perceived severity, perceived susceptibility, synergistic risk perception, social influence, and those with a nonsmoking history would be more likely to plan radon testing. Further, we hypothesized that perceived susceptibility, synergistic risk perception, social influence, and nonsmoking history would predict perceived radon severity.

Methods

A cross-sectional, nonexperimental design using survey methods assessed intention to test for radon among two groups of Kentucky homeowners. Group I was a stratified random sample of 40 property owners from each of five Kentucky counties selected via public access property rolls (n = 200). The five counties were selected to ensure variability in a) prevalence of radon testing; b) rural versus urban location; c) region; and d) lung cancer incidence, percentage of adult smokers, and average radon values. A radon test request coupon was attached to each survey. Group II was a convenience sample of 143 individuals who had requested radon test kits from the Kentucky Radon Program between January and May 2009. Of the 343 surveys mailed, 129 were returned (overall response rate = 38%). Of those who responded, 56 (43%) were from Group I (response rate = 28%), and 73 (57%) were from Group II (response rate = 51%).

Measures

The 33-item survey consisted of categorical and multiple-choice questions. The outcome, radon testing intention, was measured by asking, "What are your thoughts about testing for radon (Weinstein et al., 1991)?" Responses were dichotomized into those with testing intentions and those without.

Perceived severity was measured by asking, "How serious would an illness caused by radon be (Weinstein et al., 1991)?" Responses were dichotomized into those who perceived illness caused by radon to be "serious" or "very serious" versus "somewhat serious" or "not serious at all." Social influence was assessed by asking, "How many people do you know who have tested for radon (Weinstein et al., 1991)?"

Perceived susceptibility was measured by the sum of two items from a three-item risk perception scale (Weinstein, Lyon, Sandman, & Cuite, 1998). The two items were, "What do you believe is the likelihood of finding radon in the place you live (1 = very unlikely; 5 = very likely)?" and "What do you think is the approximate percentage of homes in your area that have radon problems (1 = less than 10%; 5 = greater than 90%)?"

Synergistic risk perception was measured by a single question modeled after a relative risk measure previously developed (Hampson, Andrews, Lee, Lichtenstein, & Barckley, 2000).

Respondents were asked to rate the risk of smoking and radon combined compared to the risk of smoking alone using a five-point scale: 1 = much less risky; 5 = much more risky.

Smoking status was assessed using two questions: (1) "Have you smoked at least 100 cigarettes in your entire life (yes/no)?" and (2) "Do you now smoke cigarettes every day, some days, or not at all (CDC, 2007)?" Current smokers were those who had smoked at least 100 cigarettes and currently smoked every day or some days.

Procedures

Institutional review board approval was obtained from the Kentucky Cabinet for Health and Family Services and from the University of Kentucky. Surveys were mailed in two batches: (1) surveys and coupons for a free radon test kit were mailed to Group I (n = 200); and (2) survey packets were included with radon test kits and mailed to Group II (n = 143). Survey methods suggested by Dillman and co-authors (2009), including attaching a two dollar bill to each survey and sending a reminder post card approximately 10 days after the survey, were used to increase response rates.

Data Analysis

Descriptive statistics including frequency distributions or means and standard deviations were used to summarize the data. These univariate analyses were carried out for the whole sample, subgroups (i.e., convenience and random samples), and outcomes (i.e., those with and without testing intentions). The Rao-Scott Chi-square test of association (for nominal explanatory variables and covariates), Mann-Whitney U test (for ordinal variables), or two-sample t-test (for continuous variables) was used to determine whether study variables were associated with intention to test for radon and perceived severity of illness from radon. Multivariate logistic regression determined predictors of intention to test. Data analysis was conducted using SAS version 9.2 with an alpha level of .05 throughout. Given the data were obtained using a complex survey design, SAS procedures appropriate for this type of design, including SURVEYFREQ, SURVEYMEANS, and SURVEYLOGISTIC, were used.

Results

Sample Characteristics

Participants lived in 17 counties, with 47% residing in Jefferson County (the largest urban area in Kentucky). The mean age was 52.4 (*SD* = 14.1) years. Most were female with an annual household income of at least $50,000, college graduates, Caucasians, and nonsmokers (Table 1). Over four in 10 participants had smoked in their lifetime, and 10% were current smokers. No significant differences were shown between the convenience and random samples (Groups I and II) in terms of smoking status, gender, income, education, race, or age. The majority (90%) reported radon awareness; half had heard of radon via television.

Associations With Intention to Test

Over half (57%) reported plans to test or having tested for radon; most (64%) who had heard of radon reported intent to test. A significant group difference was shown in education level (Table 1). No differences occurred in gender, income, race, smoking status, mean age, or length of time living in current residence between those who planned to test and those who did not.

Differences were shown between those with and without testing intentions in perceived severity (χ^2 = 11.88, p < .01); perceived susceptibility to radon exposure (z = -4.37, p < .001); and perception of radon and smoking as "much more risky" than smoking alone, versus all other responses (χ^2 = 6.85, p = .01). Over a third (36%) knew at least one person who had tested for radon. Those with testing intentions knew more households who had tested for radon than those without plans to test (t = -2.57, p = .01).

Predictors of Intention to Test

College graduates, current smokers, those with higher perceived susceptibility and perceived severity, and those who knew others who had tested for radon had higher odds of intent to test for radon (Table 2). Those who had graduated from college had 4.8 times higher odds of planning to test for radon than those who did not plan to test. Current smoking was associated with over six times the odds of planning to test for radon. The perception of radon-related illness as being serious or very serious was associated with nearly eight times higher odds of planning to test for radon. No association occurred between radon testing intentions and synergistic risk perception, gender, or age.

TABLE **1**

Descriptive Statistics and Group Comparisons Between Those Who Planned to Test and Those Who Did Not (*N* = 129)

Characteristic	Total[a] *n* (%)	Planning to Test (*n* = 74) *n* (%)	Not Planning to Test (*n* = 55) *n* (%)	Test Statistic[b]
Gender				χ^2 = 0.06
Female	80 (62.0)	47 (63.5)	33 (60.0)	
Male	46 (35.7)	26 (35.1)	20 (36.4)	
Annual household income				χ^2 = 1.60
<$50,000	34 (26.4)	17 (23.0)	17 (30.9)	
≥$50,000	83 (64.3)	52 (70.3)	31 (56.4)	
Education level				χ^2 = 10.61*
<College graduate	59 (45.7)	25 (33.8)	34 (61.8)	
≥College graduate	66 (51.2)	47 (63.5)	19 (34.5)	
Race				–
Caucasian	124 (96.1)	71 (95.9)	53 (96.4)	
Minority	2 (1.6)	2 (2.7)	–	
Smoking status				
Current				χ^2 = 0.08
Yes	13 (10.1)	7 (9.5)	6 (10.9)	
No	114 (88.4)	66 (89.2)	48 (87.3)	
Former				χ^2 = 1.54
Yes	44 (34.1)	22 (29.7)	22 (40.0)	
No	83 (64.3)	51 (68.9)	32 (58.2)	
Never				χ^2 = 1.84
Yes	70 (54.3)	44 (59.5)	26 (47.3)	
No	57 (44.2)	29 (39.2)	28 (50.9)	
Current secondhand smoke exposure at home or work				χ^2 = 0.87
Yes	36 (27.9)	19 (25.7)	17 (30.9)	
No	84 (65.1)	52 (70.3)	32 (58.2)	

[a]Sum may not equal 129 due to missing values.
[b]Group comparisons based on Rao-Scott Chi-square test.
**p* < .05.

Associations With Perceived Severity of Illness From Radon

Synergistic risk perception and female gender were associated with perceived severity of illness from radon (χ^2 = 1.11, p = .01; and χ^2 = 8.84, p < .01, respectively). Having a household income over $50,000 per year was marginally associated with perceived severity (χ^2 = 3.18, p = .07). No differences were shown between groups in perceived susceptibility, knowing others who had tested for radon, age, education, or smoking status.

Predictors of Perceived Severity of Illness From Radon

Synergistic risk perception, gender, and income were predictive of perceived severity. For every one-point increase in synergistic risk perception, the odds of believing radon-related illness is serious increased by a factor of nearly six (Table 3). Females had nearly five times higher odds of reporting higher perceived severity. Although income over $50,000 was related to perceived severity, lack of income variability in the sample makes this finding less meaningful.

TABLE 2

Factors Associated With Radon Testing Intentions Among Kentucky Homeowners (*N* = 129)

Factor	Odds Ratio	95% Confidence Interval
Age	1.04	(0.99, 1.08)
Gender		
Female	1.76	(0.56, 5.53)
Male	*	
Education		
≥College graduate	**4.80**	(1.30, 17.76)
<College graduate	*	
Currently smoking		
Yes	**6.41**	(1.06, 38.67)
No	*	
Perceived susceptibility	**2.01**	(1.21, 3.33)
Synergistic risk perception		
Yes	3.35	(1.00, 11.26)
No	*	
Perceived severity		
Serious or very serious	**7.90**	(2.21, 28.28)
Not serious or somewhat serious	*	
Know others who have tested for radon		
Yes	**6.81**	(1.98, 23.39)
No	*	
Perceived community concern about radon		
Yes	3.20	(1.00, 10.24)
No	*	

Note. Bolded values are significant at the .05 level.
*Reference group.

Discussion

Perceived severity, social influence, and education level were positively associated with radon testing intentions, as hypothesized. Contrary to our hypothesis, current smoking was related to testing intentions. As hypothesized, synergistic risk perception was associated with higher perceived severity.

Perceived severity was most strongly predictive of testing intentions. Those who perceived that the illness caused by radon was serious or very serious were more than eight times more likely to plan to test for radon than those who thought it was not serious or somewhat serious. This finding is congruent with previous work demonstrating a significant correlation between perceived illness severity and radon testing intentions (Weinstein et al., 1991).

Synergistic risk perception and female gender were the strongest predictors of perceived severity. Those who rated the combined risk of radon and tobacco smoke exposure as much riskier than smoking alone were almost six times more likely to view radon as a serious health hazard. Females comprised a majority (62%) of the sample and were almost five times more likely to rate radon-related illness as serious. Previous research has shown increased perception of radon severity in females (Duckworth, Frank-Stromborg, Oleckno, Duffy, & Burns, 2002) and a relationship between female gender (versus white males) and increased perceived risk of environmental hazards such as radon (Flynn, Slovic, & Mertz, 1994).

Social influence was an important predictor of radon testing intentions. Those who knew others who had tested for radon were almost seven times more likely to plan to test themselves. The actions or accounts of others, such as family, friends, or neighbors, may influence individuals to transition from being unengaged with radon as a health hazard to deciding to take action (Weinstein, 1988; Weinstein & Sandman, 2002). Further research is needed to design interventions that use social influence to bridge the gap between radon awareness and radon testing.

Current smokers were more than six times more likely to plan to test for radon than nonsmokers. This finding is dissimilar to previous studies reporting less concern about radon among current smokers compared with nonsmokers (Kennedy et al., 1991; Mainous & Hagen, 1993). Smokers with an awareness of or concern about radon may have been more likely to participate in our study and to plan to test. A larger sample size with more smokers is needed to assess for differences in synergistic risk perception between current smokers and nonsmokers. Previous studies have suggested that health education strategies, including radon risk reduction messages, focus on smokers (Kennedy et al., 1991; Mainous & Hagen, 1993). Increasing smokers' concern about radon, specifically about the combined risk of radon and tobacco smoke, is an important area for future interventions and research. The association between the combined perceived risks of smoking and radon may also provide rationale for combining health promotion messages both to eliminate secondhand smoke exposure and to promote smoking cessation and radon testing and mitigation.

As hypothesized, perceived susceptibility was significantly higher in the group with testing intentions. Perceived susceptibility has been cited as a prerequisite to radon testing intention and behavior (Duckworth et al., 2002; Weinstein, 1988; Weinstein & Sandman, 1992). Others have reported lack of a perceived radon problem in one's home (Kennedy et al., 1991) and a perception of environmental hazards as riskier for others than for oneself (Park, Scherer, & Glynn, 2001) as reasons for not reducing radon risk. These findings support the need for targeted radon risk communication that seeks not only to increase general public knowledge, but also to provide geographic-specific risk information (Wang et al., 2000).

Sociodemographic factors, including gender, socioeconomic status (i.e., education and income), and age were controlled for in our study. Despite increased perceived radon severity among females and previous research showing a relationship between female gender and radon testing (Halpern & Warner, 1994), females were no more likely to plan to test for radon than males. Those with a college education were nearly five times more likely to plan to test compared to those with less education. Income level was inversely associated with perceived severity, but a lack of respondent income variability decreases the relevance of this finding. Age was unrelated to either testing intentions or perceived severity. All but two participants were Caucasian; thus, the association between race and testing intentions could not be analyzed.

Selection bias is a possible limitation in our study. Group I study participants were more likely to have graduated from college, had higher incomes, and lower smoking rates than those living in their respective counties (CDC, 2010; U.S. Census Bureau, 2010). One reason for this discrepancy is that only homeowners were selected for inclusion. Those with higher socioeconomic status are more likely to be homeowners (Haurin, Herbert, & Rosenthal, 2007). Those who chose to respond to the survey may have had more radon testing knowledge or may have been more likely to engage in health promotion behaviors than nonresponders. The complexity of survey questions may have discouraged participation of those with less education. Professional status or occupation was not assessed in our study and may have impacted survey participation or responses. The low response rate for Group I was expected; the higher response rate among those requesting a test kit (Group II) may be related to self-selection and increased testing intentions.

Five participants (4%) reported having lung cancer, potentially biasing their responses. Individuals with a personal or family history of lung cancer may have been more likely to request a test kit (Group II) or more likely to respond to the survey (Group I). Individuals with a history of exposure (i.e., to lung cancer) may respond to risk information differently than those without exposure (Smith & Johnson, 1988). A final limitation of our study is its cross-sectional design. Although our study provides important information about associations between study variables and intentions to test for radon, it is unable to provide predictions about directional causality. Further research using prospective designs is needed to determine the impact of study variables, particularly synergistic risk perception, on the transition between stages of adopting radon testing behavior.

TABLE **3**

Factors Associated With Higher Perceived Radon Severity (*N* = 129)

Factor	Odds Ratio	95% Confidence Interval
Age	1.02	(0.98, 1.07)
Gender		
Female	**4.57**	(1.54, 13.58)
Male	*	
Income		
<$50,000	**4.23**	(1.16, 15.43)
≥$50,000	*	
Currently smoking		
Yes	2.61	(0.39, 17.49)
No	*	
Perceived susceptibility	1.28	(0.94, 1.74)
Synergistic risk perception		
Yes	**5.70**	(1.41, 22.98)
No	*	
Know others who have tested for radon		
Yes	0.70	(0.24, 2.04)
No	*	

Note. Bolded values are significant at the .05 level.
*Reference group.

Conclusion

Perceived severity, social influence, and smoking were the most predictive of residential radon testing intentions. College education and perceived susceptibility to radon were also related to plans to test for radon. Females and those rating combined risk from smoking and radon exposure as much riskier than tobacco smoking alone had higher perceived radon illness severity. The associations between current smoking and testing intentions and between synergistic risk perception and perceived severity provide rationale for interventions to increase synergistic risk perception among smokers, thereby decreasing radon-related lung cancer risk among smokers. Our research has implications for interventions to improve residential air quality and decrease lung cancer risk through radon risk reduction.

Acknowledgements: Our study was a collaborative effort with the Kentucky Radon Program and was supported by Clean Indoor Air Partnership, University of Kentucky College of Nursing; Delta Psi Chapter, Sigma Theta Tau International Honor Society for Nursing; and Rural Cancer Control Program Traineeship, University of Kentucky, funded by the National Cancer Institute (R25 CA098220; Dr. Mark Dignan, PI).

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INTERNATIONAL PERSPECTIVES

Prepublished online March 2013,
National Environmental Health Association.

Correlation of Arsenic Exposure Through Drinking Groundwater and Urinary Arsenic Excretion Among Adults in Pakistan

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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Abstract **Long-term exposure to arsenic has been associated with manifestation of skin lesions (melanosis/keratosis) and increased risk of internal cancers (lung/bladder). The objective of the study described here was to determine the relationship between exposure of arsenic through drinking groundwater and urinary arsenic excretion among adults ≥15 years of age living in Khairpur district, Pakistan. Total arsenic was determined in drinking groundwater and in spot urine samples of 465 randomly selected individuals through hydride generation-atomic absorption spectrometry. Spearman's rank correlation coefficient was calculated between arsenic in drinking groundwater and arsenic excreted in urine. The median arsenic concentration in drinking water was 2.1 µg/L (range: 0.1–350), and in urine was 28.5 µg/L (range: 0.1–848). Positive correlation was found between total arsenic in drinking water and in urine (r = .52, p < .01). Urinary arsenic may be used as a biomarker of arsenic exposure through drinking water.**

Introduction

Arsenic is a ubiquitous element occurring naturally in the environment in both organic and inorganic forms. Inorganic arsenic is more toxic to humans and more prevalent in groundwater either naturally through geochemical and weathering processes or by anthropogenic activities such as agriculture, industry, and mining (National Research Council, 2001).

Arsenic has been identified as a potent human carcinogen by the International Agency for Research on Cancer (2004). Millions of people are exposed to arsenic through drinking groundwater with arsenic concentrations above the World Health Organization (WHO) guideline value of 10 µg/L in many parts of the world, including parts of the U.S., Canada, Argentina, Chile, Mexico, Hungary, and various countries of Southeast Asia, including Bangladesh, West Bengal India, Nepal, China, Myanmar, Thailand, Taiwan, Vietnam, Cambodia, Laos, and Pakistan (Caussy, 2005; Rahman et al., 2001).

Long-term human exposure to arsenic through drinking water containing arsenic above 10 µg/L is associated with the following:

- characteristic skin lesions, melanosis (darkening of skin) and keratosis (thickening of the skin of palms and soles) (Kadono et al., 2002);
- increased blood pressure (Lee et al., 2005);
- decreased lung function (Pervez et al., 2008);
- cardiovascular disease (Chen, Chiou, Chiang, Lin, & Tai, 1996);
- diabetes (Rahman, Todel, Ahmed, & Axelson, 1998; Tseng et al., 2000);
- adverse reproductive outcomes (Ihrig, Shalat, & Baynes, 1998);
- stillbirth (Cherry, Shaikh, McDonald, & Chowdhury, 2008);
- cancers of lung and bladder (Michaud et al., 2004; Morales, Ryan, Kuo, Wu, & Chen, 2000; Mukherjee et al., 2003); and
- decreased intellectual functions and peripheral neuropathy (Wasserman et al., 2004).

TABLE 1

Demographics Details of the Study Population of Khairpur District (*N* = 465)

Characteristics	#	%
Age (years)		
15–29	153	32.9
30–44	163	35.1
45–59	102	21.9
≥60	47	10.1
Mean (*SE*)[a]	38 (0.7)	–
Gender		
Male	219	47.1
Female	246	52.9
Education level		
College	49	10.5
Secondary	63	13.5
Primary	71	15.3
No formal education	282	60.6
Smoking status		
Never smoked	384	82.6
Ever smoked	81	17.4
BMI (kg/m^2)		
Low (<18.5)	64	13.8
Normal (≥18.5)	401	86.2
Mean (*SE*)	23 (0.2)	–

[a]*SE* = standard error of mean.

Arsenic has a short half-life in the body; it is readily excreted in urine in 1–3 days (Calderon, Hudgens, Le, Schreinemachers, & Thomas, 1999; Chen, Amarasiriwardena, Hsueh, & Christiani, 2002; Karagas et al., 2001). Interindividual variability in arsenic excretion in urine at similar levels of arsenic exposure was also reported (Del Razo, Aguilar, Sierra-Santoyo, & Cebrian, 1999).

Accurate estimation of arsenic exposure is required for risk assessment of arsenic's adverse health effects and for making mitigation decisions. Most of the previous epidemiologic studies have measured arsenic in the available drinking water to estimate individual exposure (Rahman et al., 2006; Yu, Sun, & Zheng, 2006). In our study, human exposure to arsenic through drinking water was measured by determining the concentration of arsenic excreted in urine, testing it as a biomarker of current arsenic exposure. Many of the correlation studies of arsenic in water and urine have showed positive correlation. Those studies, however, were based on selective sampling in a few towns or communities with high arsenic exposure with a limited number of water and urine samples; for instance, 167 subjects were recruited for water (*n* = 164) and urine (*n* = 176) sample analysis (Ahsan et al., 2000); 96 subjects gave water and urine samples (Calderon et al., 1999); 43 subjects gave water (*n* = 35) and urine (*n* = 43) samples (Mera, Kopplin, Burgess, & Gandolfi, 2004); and 346 subjects gave urine and water (*n* = 86) samples (Watanabe et al., 2001).

Ours is the first study from Pakistan based on arsenic exposure estimates through urinary arsenic excretion in a population that is chronically exposed to arsenic from drinking groundwater. Our study was part of a larger investigation in which the prevalence of arsenicosis (melanosis or keratosis) was evaluated in a population chronically exposed to arsenic through drinking water (Fatmi et al., 2009). The objective of the present study was to assess the relationship between arsenic concentration in the drinking water and total arsenic excretion in urine among the adult population of one of the arsenic-affected districts (Khairpur) of Pakistan.

Materials and Methods

Study Site

Khairpur district is located in the northern part of Sindh province along the river Indus. It is a dry and hot climate area; average annual precipitation is 78 mm, relative humidity in summer is 48%, and humidity in winter is 61%. Wind speed in summer is 10 kilometers per hours (kph) and in winter it is 5 kph. The maximum monthly average temperature in June is 44°C and in January it is 23°C (Pakistan meteorological department). The district has an area of 15,910 km^2 and a population of 1,546,587, of which 76% is rural (Population Census Organization, Ministry of Economic Affairs and Statistics, Government of Pakistan, 2006). The district is divided administratively into eight talukas (subdistricts), i.e., Khairpur, Faiz Gunj, Thari Mirwah, Kot Diji, Gambat, Kingri, Sobho Dero, and Nara. The residents are primarily involved in agriculture and groundwater is utilized for drinking purposes. Khairpur district was identified as one of the arsenic-affected districts of Sindh province in a national arsenic survey in 2001 (Ahmed, Kahlown, Tahir, & Rashid, 2004).

Study Design and Subjects

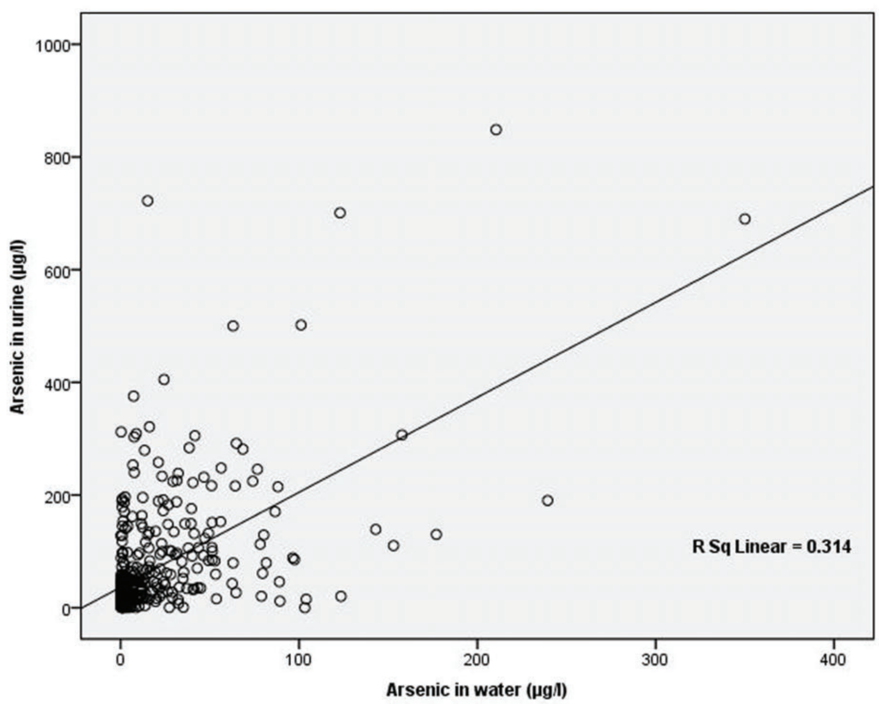
A cross-sectional survey was conducted from January to May 2006. Multistage cluster sampling was performed. Out of a total of 1,858 villages in Khairpur, 216 villages were randomly selected for a primary prevalence study. From each selected village, 10 households were identified for the survey. In each selected household, one male and one female who had been living there for at least the previous six months and who were at least 15 years old were recruited for the interview. Drinking water samples from all participants were taken for arsenic content determination. Urine samples were taken on the spot from one male participant (in household three) and from one female participant (in household seven) in each visited village. In addition, urine samples were also collected from those participants who had arsenic skin lesions (melanosis or keratosis). During the survey, a total of 505 spot urine samples were collected. Out of those, 40 urine samples were excluded from analysis because of unavailability of corresponding arsenic water results due to spillage of samples during transportation. Therefore, 465 spot urine samples were included in the analysis.

Exposure Assessment

An exposure assessment questionnaire was filled out by each individual in the study. Individuals were excluded who had eaten seafood (fish) during the past three days. Demographic information was also taken (age, sex, education, and occupation). Individuals were asked to estimate their average daily water consumption in order to determine their exposure. Other sources of arsenic exposure (environmental, occupational, and use of herbal medicines) were also assessed. Cigarette smoking and nutritional status (body mass index [BMI]) were evaluated. Drinking water and spot urine samples from study individuals were sent for laboratory analysis of arsenic concentration.

FIGURE **1**

Correlation Graph (Scatter Plot) Between Total Arsenic Concentration in Drinking Water and Arsenic Excreted in Urine



$r = .52$, $n = 465$, $p < .01$.

Daily Arsenic Intake Estimation

Daily arsenic intake was calculated directly by multiplying the arsenic concentration in drinking water by the amount of water consumed per day (based on recall) per body weight of the study individuals. Arsenic estimates were not calculated for solid food.

Water and Urine Sample Collection and Processing

The current drinking water source (home hand pump) and spot urine from our study subjects were sampled. Water samples were collected in 0.5-L (500 mL) and urine samples in 0.1-L (100 mL) polyethylene containers. Before sample collection, these containers were washed with nitric acid and then rinsed with deionized water. For water sampling, after discarding the initial water (10 pumps for one minute) from the hand pump, the water was collected in the container. For urine samples, a container was provided to the study subject for a urine sample at the time of the interview. People who had eaten fish or prawns (seafood) within three days of the interview were excluded from the analysis.

To preserve the samples, 1 mL of 1% nitric acid (HNO_3) was added to the water sample, and 2–3 drops (0.5 mL) of 3% HNO_3 were added to the urine samples. The containers were capped and sealed with cloth tape and stored at room temperature. Stored samples were dispatched within one week to the Pakistan Council for Research in Water Resources (PCRWR) laboratory, Islamabad, for total arsenic analysis.

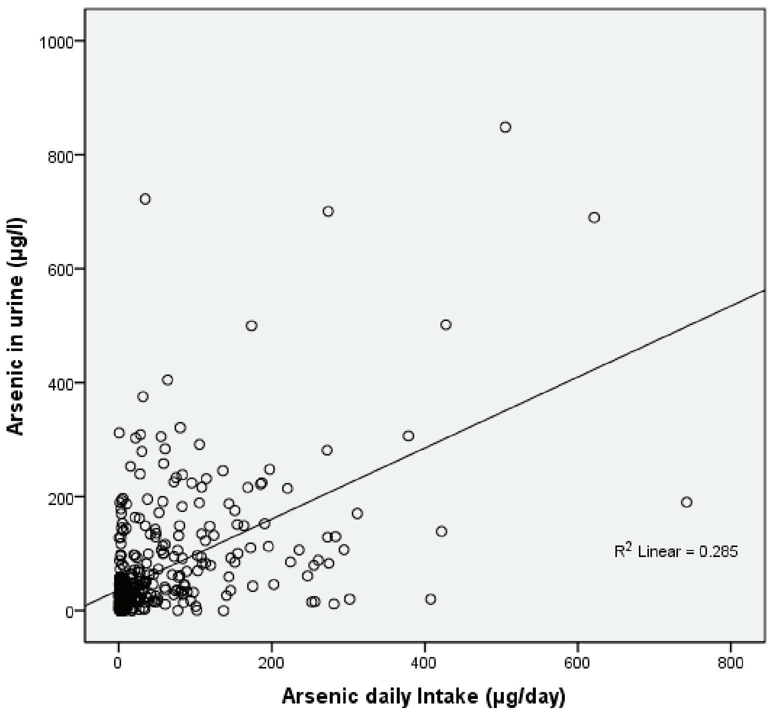
Total arsenic in water and urine samples was analyzed through hydride generation-atomic absorption spectrometry (HG-AAS) at the PCRWR laboratory, Islamabad. The minimum detection limit for arsenic was 0.1 µg/L. Calibration standards for arsenic with concentrations 0, 10, 20, 30, 40, and 50 µg/L were prepared by dilution of a certified standard solution (1,000 mg/L) from Fluka Kamica. Analytical performance was monitored by analysis of standard reference materials and internal water quality control samples.

Statistical Analysis

Descriptive analysis was performed, including frequency distribution of demographic variables (age, sex, and education), calculation of central tendency (mean/median), variability (percentiles), and arsenic concentration in water and urine. The association between the unspeciated arsenic in drinking water and the concentration of arsenic excreted in urine were evaluated graphically (scatter plot) and by calculating Spearman's rank correlation coefficient. Multivariate regression analysis was performed for adjusting potential confounders (age, sex, smoking, and BMI) of the outcome variable (arsenic in urine). All statistical analyses were computed by using SPSS v. 16.0.

FIGURE 2

Correlation Graph (Scatter Plot) Between Daily Arsenic Intake Through Drinking Water and Arsenic Excreted in Urine



r = .52, *n* = 465, *p* < .01.

Ethical Approval

The study protocol was reviewed and approved by the ethics review committee of Aga Khan University. All study participants signed an informed consent form.

Results

A total of 465 individuals were recruited to participate in the study from 216 villages of Khairpur district. Females numbered 246 (53%). The mean age of the study participants was 38 ± 15.1 years. Among participants, 61% had no formal education, 17% were smokers, and 14% had low BMI (<18.5) (Table 1). Drinking water samples (465) were collected from household water sources and analyzed for total arsenic concentration. The arsenic concentrations in drinking water samples ranged from 0.1 to 350 µg/L and the median was calculated as 2.1 µg/L. Approximately 147 (32%) drinking water samples had arsenic above the WHO guideline value of 10 µg/L and 42 (9%) samples were above the Pakistan guideline value of 50 µg/L.

A total of 465 spot urine samples were analyzed for total arsenic. Urinary arsenic concentrations ranged from 0.1 to 848 µg/L and the median was 28.5 µg/L.

In the study population the average water consumption volume was 2.8 ± 0.9 L per day (range: 0.75–5.75 L/day). Median arsenic concentration intake through drinking water was 5.7 µg/day (range: 0.1–742 µg/day), per body weight daily arsenic intake ranged from 0.1 to 14.7 µg/kg/day, and the median was 0.10 µg/kg/day. Considering oral reference dose (RfD) of daily arsenic intake is 0.30 µg/kg/day (U.S. Environmental Protection Agency, 1998), then approximately 175 (37.6%) of the study individuals had been exposed to arsenic above RfD.

Spearman's rank correlation coefficient was calculated between total arsenic in drinking water and the total arsenic excreted in urine of the study individuals. We found a moderate degree of positive correlation (*r* = .52, *p* < .01), indicating a linear relationship between total arsenic in drinking water and total arsenic excreted in urine (Figure 1). Similar positive correlation was also found when daily arsenic intake (µg/day) through drinking water was compared with the total arsenic excreted in urine (*r* = 0.52, *p* < .01) (Figure 2). Potential confounders (age, sex, smoking, and BMI) were adjusted by using multivariate linear regression and the arsenic in drinking water was found more associated with urinary arsenic (B coefficient = 1.672; *p* < .001).

Discussion

Our study provides information on arsenic exposure to humans through drinking groundwater. This was established by using the urine samples of exposed individuals as a biomarker of arsenic exposure. People in the study district (Khairpur) have continuously ingested unsafe levels of arsenic (above 10 µg/L) through drinking groundwater for a long time. No alternate safe water options exist, and people have been developing arsenic-induced skin lesions, melanosis and keratosis (Fatmi et al., 2009). Our study consolidated the evidence of arsenic exposure in the chronically exposed population solely through drinking groundwater after adjusting independent predictor variables (age, sex, smoking, and BMI) of the outcome variable (arsenic in urine) by using multivariate regression analysis.

Study drinking water samples (*n* = 465) were screened for arsenic concentration using the "gold standard" laboratory method (HG-AAS). Hence reliability of the water arsenic test results was not compromised. Previous studies performed arsenic screening for only limited commonly shared water sources (Calderon et al., 1999; Mera et al., 2004) and through using a less sensitive arsenic field test kits method (Rahman et al., 2006).

Spot urine specimens, instead of morning urine samples, were utilized for the study to determine the arsenic concentration in urine of the study population who were exposed to arsenic through drinking groundwater. Studies have reported that concentration of arsenic in urine remains stable throughout the day and over a period of five consecutive days (Calderon et al., 1999). The stability of arsenic concentration in urine suggests that the exposed population was at steady state in terms of exposure to arsenic.

Limitations of our study were that only total arsenic concentration was detected in drinking water and urine samples and we did not perform arsenic speciation due to limited laboratory capacity and financial constraints. Arsenic levels in urine were determined only through weight by volume without adjustment of urinary creatinine levels (weight/weight). Another limitation of our study was related to the methods of estimation of arsenic intake. In our study, the estimation of individual arsenic intake was calculated only from drinking water of the household water source, supposed as a main source of drinking water and a stable source of arsenic exposure; other sources of drinking water such as the workplace were not considered. Dietary (solid food) sources of arsenic were also not considered. A previous study reported that the average contribution to the total arsenic intake from solid food was 11% (Ohno et al., 2007).

Median arsenic concentration in drinking water was reported as 2.1 µg/L (right-skewed data distribution), which lies within the normal limit of WHO's guideline value. Still, nearly one-third of the study population was exposed to arsenic through drinking water above 10 µg/L at higher percentiles (Table 2).

In the study population, average daily water intake was reported high (2.8 L) possibly because of the hot climate of the area (maximum monthly average temperature in summer was 42°C); therefore body water demand increased and consequently an increased amount of arsenic was ingested. Considering this fact only from a single household drinking water source, daily arsenic exposure per body weight was above RfD in more than one-third of the study population.

Moderate correlation was reported between arsenic exposure through drinking water and urinary arsenic excretion. Considerable variability is found, however, between the level of arsenic exposure through drinking water and urinary arsenic excretion; for instance, variability observed at the median level is 14 times higher and at the 75th percentile level it is 3–4 times higher for urinary arsenic excretion (Table 2).

TABLE **2**

Arsenic Concentrations in Drinking Water, Daily Intake, and Urine of Study Population of Khairpur District (*N* = 465)

Characteristics	Drinking Water (µg/L)	Daily Intake (µg/day)	Urine (µg/L)
Maximum	350	742	848.5
90th percentile	46.5	128.9	171.1
75th percentile	15.8	41.4	67.7
Median	2.1	5.7	28.5
25th percentile	0.9	2.5	11.2
10th percentile	0.4	1.1	3.0
Minimum	0.1	0.1	0.1
# of cases	465	465	465

TABLE **3**

Age-Related Difference in Arsenic Concentration in Urine in Study Population of Khairpur District (*N* = 465)

Characteristics	Age Groups (Years)			
	15–29	30–44	45–59	≥60
Mean	72.6	60.9	56.0	55.2
Median	27.7	29.7	29.6	26.0
Minimum	0.1	0.0	0.4	1.2
Maximum	848	700.8	309	722.4
25th percentile	10.8	9.6	13.6	12.5
75th percentile	74.8	67.6	67.9	55.5

Our study could not find an age-related difference in arsenic excretion in urine (Table 3). Similar results were reported in a study conducted in a northern Argentina population chronically exposed to arsenic through drinking water (Concha, Nermell, & Vahter, 1998). When correlating with gender, no significant difference was found in urinary arsenic excretion in either males or females. This finding is in contrast with the finding reported in earlier studies in which distinct gender-related differences have been reported in the excretion of arsenic in urine (Del Razo et al., 1994; 1997).

In conclusion, a moderate correlation exists between the concentration of arsenic in drinking water and the concentration of arsenic excreted in urine. This suggests that over the range of exposure, drinking water was the predominant source of arsenic exposure. Hence arsenic in urine came out as a good predictor of arsenic exposure to humans through drinking water.

Conclusion

A moderate positive correlation exists between arsenic exposure through drinking water and the concentration of arsenic in urinary excretion. This would suggest that if the arsenic content in drinking water is higher then the arsenic excretion in urine would be higher. Thus arsenic excretion in urine can be used as a good indicator of arsenic exposure.

Acknowledgments: We would like to acknowledge the donor agency UNICEF for providing financial support, PCRWR for doing laboratory analysis, and local government Sindh for facilitating the study. We would also like to thank our field staff who made it possible to conduct this survey. Lastly we thank the people of Khairpur for their hospitality and participation in the survey.

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continued on page 54

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Prepublished online March 2013,
National Environmental Health Association.

Risk Assessment of Heavy Metals in Shellfish for the Population in Nha Trang City, Vietnam

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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Nha Trang University

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Abstract **The study described in this article was designed to estimate the dietary intake of lead, cadmium, and mercury due to the shellfish consumption of the population in Nha Trang City, Vietnam. The lead, cadmium, and mercury concentrations in the shellfish consumed popularly by the Nha Trang population were investigated by inductively coupled plasma-mass spectrometry from May 2008 to January 2009. The lead, cadmium, and mercury concentration ranges in shellfish are equal to 0.008–0.083, 0.013–0.056, and 0.028–0.056 mg/kg, respectively. The dietary intake of these elements was determined by a total diet study. The heavy metals intake was estimated for six subpopulation groups: men and women aged 18–29, 30–54, and ≥55. The dietary intakes of lead, cadmium, and mercury by the Nha Trang population are currently well below the provisional tolerable weekly intakes of lead, cadmium, and mercury, respectively. Therefore, no risk exists concerning the levels of exposure of Nha Trang consumers to the contaminants studied due to shellfish consumption.**

Introduction

Vietnam has a long coastline of 3,260 km from the northern to the southern part of the country. The residents living in the southern coastal region are significant consumers of shellfish. Unfortunately, shellfish have the capacity to accumulate heavy metals. Therefore, shellfish consumption is a significant pathway of human exposure to heavy metals and presents a potential risk for consumers. Hence, it was necessary to evaluate exposure of consumers to these contaminants and perform a risk assessment. Nha Trang City was chosen as a representative southern coastal area for Vietnam in shellfish consumption.

Among heavy metals, lead, cadmium, and mercury are the more toxic, even in trace amounts. Pollution from lead, cadmium, and mercury is a serious problem due to their toxicity and ability to accumulate in the biota. Consequently, these three heavy metals have been included in the regulations of Vietnam, the European Union, and Codex for hazardous metals (Codex Alimentarius Commission, 2001; Codex Alimentarius Commission, 2004; European Union Commission Regulation, 2006; Vietnamese Health Ministry, 2007; World Health Organization [WHO], 2006). Therefore, the content of lead, cadmium, and mercury in shellfish is worthy of consideration. Few systematic studies, however, provide data on content of lead, cadmium, and mercury in shellfish consumed popularly in Vietnam. Some studies do concern the determination of the lead, cadmium, and mercury concentration in shellfish in Vietnam (Dao, 2002; Doan, Nguyen, Nguyen, Nguyen, & Nguyen, 2003; Hsia & Huiyi, 2008; Le, 2005, 2006; Le, 2008; Le, Nguyen, Pham, & Duong, 2005; National Agro-forestry and Fisheries Quality Assurance Department, 2006; Ngo, 2008). Unfortunately, in these studies, the samples collected were not representative of shellfish consumed popularly. Furthermore, these studies were only realized in some species of shellfish. So our study was undertaken to examine the concentrations of heavy metals in shellfish mostly consumed and collected from the popular purchase places in Nha Trang in different months to consider the seasonal variation.

Materials and Methods

Heavy Metals Detection

According to the data of the shellfish consumption survey (Nguyen, Tran, Carpentier, Roudot, & Parent-Massin, 2010), 19 shell-

TABLE 1

Recovery (%) of Heavy Metals and Relative Standard Deviation (RSD %)

	Recovery (%)	RSD (%)
Cadmium	99.5	10
Lead	101.0	14
Mercury	106	12

fish species that have mean consumption rates over 1 g/day were selected for heavy metals detection. Four shellfish composite samples (bivalves, crustaceans, gastropods, cephalopods) were prepared to reduce the number of samples without modification of data precision (WHO, 1985). The proportion of shellfish in each composite was derived from the data obtained in the consumption survey (Nguyen et al., 2010). The sampling was performed randomly in the markets, temporary markets, and restaurants in Nha Trang City in May, July, September, and November 2008 and January 2009.

The samples for lead and cadmium detection were dried at 105°C using a drying oven. Samples for mercury detection were frozen at -70°C in a slurry of ethanol and dry ice and then left in a lyophilizer for about 24 hours. Dried samples were digested in nitric acid using a microwave digester. Inductively coupled plasma-mass spectrometry (ICP-MS) was used to determine the lead, cadmium, and mercury concentrations. All chemicals and standard solutions were obtained from Merck. The analyses were carried out on composite homogenized samples in the laboratory, which is officially accredited according to ISO 17025. The regular tests of controllable qualitative samples such as the blanks and duplicates were performed. Means and standard deviation were calculated. Trueness is stated quantitatively in terms of "bias," with smaller bias indicating greater trueness. Bias was investigated by spiking and recovery (Thompson, Ellison, & Wood, 2002). Relative standard deviation (RSD %) was used in evaluation of the precision of the methods used. RSD (%) was calculated as follows: RSD (%) = *SD* * 100/X, where RSD (%) = relative standard deviation in samples, *SD* = standard deviation (mg/kg), and X = mean (mg/kg). Statistical analyses were performed using SPSS v. 16. The significance level was $p < .05$. Differences in heavy metals concentrations between months were tested by one-way ANOVA with Tukey tests.

Calculation of Dietary Intake

The dietary intake of lead, cadmium, and mercury by the population of Nha Trang City has been calculated according to consumption data (Nguyen et al., 2010) and contamination data determined in our study. The probabilistic analyses were performed with @Risk for Excel International, version 4.5.6. The Monte Carlo method and Latin Hypercube sampling were used. The number of Monte Carlo iterations used for the calculations was 10.000.

The dietary intake by the population of Nha Trang was calculated for each heavy metal by the following equation:

$$D = \sum_{i=1}^{n} Q_i C_i$$

where

D = distribution of intake (μg/kg body weight/day) of lead, cadmium, or mercury by a particular population subgroup (six population subgroups: men and women (18–29, 30–54, and ≥55)

Q_i = distribution of consumption of shellfish's group i (g/kg body weight/day)

C_i = maximum concentration of contamination in shellfish's group i

n = number of shellfish's group

Results

Heavy Metals Contamination

The results of analysis showed good recovery when spiked with standards. Concretely, 97%, 101%, and 106% recovery was obtained for cadmium, lead, and mercury (Table 1).

Lead concentrations in bivalves, crustaceans, gastropods, and cephalopods in five sampled months were in the ranges of 0.051–0.114, 0.01–0.064, 0.04–0.103, and 0.007–0.009 mg/kg, respectively. The mean lead concentration in bivalves, gastropods, crustaceans, and cephalopods was 0.083, 0.073, 0.032, and 0.008 mg/kg, respectively (Table 2).

Figure 1 shows the seasonal variations of the lead concentrations in bivalves, crustaceans, gastropods, and cephalopods. The tendency of these seasonal variations is a rise from May to November and then a slight decline from November to January. The concentrations of lead in bivalves, crustaceans, and gastropods are the highest in November. The differences of concentrations of lead in bivalves and gastropods between September, November, and January are significant ($p < .05$). But they are not significant between May and July ($p > .05$). The lead concentration in crustaceans is different significantly ($p < .05$) between May and July, but the difference is slight. The lead concentrations in bivalves, crustaceans, and gastropods in the rainy season (September, November, and January) are significantly higher than in the dry season (May, July). The maximal lead concentrations in the shellfish except for cephalopods are observed in November. The lead concentrations in cephalopods are low and constant in all months (Figure 1).

Cadmium concentrations in bivalves, crustaceans, gastropods, and cephalopods in five sampled months were in the ranges of 0.032–0.073, 0.019–0.034, 0.033–0.072, and 0.012–0.013 mg/kg, respectively. The mean concentration of cadmium in bivalves, gastropods, crustaceans, and cephalopods is 0.056, 0.054, 0.026, and 0.013 mg/kg, respectively (Table 2). The cadmium concentrations in bivalves, crustaceans, and gastropods in the rainy season are significantly higher (Table 2) than in the dry season. The maximal cadmium concentrations in the shellfish except for cephalopods are observed in November. The cadmium concentrations in cephalopods are fairly low and constant in all months (Figure 2).

TABLE **2**

Lead, Cadmium, and Mercury Concentrations (mg/kg Wet Weight) in Various Shellfish Consumed in South Coastal Vietnam

Shellfish	Month	Lead		Cadmium		Mercury	
		Concentrations (mg/kg wet weight)	RSD[a] (%)	Concentrations (mg/kg wet weight)	RSD (%)	Concentrations (mg/kg wet weight)	RSD (%)
Bivalves	May	0.051	4.7	0.032	4.7	0.023	4.5
	July	0.054	2.8	0.032	3.6	0.024	5.7
	September	0.096	1.5	0.068	1.3	0.081	1.5
	November	0.114	2.4	0.073	1.2	0.087	1.4
	January	0.101	2.4	0.073	1.1	0.066	2.0
	Mean±SD	0.083±0.026	–	0.056±0.02	–	0.056±0.028	–
Crustaceans	May	0.010	10.2	0.019	5.4	0.027	3.8
	July	0.015	12.7	0.019	6.7	0.020	5.6
	September	0.033	5.3	0.026	4.0	0.033	3.0
	November	0.064	2.6	0.034	2.9	0.032	3.7
	January	0.037	4.6	0.032	3.2	0.028	2.9
	Mean±SD	0.032±0.02	–	0.026±0.006	–	0.028±0.005	–
Gastropods	May	0.040	4.0	0.033	3.6	0.024	3.7
	July	0.040	5.0	0.033	3.0	0.021	5.3
	September	0.084	1.9	0.064	1.6	0.053	1.8
	November	0.103	1.0	0.072	1.1	0.064	1.6
	January	0.096	1.9	0.067	1.7	0.061	1.4
	Mean±SD	0.073±0.028	–	0.054±0.017	–	0.045±0.019	–
Cephalopods	May	0.007	15.5	0.012	8.9	0.021	4.7
	July	0.008	15.1	0.013	8.6	0.023	4.6
	September	0.008	13.0	0.013	6.2	0.064	1.3
	November	0.009	13.0	0.013	4.6	0.073	1.1
	January	0.009	12.9	0.013	6.6	0.068	1.4
	Mean±SD	0.008±0.001	–	0.013±0.001	–	0.050±0.023	–

[a]RSD = relative standard deviation.

Mercury concentrations in bivalves, crustaceans, gastropods, and cephalopods were in the ranges of 0.023–0.087, 0.020–0.033, 0.021–0.064, and 0.021–0.073 mg/kg, respectively. The mean concentration of mercury in bivalves, cephalopods, gastropods, and crustaceans is 0.056, 0.050, 0.045, and 0.028 mg/kg, respectively (Table 2). The mercury concentrations in bivalves, crustaceans, and gastropods in the rainy season are significantly higher (Table 2) than in the dry season. The maximal mercury concentrations in the shellfish except for crustaceans are observed in November. The mercury concentrations in crustaceans are fairly low and constant in all months (Figure 3).

Estimation of the Dietary Intake of Heavy Metals

In Table 3, the rows reflect the percentiles of lead, cadmium, and mercury exposures for six population subgroups (men and women aged 18–29, 30–54, and ≥55). The columns reflect variability in lead, cadmium, and mercury exposures as a function of the quantity of shellfish consumed by different age groups. For all population subgroups, the lead, cadmium, and mercury exposure was lower than the provisional tolerable weekly intake (PTWIs) for lead (25 µg/kg body weight/week [bw/w]), cadmium (2.5 µg/kg bw/w), and mercury (5 µg/kg bw/w), respectively (European Food Safety Authority, 2009; WHO, 1999, 2003).

Discussion

Heavy Metals Contamination

Dry season occurs in Nha Trang from May to July, while September, November, and January are during the rainy season. Shellfish are more contaminated during the rainy season. This could be due to runoff of heavy metals after heavy rains. The Food and Agriculture Organization (2008) identified several hazards induced by increase of rains due to climate change. An increase of runoff of heavy metals induced by storms and an increase of shellfish contamination by heavy metals in coastal zones are mentioned as two of these hazards.

FIGURE **1**

Lead Concentrations in Shellfish

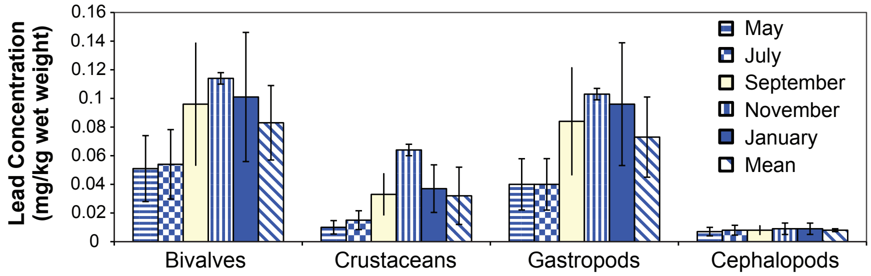


FIGURE **2**

Cadmium Concentrations in Shellfish

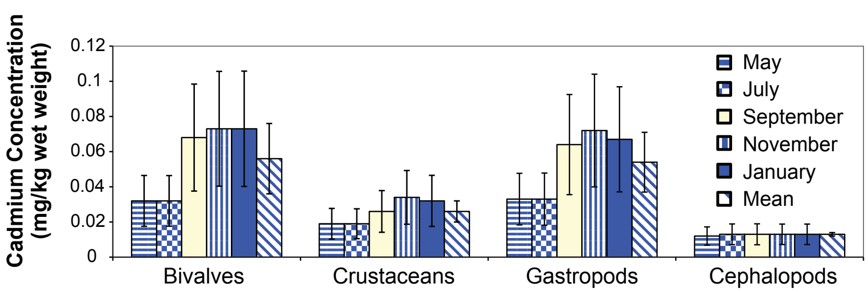
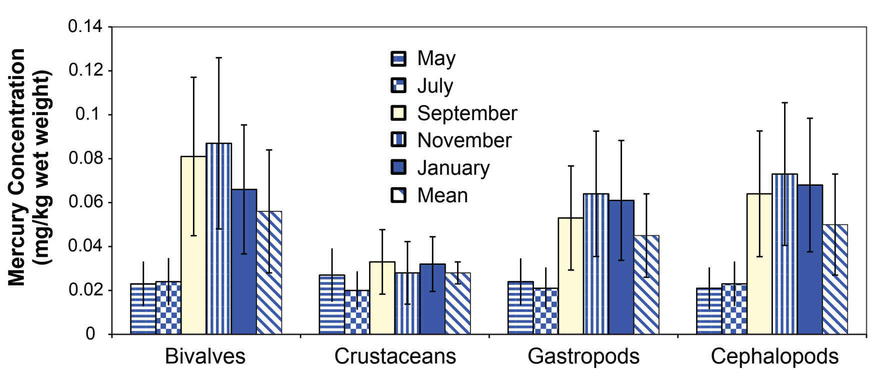


FIGURE **3**

Mercury Concentrations in Shellfish



The heavy metal contamination data in shellfish in our study have to be compared to those of other studies in Vietnam (Dao, 2002; Doan et al., 2003; Le, 2005, 2006; National Agro-forestry and Fisheries Quality Assurance Department, 2006; Ngo, 2008); in Asia (Health Canada, 2007; Hsia & Huiyi, 2008; Ip, Li, Zhang, Wong, & Zhang, 2005; Liang, Shi, He, Jiang, & Yuan, 2003; Lin, Wong, & Li, 2004; Nakagawa, Yumita, & Hiromoto, 1997; Soegianto & Hamami, 2007; Soegianto & Supriyanto, 2008); and in other countries in the world (Chindah, Braide, & Sibeudu, 2004; Cre´pet, Tressou, Verger, & Leblanc, 2004; European Commission, 2004; Gagnon, Tremblay, Rouette, & François, 2004; Juresa & Blanusa, 2003; Kehrig, Costa, Moreira, & Malm, 2006; Leblanc, Volatier, Sirot, & Bemrah-Aouachria, 2006; Legrand, Arp, Ritchieb, & Chan, 2004; Mozaffarian & Rimm, 2006; Sanzo et al., 2001; Sivaperumal, Sankar, & Viswanathan Nair, 2007; Soliman, 2006; Tollefson & Cordle, 1986; Vannoort & Thomson, 2005; WHO, 2006).

The data comparison has to be considered carefully because of the differences in the sampling, in the sample preparation methods, and in the analytic methods. The uniqueness of our study, however, rests on the shellfish composite analysis, instead of shellfish individual analysis. In short, the data comparisons between studies are not significant because of the difference of methodology applied. But, we note that the lead, cadmium, and mercury concentrations in shellfish in our study and other studies are the same order of magnitude. The heavy metal concentrations in shellfish in our study are relatively low. Furthermore, the mean concentrations of lead, cadmium, and mercury detected in shellfish in Nha Trang is lower than the maximum limit of regulatory of European Community, Vietnam regulation, and Codex Alimentarius recommendations (Table 4).

Estimation of the Dietary Intake of Trace Elements

The contributions of the shellfish groups to the lead, cadmium, and mercury intake due to shellfish consumption are shown in Figures 4, 5, and 6. Bivalves, gastropods, and crustaceans contribute most to the total intake of lead (62%, 23%, 13%, respectively), cadmium (59%, 23%, 14%, respectively), and mercury (49%, 16%, 22%, respectively). But cephalopods have a small contribution (2% lead, 4% cadmium, and 12% mercury) to the total intake figure.

TABLE **3**

Lead (Pb), Cadmium (Cd), and Mercury (Hg) Intakes (μg/kg body weight/week) of Consumers in Southern Coastal Vietnam

Percentile of Exposure	Men (Age)								
	18–29			30–54			≥55		
	Pb	Cd	Hg	Pb	Cd	Hg	Pb	Cd	Hg
5	0.427	0.285	0.327	0.335	0.226	0.321	0.834	0.563	0.585
10	0.496	0.332	0.379	0.399	0.267	0.379	0.928	0.624	0.649
15	0.545	0.363	0.418	0.448	0.300	0.419	0.992	0.667	0.701
20	0.584	0.392	0.449	0.488	0.326	0.452	1.044	0.702	0.737
25	0.619	0.412	0.475	0.522	0.348	0.483	1.092	0.734	0.771
30	0.651	0.434	0.499	0.555	0.370	0.511	1.137	0.761	0.805
35	0.681	0.453	0.523	0.587	0.390	0.537	1.178	0.789	0.839
40	0.712	0.472	0.547	0.618	0.412	0.561	1.221	0.816	0.871
45	0.739	0.491	0.572	0.649	0.430	0.584	1.263	0.841	0.901
50	0.767	0.508	0.597	0.680	0.451	0.608	1.302	0.867	0.932
55	0.796	0.527	0.620	0.713	0.472	0.631	1.342	0.892	0.959
60	0.825	0.548	0.645	0.744	0.493	0.657	1.387	0.921	0.991
65	0.859	0.569	0.672	0.777	0.513	0.683	1.429	0.948	1.026
70	0.893	0.589	0.700	0.812	0.534	0.708	1.476	0.978	1.062
75	0.929	0.610	0.732	0.851	0.559	0.740	1.533	1.012	1.106
80	0.971	0.639	0.773	0.898	0.588	0.772	1.598	1.054	1.150
85	1.021	0.671	0.825	0.952	0.621	0.811	1.671	1.102	1.210
90	1.089	0.715	0.899	1.019	0.664	0.863	1.771	1.164	1.288
95	1.187	0.780	1.041	1.135	0.734	0.943	1.924	1.261	1.409
Mean	**0.783**	**0.518**	**0.632**	**0.700**	**0.461**	**0.617**	**1.330**	**0.883**	**0.953**
Percentile of Exposure	**Women (Age)**								
	18–29			30–54			≥55		
	Pb	Cd	Hg	Pb	Cd	Hg	Pb	Cd	Hg
5	0.629	0.411	0.490	0.641	0.405	0.494	0.481	0.308	0.384
10	0.748	0.484	0.570	0.745	0.469	0.576	0.566	0.363	0.449
15	0.826	0.533	0.632	0.820	0.519	0.632	0.627	0.400	0.496
20	0.888	0.576	0.681	0.880	0.558	0.676	0.677	0.433	0.536
25	0.946	0.613	0.724	0.933	0.590	0.716	0.723	0.461	0.569
30	1.002	0.645	0.765	0.977	0.619	0.754	0.765	0.489	0.603
35	1.055	0.681	0.804	1.024	0.647	0.789	0.806	0.515	0.634
40	1.103	0.711	0.842	1.070	0.678	0.821	0.848	0.542	0.666
45	1.155	0.741	0.880	1.116	0.705	0.852	0.887	0.565	0.697
50	1.205	0.774	0.916	1.161	0.735	0.884	0.932	0.595	0.731
55	1.256	0.807	0.955	1.202	0.762	0.918	0.975	0.625	0.765
60	1.314	0.842	0.992	1.248	0.789	0.952	1.020	0.653	0.801
65	1.370	0.879	1.033	1.296	0.822	0.990	1.069	0.684	0.839
70	1.431	0.916	1.079	1.348	0.855	1.032	1.124	0.722	0.883
75	1.497	0.958	1.127	1.405	0.892	1.081	1.188	0.763	0.933
80	1.572	1.008	1.186	1.474	0.933	1.132	1.259	0.808	0.987
85	1.664	1.068	1.256	1.554	0.985	1.197	1.350	0.862	1.053
90	1.785	1.139	1.342	1.651	1.050	1.274	1.464	0.947	1.142
95	1.983	1.267	1.489	1.817	1.151	1.387	1.663	1.077	1.287
Mean	**1.243**	**0.798**	**0.942**	**1.184**	**0.750**	**0.909**	**0.985**	**0.632**	**0.771**

TABLE **4**

Concentration of Heavy Metals in Shellfish in Nha Trang Compared to the Maximum Limit of Regulations of European Community (EC), Codex (CAC), and Vietnam

Shellfish	Lead (mg/kg)				Cadmium (mg/kg)				Mercury (mg/kg)		
	Concentration	Regulation			Concentration	Regulation			Concentration	Regulation	
		EC[a]	Codex[b]	Vietnam[c]		EC[a]	Codex[b]	Vietnam[c]		EC[a]	Vietnam[c]
Bivalves	0.083±0.026	1.5	1	1.5	0.056±0.02	1	1	1	0.056±0.028	0.5	0.5
Gastropods	0.073±0.028	1.5	1	1.5	0.054±0.017	1	1	1	0.045±0.019	0.5	0.5
Crustaceans	0.032±0.02	0.5	0.5	0.5	0.026±0.006	0.5	0.5	0.5	0.028±0.005	0.5	0.5
Cephalopods	0.008±0.001	1	1	–	0.013±0.001	1	1	–	0.05±0.023	0.5	0.5

[a]European Union Commission Regulation, 2006.
[b]CAC, 2001; CAC, 2004; World Health Organization, 2006.
[c]Vietnamese Health Ministry, 2007.

FIGURE **4**

Contributions of the Shellfish Groups to the Lead Intake

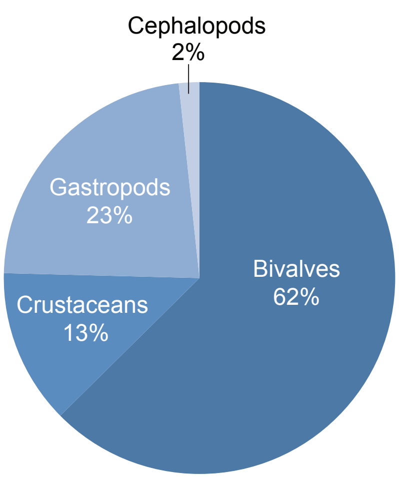


FIGURE **5**

Contributions of the Shellfish Groups to the Cadmium Intake

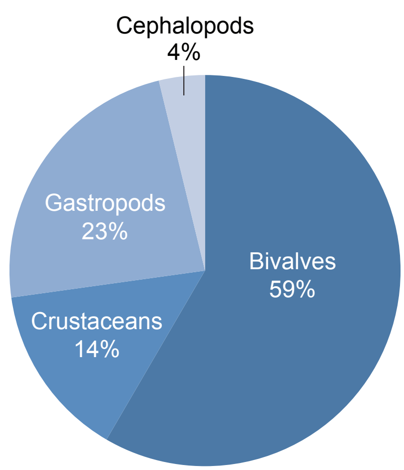
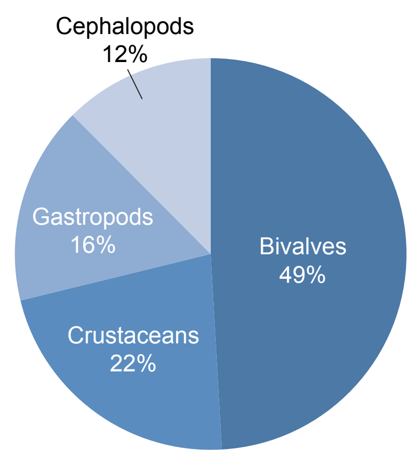


FIGURE **6**

Contributions of the Shellfish Groups to the Mercury Intake



The data of intake of lead, cadmium, and mercury due to shellfish consumption from different countries are rare. In general, the comparison of the mean weekly intake of lead, cadmium, and mercury for Vietnamese consumers due to shellfish consumption obtained in our study with the other studies around the world shows that the data are in the same order of magnitude and a large variability exists of the lead, cadmium, and mercury intake for shellfish consumers, depending on the countries (Figures 7, 8, and 9).

Figure 7 shows that mean intake of lead for the Vietnamese population due to shellfish consumption estimated in our study is lower than the one for the population in Holland due to crab, shrimp, lobster, mussel, and cephalopod consumption (Sorkina, Bakker, van Donkersgoed, & van Klaveren, 2003). The mean level of lead intake estimated in our study, however, is higher than the ones reported in the United Kingdom, Taiwan, Greece, Canada, Norway, France, Portugal, Ireland, Denmark, and Finland (European Commission, 2004; Gagnon et al., 2004; Lin et al., 2004).

Figure 8 shows that mean intake of cadmium for the Vietnamese population due to shellfish consumption estimated in our study is lower than the one for the population due to crab consumption in the United Kingdom (Food Standards Agency, 2006), due to mollusk, crustacean, and cephalopod consumption in Greece and due to bivalve, crustacean, and cephalopod consumption in Portugal (European Commission, 2004). By contrast, the mean level of cadmium intake estimated in our study is higher than the ones reported in Holland, Taiwan, France, Norway, Canada, Belgium, Ireland, Finland, Denmark, Germany, Sweden, and Italy (European Commission, 2004; Gagnon et al., 2004; Lin et al., 2004).

Figure 9 shows that mean intake of mercury for the Vietnamese population due to shellfish consumption estimated in our study is lower than the one for the population due to bivalve, crustacean, and cephalopod consumption in Japan (Nakagawa et al., 1997). By contrast, the mean level of cadmuim intake estimated in our study is higher than the ones reported in France, Taiwan, Portugal, Norway, Greece, Canada, Belgium, Holland, Ireland, Germany, and Denmark (Chen & Chen, 2006; European Commission, 2004; Gagnon et al., 2004; Lin et al., 2004; Sorkina et al., 2003; WHO, 2004).

In effect, the levels of lead, cadmium, and mercury intake for the population studied depend on the lead, cadmium, and mercury concentration in shellfish and the data of shellfish consumption of this population. The lead, cadmium, and mercury concentrations in shellfish in southern coastal Vietnam are relatively low in comparison to the data of many countries, but the level of shellfish consumption in Vietnam is relatively important (Nguyen et al., 2010). Consequently, it is reasonable to find out that the lead, cadmium, and mercury intake for the Nha Trang population due to shellfish consumption is rather important.

Conclusion

Ours was the first systematic study in Vietnam to determine the lead, cadmium, and mercury concentrations in shellfish consumed popularly and to evaluate exposure of Vietnamese consumers to these contaminants in shellfish consumed popularly. Our data confirm that the concentrations of lead, cadmium, and mercury in shellfish are lower than the limits fixed by the Vietnamese and inter-

FIGURE **7**

Mean Intake of Lead for the Population of Various Countries due to Shellfish Consumption

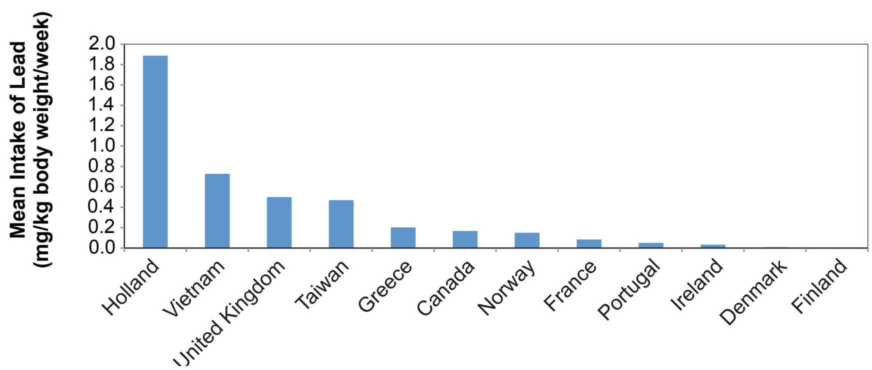


FIGURE **8**

Mean Intake of Cadmium for the Population of Various Countries due to Shellfish Consumption

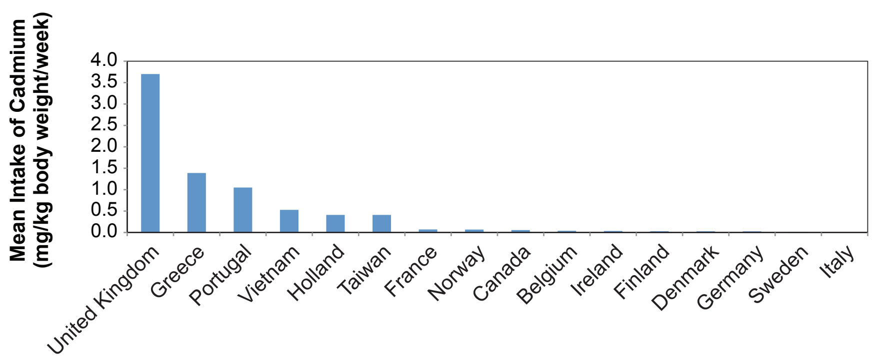
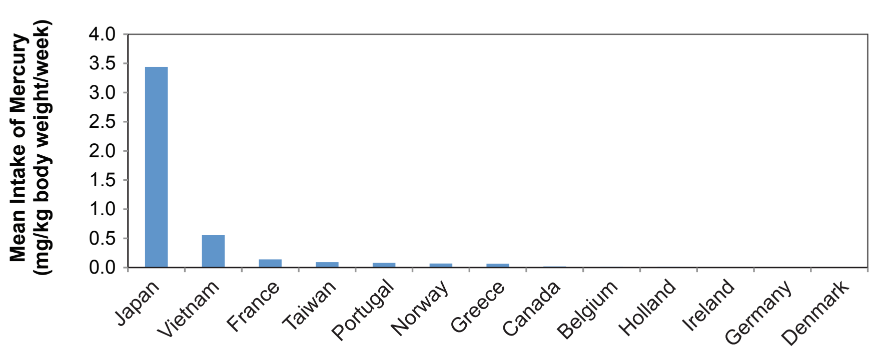


FIGURE **9**

Mean Intake of Mercury for the Population of Various Countries due to Shellfish Consumption



national regulations. Degrees of exposure of the studied population to these contaminants are rather important. The comparisons of these levels with the PTWI of lead, cadmium, and mercury permit us to conclude that no risk exists concerning the levels of exposure of consumers in the southern coastal region of Vietnam to the contaminants studied due to shellfish consumption.

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continued on page 64

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GUEST COMMENTARY/ INTERNATIONAL PERSPECTIVES

Prepublished online April 2013, National Environmental Health Association.

A Negative Correlation Between Dengue and Bushfires in Brazil

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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The number of bushfire focus in Brazil has grown tenfold in the last 10 years, from slightly more than 100,000 in 2000, peaking to around one million in 2007, with a total of 5.9 million focus observed in the 10 year period from 2000 to 2009 (Ciencia e Technologia e Meio Ambiente, 2010). The main reasons for this growth in the number of bushfires are the expansion of the agriculture frontier and, occasionally, droughts. A relationship between dengue and bushfires was first proposed by Massad and co-authors (2010), who hypothesized that haze caused by bushfires in Indonesia was responsible for an unexpectedly low number of dengue cases in Singapore in 2006. This hypothesis was rebutted by Wilder-Smith and co-authors (2010), who demonstrated through an autoregressive integrated moving average model the lack of association between dengue and haze (caused by bushfire) in Singapore. In Singapore, however, the level of haze related to bushfires is below the national alert level in the great majority of the weeks analyzed by Wilder-Smith and co-authors (2010). In the case described here, the intensity of haze caused by bushfires is much more severe than what is generally observed in Singapore and presumably above the hypothesized threshold for effects on mosquitoes. We carried out a univariate correlation analysis between the number of focus of bushfires and dengue cases in four Brazilian states for the period from 2000 to 2009. These states are responsible for 50% of the total number of focus of bushfires in Brazil in the 10 years analyzed. Results point to a negative correlation between bushfires and dengue that are highly significant in the four states studied (Figure 1).

It is noticeable in the figure that the time-series shows that in years with a high number of focus of bushfires the number of dengue cases is low, and vice versa, independently of the known seasonal variation in the number of dengue cases and bushfires.

Although the correlations found point to a negative association between bushfires and dengue, further studies involving multivariate analysis taking into account potential confounders are necessary to confirm this possible association. In addition, the main hypothesis to explain the negative association found between bushfires and dengue, namely that mosquitoes are killed by the haze resulting from bushfires, should be tested experimentally in the laboratory.

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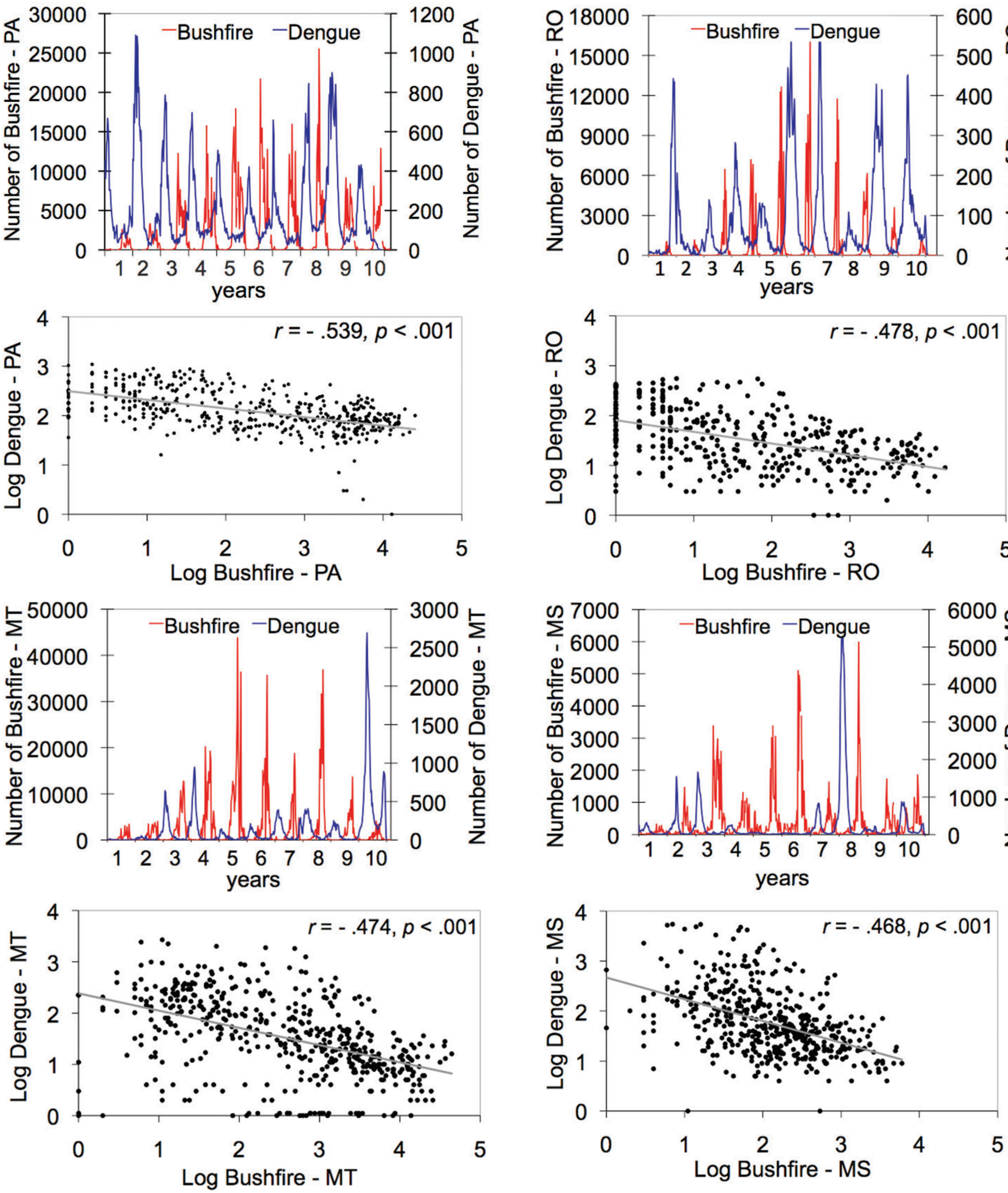
FIGURE **1**

Time Series and Correlations Between Dengue and Bushfires in Four Brazilian States



Red indicates the number of bushfires and blue indicates the number of dengue cases. State abbreviations are Para (PA), Rodonia (RO), MatoGrosso do Sul (MS), and MatoGrosso (MT).

INTERNATIONAL PERSPECTIVES

Prepublished online April 2013,
National Environmental Health Association.

Effectiveness and Acceptance of Total Release Insecticidal Aerosol Cans as a Control Measure in Reducing Dengue Vectors

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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Abstract **The effectiveness of regular application of insecticidal fogging in reducing dengue is questionable, since delays occur between peak time of outbreak and insecticide administrations. Moreover, many residents do not accept indoor application because of concern about insecticide contamination of household items. The study described in this article was designed to evaluate the effectiveness and acceptance of insecticidal aerosol cans to reduce dengue vectors inside and outside of homes. Residents in Kaohsiung City of South Taiwan were provided with two formulations of aerosol cans (permethrin 3.75% weight/weight [w/w] and cypermethrin 1.716% w/w) and were requested to use these aerosol cans. Although the indoor ovitrap index of the permethrin group returned to the original level in week 3, the index of the cypermethrin group decreased 60% to 20%. The residents accepted the insecticidal aerosol cans but complained of unfavorable effects caused by traditional insecticidal fogging. Results indicate that the insecticidal aerosol cans may serve as a supplementary household control measure for dengue vectors during the time period between the peak of outbreak and the administration of government-organized insecticide fogging.**

Introduction

Dengue has emerged as an international public health problem (Jacobs, 2000). This mosquito-borne viral infection is endemic in more than 100 countries of Africa, Central and South America, Asia, and the Western Pacific (Castleberry & Mahon, 2003). Although the etiologic agents (DEN-1, DEN-2, DEN-3, and DEN-4) and principal vectors (*Aedes aegypti* and *Ae. albopictus*) are well studied, no specific management of the clinical cases exists. In addition, vaccines against dengue remain commercially unavailable. Vector control is considered the only strategy to control the transmission of the disease (Guzmán & Kourí, 2002). Elimination of adult mosquitoes in the endemic area is one of the general approaches to dengue control. The World Health Organization (WHO) has documented that space-spray is effective against adult mosquitoes in dengue control activities (WHO, 1996).

As a principal vector, *Ae. aegypti* is able to transmit dengue at temperatures over 20°C. It loses this capability, however, below 16°C (Blanc & Caminopetros, 1930). This species has a flight range of 25 m to over 100 km in open areas and may fly 2.5 km/day (Wolfinsohn & Galun, 1953). *Ae. albopictus* may also transmit dengue (Sabin, 1952). In Taiwan, *Ae. albopictus* distributes throughout the island whereas *Ae. aegypti* is commonly found in the regions south of the Tropic of Cancer. *Ae. aegypti* breeds in artificial water containers such as vessels and discarded tires whereas *Ae. albopictus* breeds in natural ones such as tree holes (Teng, 1996). The abundance of these two species has seasonal variations. *Ae. albopictus* was found to have a significantly higher proportion than *Ae. aegypti* throughout the year (Pai & Lu, 2009).

The first records of dengue in Taiwan can be traced back to 1870. Since then, a number of epidemics have occurred (Ko, 1989). In 1981, an outbreak occurred on

Liouchyou district, an offshore island in southwest Taiwan. The causative agent was identified to be DEN-2 and it was probably imported from the Philippines (Hsieh et al., 1982; Wu, 1986). Between 1987 and 1988, another outbreak due to DEN-1 appeared in South Taiwan and more than 10,000 cases were reported (Ko, 1989). After subsiding for two years, dengue reemerged in 1991, although the severity was milder (Harn, Chiang, Tian, Chang, & Ko, 1993). In addition to the southern part of the island, an outbreak also affected Taipei County of North Taiwan in 1995 (Teng et al., 1998). In June 2002, more than 5,000 cases of dengue fever and 241 cases of dengue hemorrhagic fever were reported in South Taiwan. Due to an extensive history of dengue burden, control measures such as early case detection in sentinel health facilities, early warning systems, and prompt extensive vector control have been applied in this region.

As a general control measure, insecticidal fogging is regularly administered to endemic and/or nonendemic areas in South Taiwan. The effectiveness of this control strategy is questionable, however, since delays occur between peak time of outbreak and insecticide administrations. Many residents refuse to allow government workers to conduct indoor insecticidal fogging because of concern over insecticide contamination of household items. WHO has recommended the use of personal protective measures such as aerosol insecticide spraying at home and in the peridomestic environment (WHO, 1997). In order to evaluate the effectiveness and acceptance of the personal use of insecticidal aerosols as a measure to reduce dengue fever vectors, we provided residents in experimental household groups with different formulations of total release insecticidal aerosol cans. The effectiveness of these formulations and insecticidal fogging in reducing dengue vector populations were compared by ovitrap index. The acceptance of these measures was also evaluated by a questionnaire.

Methods

Study Areas and Insecticidal Aerosol Can Formulations

Our study, conducted in May 2006, was approved by the Department of Environmental Sanitation and Toxic Substance Management, Environmental Protection Administration, Executive Yuan, Republic of China. Three boroughs were selected randomly from each of two randomly selected districts of Kaohsiung City in South Taiwan and divided into three experimental groups. In the first treatment group, insecticidal fogging was administered indoor and outdoor by the government.

In addition to the dengue surveillance measures (Center for Disease Control, Republic of China, 2003) and insecticidal fogging administered outdoor by the government, households in the other two experimental groups were provided with one of two formulations of total release insecticidal aerosol cans (permethrin 3.75% weight/weight [w/w] and cypermethrin 1.716% w/w) for indoor personal administration. The aims, application procedures, and cautions of our study were clearly explained to the participants by well-trained inspectors. These cans were then returned to our laboratory to determine the status of usage.

Application of Insecticidal Aerosol Cans

Before release of the insecticidal aerosol, the residents were requested to keep their windows and doors closed and to cover all food materials and drinking water. The content of each aerosol can was applied to an area of 11 m^2. Each can was placed in the center of the room before pressing the button on the can to release the insecticidal aerosol in about 60 seconds. The windows and doors were then opened 30 minutes later. These cans were then returned to our laboratory to determine the status of usage.

Evaluation of the Immediate Effectiveness of Insecticidal Aerosol Cans

In 20 randomly selected households within the aerosol can experimental groups, three mosquito cages were set up vertically with 70-cm spaces between each cage. Each cage contained 20 female mosquitoes of the Bora Bora strain and local strains of *Ae. aegypti* and *Ae. albopictus* (3–6 days old and reared in the laboratory within three generations) and were positioned in the testing area before insecticidal aerosol application. These insects were prepared according to Lin and co-authors (2003). The knockdown time (KT_{50}) and mortality after 24 hours were determined to evaluate the immediate effectiveness of the insecticidal aerosol cans.

Evaluation of Short-Term Effectiveness of Control Measures

Indoor and outdoor densities of dengue vectors were quantified using an ovitrap index before insecticide application and in weeks 1–3 after intervention for change comparison among participant households. Ovitraps were designed according to Jakob and Bevier (1969). Each trap was a black cylindrical jar with a water-wetted paper strip inside. Traps were placed inside and outside of each household for five days. The ovitrap index was determined for both inside and outside homes as the number of traps with laid eggs/total number of traps × 100.

Questionnaire

Participants in the experimental groups were asked to complete a structural questionnaire to determine the acceptance level of the insecticidal aerosol cans and the insecticidal fogging administered by the government.

Statistical Analysis

KT_{50} was calculated according to Finney (1971). Rates were analyzed using the χ^2 test. Statistical significance was set at $p < .05$.

Results

Immediate Effectiveness of Insecticidal Aerosol Can Application

The immediate effectiveness of insecticidal aerosol cans was quantified using KT_{50} and mortality rates after 24 hours posttreatment in 20 randomly selected households within each experimental group (Table 1). Among mosquitoes of the Bora Bora strain, the overall KT_{50} was shorter in the permethrin group (1.60±0.62 minutes) than in the cypermethrin group (2.02±1.33 minutes). In the local strains of *Ae. aegypti* and *Ae. albopictus*, however, the overall KT_{50} was shorter in the cypermethrin group (*Ae. aegypti* 1.45±1.03 minutes and *Ae. albopictus* 2.94±1.55 minutes) than in the permethrin group (*Ae. aegypti* 1.49±1.02 minutes and *Ae. albopictus* 3.04±1.35 minutes). Mortality rates of all three strains at 24 hours postinsecticide application were 100%.

Long-Term Effectiveness of Control Measures

Short-term residual effects of the two formulations of insecticidal aerosol cans and insecticidal fogging applications were evaluated by determining changes in the ovitrap index in study households before control and weeks 1–3 after insecticide application (Figures 1a and 1b). The indoor ovitrap index of the permethrin group decreased from 60% before the control to 45% in weeks 1 and 2 after control. The index returned to the original level of 60% in week 3. In the cypermethrin group, the index decreased from 65% before control to 5% in week 1 and then increased to 20% in weeks 2 and 3. The insecticidal fogging only group showed an index of 35% before control, decreased to 5% in week 1, and finally increased to 15% in week 3 (Figure 1a).

Outdoor ovitrap indices were higher than the corresponding indoor ones. For the permethrin group, the outdoor index decreased from 90% before control to 65% in week 1 and then increased to 80% in weeks 2 and 3. Before control, the cypermethrin group had an index of 75%. This index then decreased to 25% in week 1 and 5% in week 2 before increasing to 50% in week 3. Before control, the insecticidal fogging only group had an index of 85% that then decreased to 15% in week 1 before returning to 50% and 45% in weeks 2 and 3, respectively (Figure 1b).

TABLE **1**

Knockdown Times (KT_{50}) of Bora Bora Strain, *Aedes aegypti*, and *Aedes albopictus* After Application of Insecticidal Aerosol Cans in 20 Randomly Selected Households

Mosquito Cage	KT_{50} (Minutes)	
	Permethrin (3.75% weight/weight)	Cypermethrin (1.716% weight/weight)
Bora Bora strain		
Upper	1.74±0.69	2.25±1.57
Middle	1.55±0.63	1.82±0.98
Lower	1.50±0.55	2.01±1.43
Overall	1.60±0.62	2.02±1.33
Ae. aegypti		
Upper	1.84±1.20	1.77±1.13
Middle	1.30±0.99	1.30±0.95
Lower	1.38±0.83	1.29±0.99
Overall	1.49±1.02	1.45±1.03
Ae. albopictus		
Upper	3.12±1.33	3.02±1.84
Middle	2.87±1.20	2.72±1.73
Lower	3.11±1.56	2.76±1.75
Overall	3.04±1.35	2.94±1.55

Demographic Characteristics of the Respondents to the Questionnaire

The demographic characteristics of the survey respondents are shown in Table 2. No significant differences were found in the sex, education, occupation, and residence type among the permethrin, cypermethrin, and insecticidal fogging only groups. Only one household in the permethrin group and nine households in the insecticidal fogging group reported having a history of dengue fever/dengue hemorrhagic fever in family members.

Application of Insecticides in Households

Among 600 households surveyed, 282 (47.3%) used household insecticide products without a definite time schedule and 194 (32.6%) did not use any insecticides within their homes. Regular personal usage of insecticides was found only in 120 (20.1%) of the households. Only 284 (48.1%) households received routine indoor and outdoor administration of insecticides by the local government and 193 (32.3%) reported having had outdoor administration only. A total of 117 (19.6%) households did not receive any insecticide administration from the local government. Of those households that were approached by the local government during vector control campaigns, only 182 (30.7%) cooperated with indoor and outdoor administration of insecticides by the government and 274 (46.2%) cooperated for outdoor application only. A total of 20 (3.4%) households surveyed did not cooperate with insecticide administration by the government.

Unfavorable Effects Caused by Regular Insecticide Administration

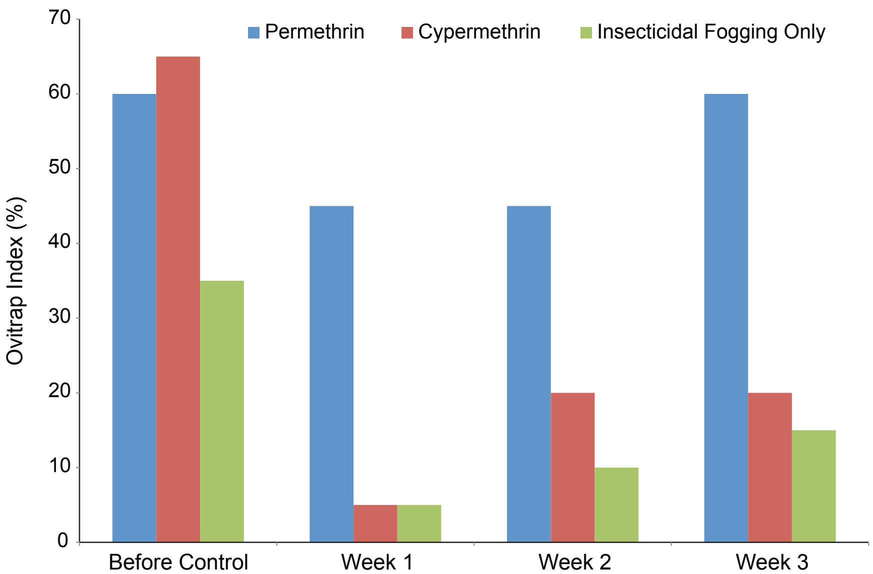
Although 194 (32.3%) respondents did not complain after receiving insecticide administration by the local government, 229 (38.2%) considered the odor unacceptable, and 111 (18.5%) reported the contamination of insecticide not easy to clean. In addition, 88 (14.7%) felt inconvenienced by the insecticide administration team (i.e., waiting for arrival, etc.). Chemical wetting of the floor and potential for causing illness were complaints from 45 (7.5%) and 26 (4.3%) respondents, respectively. Additional complaints consisted of causing damage to furniture (1.2%), damage to the floor (1%), death of aquarium fish (0.5%), and death of pets (0.5%).

Acceptance of Insecticidal Aerosol Cans

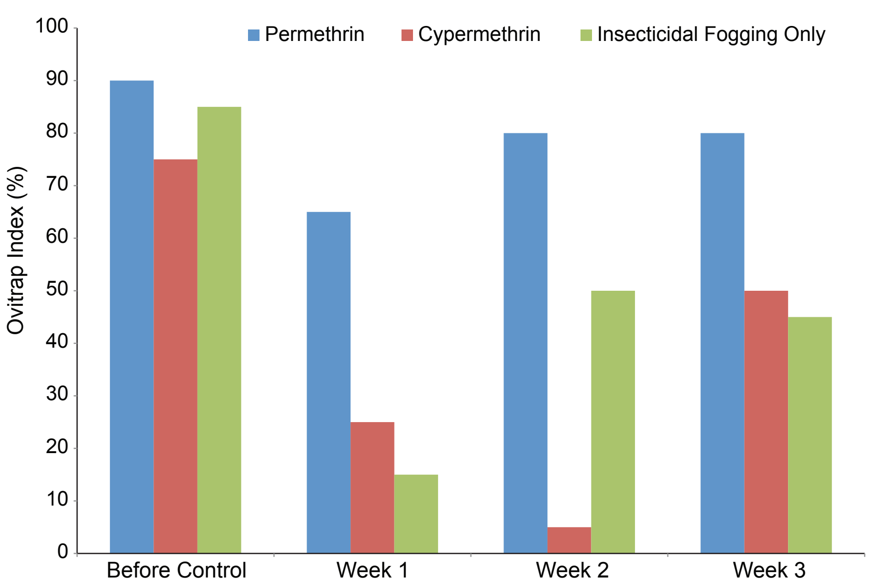
Although only six (3.0%) respondents in the permethrin group and 24 (1.2%) in the cypermethrin group had heard of the insecticidal aerosol cans, they applied the two formulations of aerosol cans to their households. The number of empty cans returned was found to match the number of cans provided. After application, 166 (83.0%) respondents/households in the permethrin group and 110 (55.0%) in the cypermethrin group preferred the insecticidal aerosol cans to the regular insecticide administration by the government.

FIGURE 1

a. Indoor Ovitrap Index of Dengue Vectors Before and After Control by Total Release Insecticidal Aerosol Cans



b. Outdoor Ovitrap Index of Dengue Vectors Before and After Control by Total Release Insecticidal Aerosol Cans



In the experimental groups, formulations of permethrin (3.75% weight/weight [w/w]) and cypermethrin (1.716% w/w) were administrated. Households receiving only insecticidal fogging administered by the environmental protection unit were used as a control.

Unacceptable Reasons for the Insecticidal Aerosol Cans

Of the survey respondents in the experimental groups, 175 (87.5%) in the permethrin group and 125 (62.5%) in the cypermethrin group did not have reasons to refuse the insecticidal aerosol cans (Table 3). Forty-three (21.5%) respondents had complaints of unacceptable odor in the cypermethrin group, significantly higher than the five (2.5%) respondents in the permethrin group ($p < .05$). The percentages of the remaining reasons were lower than or equal to 5%.

Discussion

In a previous study, we determined that the abundance of dengue vectors peaked in May, June, and September in the endemic districts and in May and October in non-endemic districts in Kaohsiung (Pai & Lu, 2009). Therefore, we conducted the evaluation of the effectiveness and acceptance of the insecticidal aerosol cans in May. Our current study was conducted in two nonendemic districts of this city.

Insecticidal aerosol cans have been previously evaluated to be effective and feasible for dengue fever control in southern Vietnam. The reduction rate in the number of dengue hemorrhagic fever cases was significantly higher in the study area than in the control area. The cost of the insecticidal aerosol cans was also lower than for ultralow-volume fogging (Osaka, Ha, Sakakihara, Khiem, & Umenai, 1999).

In our current study, we evaluated two formulations of insecticidal aerosol cans. These formulations were not only demonstrated to have immediate effects to dengue vectors using KT_{50} and 24-hour mortality rates as indicators, but also demonstrated short-term effects in reducing the abundance of vectors after application. These findings signified their effectiveness in the chemical control of vectors harboring the dengue virus. Although permethrin (3.75% w/w) was found to be effective to reduce the ovitrap indices in week 1 after control, the abundance of dengue vectors returned to the original levels in week 3. The effect of cypermethrin (1.716% w/w) to reduce ovitrap indices was more residual than permethrin (Figure 1). The KT_{50} of cypermethrin was also shorter than permethrin. These findings may be due to the existence of resistance to permethrin among mosquitoes in South Taiwan. This suggestion requires further experimental evidence, however.

To prevent the transmission of dengue, community-based control projects have been administered in different countries (Chiaravalloti Neto, de Moraes, & Fernandes, 1998; Kay et al., 2002; Kroeger et al., 1995; Leontsini, Gil, Kendall, & Clark, 1993; Reiter et al., 1994; Van Benthem et al., 2002). The effectiveness of these campaigns is strongly dependent on the disease knowledge and behavior of community members. Despite this, education programs can be implemented to not only increase dengue-related knowledge and generate awareness of the importance of vector control for preventive measures against dengue (Espinoza-Gomez, Hernandez-Suarez, & Coll-Cardenas, 2002; Fajardo, Monje, Lozano, Realpe, & Hernandez, 2001; Madeira, Macharelli, Pedras, & Delfino, 2002; Winch et al., 2002). Even a short-term community-based cleanliness educational program may change the dengue-related behavior of people and reduce the abundance of dengue vectors (Pai, Hong, & Hsu, 2006).

The insecticidal aerosol cans evaluated in our current study are products that may supplement such community-based campaigns. Although insecticidal fogging is effective in reducing indoor and outdoor abundance of dengue vectors, a portion of the study households considered the odor unacceptable and the contamination of insecticide not easy to clean. The residents also felt inconvenienced in waiting for the insecticide administration team during scheduled indoor fogging applications. These findings indicate that household products, such as the aerosol cans, should be well received in this community. In fact, most of the participants were not aware of the insecticidal aerosol cans prior to our study, but were willing to apply the product within their homes according to our instructions. In addition to their cooperation, participants were accepting of the new product as an alternative to the regular insecticide administration by the government, although a portion of residents did consider cypermethrin to have an unacceptable odor. Improvement of odor may be modified, however, by improving the formulation in future postproduct development.

TABLE **2**

Demographic Characteristics of Respondents to the Acceptance of Insecticidal Aerosol Cans Against Dengue Transmission

Factor	Permethrin Group (n = 200)		Cypermethrin Group (n = 200)		Insecticidal Fogging Only Group (n = 200)	
	#	%	#	%	#	%
Sex						
Male	90	45.0	98	49.0	74	37.2
Female	110	55.0	102	51.0	125	62.8
Education						
Secondary or below	162	81.0	122	61.0	116	58.0
College or above	34	17.0	37	18.5	50	25.0
Occupation						
Household affairs	68	34.0	58	29.0	63	32.1
Others	128	64.0	124	62.0	133	67.9
Type of residence						
Single family	147	73.5	102	51.0	157	78.5
Apartment	53	26.5	56	28.0	43	21.5
Family member with history of dengue fever/dengue hemorragic fever						
Yes	1	0.5	0	–	9	4.6
No	199	99.5	200	100.0	191	95.4

TABLE **3**

Unacceptable Reasons for Insecticidal Aerosol Cans in the Survey Households

Reason	Permethrin Group (n = 200)		Cypermethrin Group (n = 200)	
	#	%	#	%
No unacceptable reasons	175	87.5	125	62.5
Using the cans only once is wasteful	0	–	10	5.0
Unacceptable smelling	5	2.5	43	21.5*
The traditional method is good	1	0.5	4	2.0
Contamination not easy to clean	3	1.5	0	–
May cause damage to children, plants, and pets	1	0.5	5	2.5
This new product may not ensure safety	1	0.5	1	0.5
Troublesome	10	5.0	7	3.5
Unnecessary	3	1.5	5	2.5

*χ^2 test: $p < .05$.

Conclusion

The insecticidal aerosol cans when used in combination with outdoor fogging were effective in reducing the abundance of dengue vectors both inside and outside of homes. The insecticidal aerosol cans may be applied as a supplementary chemical control measure for dengue vectors during the time period between the peak of clinical case outbreak and administration of insecticide fogging. The insecticidal aerosol cans may also be used as a personal protective measure in reducing the abundance of dengue indoors in endemic areas.

Acknowledgements: This study was supported in part by a grant (DOH95-DC-1039) from the Center for Disease Control, Department of Health, Executive Yuan, Republic of China.

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continued on page 74

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▸ **INTERNATIONAL PERSPECTIVES**

Prepublished online May 2013,
National Environmental Health Association.

Acute Air Pollution–Related Symptoms Among Residents in Chiang Mai, Thailand

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

Phongtape Wiwatanadate, MD, PhD
Department of Community Medicine
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Abstract **Open burnings (forest fires, agricultural, and garbage burnings) are the major sources of air pollution in Chiang Mai, Thailand. A time series prospective study was conducted in which 3,025 participants were interviewed for 19 acute symptoms with the daily records of ambient air pollutants: particulate matter less than 10 µm in size (PM_{10}), carbon monoxide (CO), nitrogen dioxide (NO_2), sulfur dioxide (SO_2), and ozone (O_3). PM_{10} was positively associated with blurred vision with an adjusted odds ratio (*OR*) of 1.009. CO was positively associated with lower lung and heart symptoms with adjusted *OR*s of 1.137 and 1.117. NO_2 was positively associated with nosebleed, larynx symptoms, dry cough, lower lung symptoms, heart symptoms, and eye irritation with the range of adjusted ORs (ROAORs) of 1.024 to 1.229. SO_2 was positively associated with swelling feet, skin symptoms, eye irritation, red eyes, and blurred vision with ROAORs of 1.205 to 2.948. Conversely, O_3 was negatively related to dry cough, red eyes, and blurred vision with ROAORs of 0.891 to 0.979.**

Introduction

The air pollution in northern Thailand has been well recognized as a smog crisis from January to April every year since 2007. For example, the Pollution Control Department's monitoring stations in the northern areas showed that in March 2010, the levels of particulate matter less than 10 µm in aerodynamic diameter (PM_{10}) exceeded Thailand's 24-hour average standard level of 120 µg/m^3 (Pollution Control Department, Thailand, 2012) every day. The highest level was on March 18, 2010, at 518.5 µg/m^3, which is the all-time record (Pollution Control Department, Thailand, 2011).

Much research has been well documented showing the adverse effects of air pollution such as asthma, chronic obstructive pulmonary disease, ischemic heart diseases, congestive heart failure, heart rhythm disorders, and diabetes (Brunekreef & Holgate, 2002; Forastiere, D'Ippoliti, & Pistelli, 2002; Pope & Dockery, 2006; Wiwatanadate & Trakultivakorn, 2010). Studies have been conducted to assess the effects of pollutants on the respiratory symptoms (Bayer-Oglesby et al., 2006; Cesaroni, Badaloni, Porta, Forastiere, & Perucci, 2008; Forsberg, Stjernberg, & Wall, 1997; Garshick, Laden, Hart, & Caron, 2003; Lai et al., 2010; Lee et al., 2005; Schindler et al., 2009), heart rate (Pope et al., 1999), and ophthalmological symptoms (Bourcier et al., 2003). Nearly all of these studies investigated the health effects due to exposure to traffic- or industry-related pollution. Other studies have shown the effects of smoke from forest fires on the increased rates of hospitalization (Ignotti et al., 2010; Mott et al., 2005) or seeking care in various groups (Lee, Falter, Meyer, Mott, & Gwynn, 2009; Moore et al., 2006). Air pollution from forest fire smoke also elicits inflammation within the lungs in firefighters (Swiston et al., 2008). The summary of epidemiological studies of short-term health effects of air pollutants is shown in Table 1.

No studies thus far have assessed the acute effects of open burning–related air pollution in a general adult population in Chiang Mai. In this study, the aim was to assess the effects of air pollutants on acute symptoms of participants residing in Mae Rim district, Chiang Mai, Thailand.

Materials and Methods

Design and Study Participants

A time series prospective study was chosen as the design, which involved daily interviewing about air pollution–related symptoms in a general population and daily records of air quality and meteorological parameters. It was conducted in Chiang Mai, which is one of the largest cities in Thailand, located 696 km north of Bangkok. Chiang Mai's population in 2012 was around 1.7 million people distributed in 24 administrative districts, including Mae Rim district (the study area; see Figure 1).

TABLE 1

Summary of Epidemiological Studies of Short-Term Health Effects of Air Pollutants

Authors	Health Effects	Location of Study	Period of Study	Pollutants[b]
Gold et al. (2000)	Heart rate variability[a]	Boston	Summer	$PM_{2.5}$ and O_3
Schwartz et al. (2005)	Heart rate variability[a]	Boston	Summer	$PM_{2.5}$ and CO
Park et al. (2005)	Heart rate variability[a]	Boston	All seasons	$PM_{2.5}$ and O_3
Riojas-Rodriguez et al. (2006)	Heart rate variability[a]	Mexico City	All seasons	CO
Holguin et al. (2003)	Heart rate variability[a]	Mexico City	All seasons	$PM_{2.5}$ and CO
Chan et al. (2005)	Heart rate variability[a]	Urban	Winter	NO_2
Min et al. (2009)	Heart rate variability[a]	Tae-in Island, South Korea	Winter	CO
Wheeler et al. (2006)	Heart rate variability[a]	Atlanta	Fall and spring	$PM_{2.5}$ and NO_2
Peters et al. (1999)	Heart rate	Augsburg, Germany	All seasons	CO
Liao et al. (2004)	Heart rate	Suburban	All seasons	CO
Metzger et al. (2007)	Arrhythmia	Urban	Winter	SO_2
Dockery et al. (2005)	Arrhythmia	Boston	All seasons	CO
Peters et al. (2000)	Arrhythmia	Eastern Massachusetts	All seasons	CO
Berger et al. (2006)	Arrhythmia	Erfurt, Germany	Winter	CO
Schwartz et al. (1994)	Cough	Urban	Summer	NO_2
Mortimer et al. (2002)	Cough, wheeze, shortness of breath	Urban	All seasons	O_3, SO_2, NO_2, and PM_{10}
Schildcrout et al. (2006)	Cough, asthmatic symptoms	Urban	All seasons	CO and NO_2
van der Zee et al. (2000)	Upper respiratory symptoms	Urban	Winter	PM_{10}
Harré et al. (1997)	Chest symptoms	Urban	Winter	PM_{10}
Ségala et al. (2004)	Rhinorrhea and cough	Urban	Winter	NO_2 and PM_{10}

[a]Heart rate variability refers to the beat-to-beat alterations in the heart rate.
[b]$PM_{2.5}$ = particulate matter less than 2.5 µm in size; O_3 = ozone; CO = carbon monoxide; NO_2 = nitrogen dioxide; SO_2 = sulfur dioxide; PM_{10} = particulate matter less than 10 µm in size.

Since Mae Rim has been annually affected by the smog crisis, especially from January to April, and it has an air quality monitoring station installed at the center of the district, the participants were recruited with the following eligibility criteria: (1) age greater than 14 years old; (2) nonsmoker; and (3) had lived in Mae Rim for more than one year. The households in the area were used as the sampling unit. Each day, starting January 1 until April 30, 2008, 25 households were randomly selected without replacement; then, one member in that household who met the above criteria was randomly selected in the field for an interview by a trained research assistant of current day's 19 symptoms (answering as "yes" or "no"). Because of the time constraint, if no eligible member was available, an adjacent household in the radius of 100 m would replace it. The participants were selected on a non-duplicable basis to avoid the autocorrelation. All participants lived within a 10-km radius of the air quality monitoring station; the majority lived within 5 km.

The study protocol was approved by the research ethics committee of the Faculty of Medicine, Chiang Mai University. An informed written consent was obtained from all subjects prior to the interview. If the subjects were less than 20 years old, an informed written consent was obtained from their legal guardians.

Measurements of Air Pollutants and Meteorological Parameters

Ambient air concentrations of pollutants were measured by the Pollution Control Department, Ministry of National Resources and Environment with the continuous automated air sampling monitoring station located at the center of the city. The analysis method for each pollutant was as follows: for carbon monoxide (CO) concentrations, nondispersive infrared detection; for sulfur dioxide (SO_2) concentrations, pararosaniline technique; for nitrogen dioxide (NO_2) and ozone (O_3) concentrations, chemiluminescence technique; and for PM_{10}, gravimetric technique (Air Quality and Noise Management Bureau, 2004). All techniques were certified by the U.S. Environmental Protection Agency. The calibration of the instruments was performed every 15 days. The quality control of the instrumental maintenance was guaranteed to accurately report all parameters not less than 90% of the total days of monitoring. The meteorological data, i.e., pressure, temperature, relative humidity, visibility, and wind speed, were obtained from the Northern Meteorology Center, Chiang Mai province on a daily 24-hour-average basis. The wind direction during the study period was mostly from the northeast.

Statistical Analysis Methods

Since some redundancy might occur in the dependent variables, i.e., some of the symptoms are correlated with one another possibly because they are measuring the same construct, the principal component analysis (PCA) was applied. PCA is a statistical technique that transforms the original data matrix into new variables, the so-called principal components (PC), which are orthogonal (uncorrelated) to each other (McNabola, Broderick, & Gill, 2009). Although PCA is mostly used for reducing the dimensionality associated to multivariate analyses, it was applied in this study as a nonparametric method of classification, in order to classify the symptoms into groups (PCs). The binary logistic regression was then used to determine the associations among each symptom and air pollutants and meteorological parameters. The relative risks were estimated using odds ratios (*OR*) with 95% confidence intervals (*CI*). Since all air pollutants and meteorological variables were recorded as a time series and it is assumed that they had lagged (or delayed) effects on the symptoms, the analyses were conducted by performing a separate univariate analysis of each air pollutant and meteorological parameter at lags 0–6 with each of the symptoms. The best lagged effect, i.e., the least *p*-value, (Wiwatanadate & Liwsrisakun, 2011; Wiwatanadate & Trakultivakorn, 2010) was entered in the next step along with key outcome-related personal (gender and age) and temporal (date [time trend] and day of week) factors as well as the interaction terms between each of the pollutants and meteorological parameters that were found significantly correlated. The forward stepwise likelihood ratio test—the stepwise selection method with entry testing (*p*-value $< .05$) based on the significance of the score statistic, and removal testing (*p*-value $> .1$) based on the probability of a likelihood-ratio statistic based on the maximum partial likelihood estimates—was subsequently applied to each model to select the final model (Kleinbaum & Klein, 2002). The statistical software used in the analyses was the R version 2.8.1.

FIGURE **1**

Study Area and Locations Where the Majority of Participants Live

Photo retrieved from DigitalGlobe/Google Earth on December 25, 2010.

Results

Demographic and Characteristics of Participants

Table 2 shows the details. The number of the subjects participating in the study was 3,025 (1,001 males and 2,024 females). Their median age was 51 years with a range of 15–91 years. The top five highest percentages of symptoms found in the survey were stuffy nose (34.8%), tiredness (33.7%), body itch (32.9%), burning or itching eyes (30.7%), and running nose (28.7%).

Description of Ambient Exposure Measurement

Table 3 indicates the exposure parameters. Due to air quality monitoring equipment malfunction, one out of 121 days (0.8%) was missing for the PM_{10}, CO, O_3, NO_2, and SO_2 data. Three out of 120 days (2.5%) had a 24-hour average of PM_{10} that exceeded the current Thailand's National Ambient Air Quality Standards (NAAQS) of 120 µg/m^3. The SO_2 concentrations were within Thailand NAAQS limit of 120 parts per billion (ppb). Pearson correlation coefficients among air pollutants and meteorological parameters are shown in Table 4. The strongest correlation was between visibility and PM_{10} with a coefficient of –0.59.

Principal Component Analysis

PCA was applied to the 19 pollutant-related symptoms to assist in grouping of symptoms. At first, Kaiser's criterion was adopted to decide the tentative number of factors to be retained (only factors with eigenvalues >1) (Kaiser, 1958), and only six factors met the criteria with the total variance explained of 52.9%. Six more factors were added, however, because their initial eigenvalues were close to 1 (0.800–0.980). Keeping the 12 factors in the analysis, accounting for the total variance of 80.7%, would facilitate in grouping symptoms effectively. Variables with high loadings (weightings) indicate a strong interrelationship; the loading would be considered high if its coefficient is 0.5 or greater (McNabola et al., 2009). As shown in Table 5, for the first component, the symptoms with high correlations were stuffy nose, running nose, and burning nose; therefore, these three were grouped as "rhinitic symptoms." By the same token, the body itch and body rash in the second component were grouped as "skin symptoms"; the burning throat and hoarseness in the third component were grouped as "larynx symptoms"; the burning or itching eyes and watery eyes in the fourth component were grouped as "eye irri-

TABLE **2**

Descriptive Statistics for 3,025 Study Participants in Mae Rim District, Chiang Mai, Thailand, During Study Period

Subject Characteristic	Value
Median age (age range, years)	51 (15–91)
# of Males/females	1001/2024
# With symptoms (%)	
Stuffy nose	1052 (34.8)
Running nose	869 (28.7)
Burning nose	594 (19.6)
Nosebleed	9 (0.3)
Burning throat	632 (20.9)
Hoarseness	382 (12.6)
Dry cough	681 (22.5)
Productive cough	541 (17.9)
Shortness of breath	453 (15.0)
Whistling breath	148 (4.9)
Tiredness	1019 (33.7)
Swelling feet	27 (0.9)
Palpitation	171 (5.7)
Body itch	996 (32.9)
Body rash	397 (13.1)
Burning or itching eyes	930 (30.7)
Red eyes	37 (1.2)
Watery eyes	426 (14.1)
Blurred vision	566 (18.7)

TABLE **3**

Daily Meteorological and Air Pollution Measurements in Mae Rim District, Chiang Mai, Thailand, During Study Period

Exposure (24-Hour Average)	Observations (#)	Minimum/ Maximum	Mean (*SD*)	90th Percentile	# Days Standards Exceeded (%)
Pressure (hPa)	121	1003.65/1044.50	1010.54 (4.42)	1014.50	–
Temperature (°C)	121	16.50/37.50	26.66 (4.11)	32.40	–
Relative humidity (%)	121	18.00/89.00	44.27 (12.15)	59.70	–
Visibility (km)	121	2.00/15.00	8.57 (2.26)	12.00	–
Wind speed (km/h)	121	0.00/24.10	6.76 (5.12)	14.64	–
PM_{10} (μg/m^3)[a]	120	17.00/171.30	61.62 (25.95)	94.40	3 (2.5)[b]
CO (parts per million)[a]	120	0.10/9.00	0.72 (0.80)	0.90	–[c]
O_3 (parts per billion [ppb])[a]	120	10.40/54.30	31.29 (8.89)	42.57	–[c]
NO_2 (ppb)[a]	120	0.50/27.30	10.89 (4.53)	16.36	–[c]
SO_2 (ppb)[a]	120	0.00/2.50	0.48 (0.53)	1.20	0 (0.0)[d]

[a]Missing because of equipment malfunction. PM_{10} = particulate matter less than 10 μm in size; CO = carbon monoxide; O_3 = ozone; NO_2 = nitrogen dioxide; SO_2 = sulfur dioxide.
[b]According to Thailand's National Ambient Air Quality Standards (NAAQS) = 120 μg/m^3.
[c]According to Thailand NAAQS, 24-hour average standards are not available.
[d]According to Thailand NAAQS = 120 ppb.

tation symptoms"; the shortness of breath and whistling breath in the fifth component were grouped as "lower lung symptoms"; the tiredness and palpitation in the sixth component were grouped as "heart symptoms." Other symptoms in components 7–12: red eyes, swelling feet, nosebleed, productive cough, blurred vision, and dry cough were kept "as is." The new variables were then recoded to "1" if any of the original variables was coded "1" and was recoded to "0" if and only if all of the original variables were coded "0."

Association of Exposure and Symptoms

Table 6 shows the analytic results of adjusted *OR*s with 95% *CI* of parameters statistically associated with symptoms. Regarding the pollutants, PM_{10} was significantly positively associated with blurred vision with adjusted *OR*s of 1.009 (95% *CI* = 1.004 to 1.014) per 1 μg/m^3. The CO concentrations were significantly positively associated with lower lung symptoms and heart symptoms with adjusted *OR*s of 1.137 (95% *CI* = 1.027 to 1.258) and 1.117 (95% *CI* = 1.018 to 1.224) per 1 part per million, respectively. The NO_2 concentrations were significantly positively associated with nosebleed, larynx symptoms, dry cough, lower lung symptoms, heart symptoms, and eye irritation with adjusted *OR*s of 1.229 (95% *CI* = 1.070 to 1.412), 1.044 (95% *CI* = 1.023 to 1.065), 1.034 (95% *CI* = 1.014 to 1.055), 1.039 (95% *CI* = 1.015 to 1.063), 1.024 (95% *CI* = 1.007 to 1.041), and 1.051 (95% *CI* = 1.032 to 1.070) per 1 part per billion (ppb), respectively.

The SO_2 concentrations were significantly positively associated with swelling feet, skin symptoms, eye irritation, red eyes, and blurred vision with adjusted *OR*s of 2.416 (95% *CI* = 1.233 to 4.731), 1.205 (95% *CI* = 1.035 to 1.402), 1.264 (95% *CI* = 1.085 to 1.471), 2.948 (95% *CI* = 1.463 to 5.940), and 1.210 (95% *CI* = 1.004 to 1.459) per 1 ppb, respectively. In contrast, the O_3 concentrations were significantly negatively associated with dry cough, red eyes, and blurred vision with adjusted *OR*s of 0.979 (95% *CI* = 0.968 to 0.990), 0.891 (95% *CI* = 0.847 to 0.938), and 0.962 (95% *CI* = 0.947 to 0.977) per 1 ppb, respectively. Furthermore, the synergistically multiplicative effects of pollutants could be seen on the lower lung symptoms (NO_2 and CO), heart symptoms (NO_2 and CO), eye irritation (SO_2 and NO_2), and blurred vision (SO_2 and PM_{10}).

Discussion

Some of the findings in this study are consistent with those of the previous studies showing that traffic-related pollutants were related to breathlessness (Bayer-Oglesby et al., 2006); wheeze (Bayer-Oglesby et al., 2006; Garshick et al., 2003); bronchitic symptoms, i.e., productive cough (Bayer-Oglesby et al., 2006; Forsberg et al., 1997; Garshick et al., 2003; Lai et al., 2010); throat and nose irritation

TABLE **4**

Correlation Matrix of Ambient Air Pollutants and Meteorological Parameters, Mae Rim District, Chiang Mai, Thailand, During Study Period

Factor	Temp[a]	RH[a]	Visibility	WS[a]	PM_{10}[a]	CO[a]	O_3[a]	NO_2[a]	SO_2[a]
Pressure	–0.36**	0.25**	0.08	0.01	–0.04	0.06	–0.11	–0.09	–0.15
Temp		–0.19*	–0.28**	–0.06	0.16	–0.07	0.46**	0.09	0.27**
RH			0.12	0.15	–0.33**	–0.10	–0.22*	–0.22*	–0.20*
Visibility				–0.13	–0.59**	–0.17	–0.38**	–0.34**	–0.25**
WS					–0.09	0.00	–0.17	0.14	0.00
PM_{10}						0.35**	0.53**	0.57**	0.39**
CO							0.08	0.16	–0.01
O_3								0.13	0.30**
NO_2									0.17

[a]Temp = temperature; RH = relative humidity; WS = wind speed; PM_{10} = particulate matter less than 10 µm in size; CO = carbon monoxide; O_3 = ozone; NO_2 = nitrogen dioxide; SO_2 = sulfur dioxide.
*Correlation is significant at $p < .05$.
**Correlation is significant at $p < .01$.

TABLE **5**

Factor Loadings for the First 12 Rotated Components and Percentage of Variance Explained

Symptom	Component[a]											
	1	2	3	4	5	6	7	8	9	10	11	12
Stuffy nose	**0.833**	0.018	0.130	0.094	0.155	0.064	–0.010	0.021	0.015	0.109	0.007	0.080
Running nose	**0.831**	–0.017	0.088	0.017	0.051	0.041	0.038	0.008	0.034	0.226	0.028	0.095
Burning nose	**0.598**	0.046	0.402	0.155	0.165	0.040	0.000	–0.001	–0.020	–0.311	0.072	–0.130
Nosebleed	0.033	–0.016	0.007	–0.009	–0.010	0.016	–0.005	–0.005	**0.996**	–0.011	0.004	–0.005
Burning throat	0.223	0.059	**0.768**	0.115	0.129	0.100	–0.017	–0.023	–0.021	0.053	0.008	0.037
Hoarseness	0.092	–0.006	**0.810**	0.020	0.114	0.060	0.036	0.053	0.030	0.132	0.043	0.136
Dry cough	0.104	0.036	0.147	0.082	0.063	0.033	0.013	0.011	–0.005	–0.066	0.037	**0.955**
Productive cough	0.218	0.018	0.171	0.046	0.101	0.074	–0.015	–0.020	–0.014	**0.874**	0.011	–0.080
Shortness of breath	0.255	0.006	0.190	0.145	**0.718**	0.180	–0.021	–0.016	–0.016	0.034	0.043	–0.008
Whistling breath	0.054	0.040	0.094	0.011	**0.885**	0.080	0.027	0.042	0.003	0.062	0.026	0.070
Tiredness	0.071	0.093	0.122	0.098	0.160	**0.672**	0.032	0.024	–0.043	0.187	0.162	0.049
Swelling feet	0.023	–0.018	0.026	0.036	0.028	0.019	0.020	**0.994**	–0.005	–0.017	0.024	0.010
Palpitation	0.043	0.028	0.043	0.048	0.079	**0.873**	–0.003	0.000	0.052	–0.077	–0.014	–0.007
Body itch	0.031	**0.874**	–0.020	0.061	–0.024	0.091	0.008	0.004	–0.015	0.026	0.022	0.050
Body rash	–0.013	**0.878**	0.070	0.049	0.069	0.015	0.014	–0.022	–0.001	–0.011	0.022	–0.015
Burning or itching eyes	0.117	0.062	0.058	**0.843**	0.054	0.088	–0.032	–0.036	–0.036	–0.035	0.046	0.036
Red eyes	0.022	0.021	0.016	0.099	0.009	0.023	**0.988**	0.020	–0.005	–0.012	0.031	0.012
Watery eyes	0.037	0.052	0.080	**0.814**	0.070	0.046	0.142	0.078	0.026	0.074	0.091	0.047
Blurred vision	0.053	0.044	0.050	0.133	0.057	0.115	0.032	0.025	0.005	0.007	**0.970**	0.036
Eigenvalues	3.735	1.687	1.331	1.217	1.056	1.020	0.980	0.921	0.906	0.862	0.810	0.800
% of Variance	19.658	8.879	7.006	6.405	5.559	5.368	5.158	4.846	4.766	4.536	4.265	4.210
Cumulative %	19.658	28.538	35.544	41.949	47.509	52.876	58.034	62.880	67.646	72.182	76.447	80.657

[a]Bold number indicates the principal component of high significance (>0.5).

TABLE **6**

Adjusted Odds Ratios (*OR*s) (95% Confidence Interval [*CI*]) of Parameters Statistically Associated With Symptoms, Mae Rim District, Chiang Mai, Thailand, During Study Period

Symptoms	Parameters[a]	Lagged (Days)	Adjusted *OR*s (95% *CI*)[b]
Rhinitic symptoms[c]	Date (time trend)	–	0.993 (0.990–0.995)
	Visibility	3	0.945 (0.912–0.978)
	Humidity	1	0.988 (0.982–0.995)
Nosebleed	NO_2	6	1.229 (1.070–1.412)
	Wind speed	3	0.779 (0.630–0.964)
Larynx symptoms[c]	NO_2	0	1.044 (1.023–1.065)
	Visibility	6	0.940 (0.903–0.978)
	Humidity	2	0.988 (0.979–0.997)
	Pressure	6	1.079 (1.047–1.112)
	Wind speed	2	0.971 (0.953–0.990)
Dry cough	Age	–	1.009 (1.003–1.015)
	NO_2	0	1.034 (1.014–1.055)
	O_3	2	0.979 (0.968–0.990)
	Temperature	0	0.944 (0.920–0.968)
	Wind speed	3	0.959 (0.941–0.979)
Productive cough	Gender (male)	–	1.348 (1.098–1.654)
	Age	–	1.019 (1.012–1.025)
	Visibility	1	0.952 (1.000–1.220)
	Pressure	0	1.024 (0.912–0.994)
Lower lung symptoms[c]	Date (time trend)	–	0.994 (0.991–0.998)
	NO_2	1	1.039 (1.015–1.063)
	CO	3	1.137 (1.027–1.258)
	Humidity	2	0.985 (0.974–0.996)
	Wind speed	0	0.971 (0.949–0.993)
Heart symptoms[c]	Age	–	1.011 (1.005–1.016)
	NO_2	0	1.024 (1.007–1.041)
	CO	3	1.117 (1.018–1.224)

Symptoms	Parameters[a]	Lagged (Days)	Adjusted *OR*s (95% *CI*)[b]
Swelling feet	SO_2	3	2.416 (1.233–4.731)
	Visibility	3	1.318 (1.106–1.570)
	Wind speed	6	0.850 (0.761–0.949)
Skin symptoms[c]	Age	–	1.006 (1.000–1.011)
	SO_2	3	1.205 (1.035–1.402)
	Temperature	2	1.037 (1.016–1.059)
	Wind speed	0	1.033 (1.017–1.049)
Eye irritation[c]	Gender (male)	–	0.777 (0.652–0.925)
	Age	–	1.015 (1.010–1.021)
	SO_2	3	1.264 (1.085–1.471)
	NO_2	4	1.051 (1.032–1.070)
	Pressure	6	1.038 (1.007–1.070)
	Temperature	0	0.964 (0.941–0.987)
Red eyes	Date (time trend)	–	0.977 (0.955–0.999)
	SO_2	2	2.948 (1.463–5.940)
	O_3	3	0.891 (0.847–0.938)
	Temperature	6	1.225 (0.930–1.000)
	Wind speed	3	0.879 (1.081–1.388)
Blurred vision	Age	–	1.035 (1.028–1.042)
	SO_2	2	1.210 (1.004–1.459)
	O_3	2	0.962 (0.947–0.977)
	PM_{10}	1	1.009 (1.004–1.014)
	Relative humidity	5	0.975 (0.965–0.985)
	Pressure	6	1.064 (1.028–1.102)

[a]NO_2 = nitrogen dioxide; O_3 = ozone; CO = carbon monoxide; SO_2 = sulfur dioxide; PM_{10} = particulate matter less than 10 μm in size.
[b]All models were analyzed with the binary logistic regression using forward stepwise likelihood ratio test as a model selection method.
[c]Regrouped according to principal component analysis.

(Forsberg et al., 1997); rhinitis, i.e., running nose and itching nose (Cesaroni et al., 2008; Forsberg et al., 1997); increased heart rate (Pope et al., 1999); and conjunctivitis, i.e., burning or itching eyes and red eyes (Bourcier et al., 2003). Furthermore, this study revealed that the pollutants can cause the same health effects as a synergism. For example, SO_2 and NO_2 synergistically increase risk of eye irritation as multiplicative effects. In addition, the individual pollutant is able to cause more than one symptom of the same organ or multiorgan systems.

The finding that pollutants have lagged effects means that they can be deemed as any biological agent that has an incubation period. We can make use of this finding to predict the symptoms beforehand. Additionally, the model expressions can be used to calculate the number of the population who will get sick once the pollutant level is known.

For those negative relationships between symptoms and ozone, they might be caused by the poor control of other confounding factors such as indoor air pollutants (cooking smoke, chemicals, tobacco smoke, etc.); environmental pollens; other pollutants, e.g., volatile organic compounds; and participants' underlying diseases. By contrast, it is possible that low-dose exposure might be a protective factor. This can be compared to the fact that some toxic agents might be beneficial to health at low dose; for example, bacterial endotoxins might help reduce the risk of asthma in children (Obihara, Kimpen, & Beyers, 2007; Sordillo et al., 2010) and lifelong farm exposure may strongly reduce the risk of asthma in adults (Douwes et al., 2007). More studies are needed, however, to fully illuminate the finding.

The strengths of this study should be noted: most studies worldwide have been performed on hospital-based patients, which might be underestimated due to the fact that the severe patients tend to go to the hospital, while the less severe patients (accounting for the majority) tend to seek a self-treatment (Sreeramareddy et al., 2006). In this study, using the survey method of collecting all relevant symptoms helps enhance the power of statistical significances as we found the associations of nearly all symptoms with all pollutants. The advantage of hospital-based studies, however, is that they are more convincing. Nevertheless, although most of the symptoms in this study were not severe, some such as heart symptoms (tiredness and palpitation), swelling feet, etc., are early warnings of cardiovascular failure, which, if untreated, might become critical and require hospitalization. As such, air quality and health surveillance development should be focused on the methods to detect the affected persons as much as possible.

Some weaknesses in this study are as follows. First, as is always found in this type of study, the ambient pollution concentrations may not adequately reflect exposures of individuals (Agócs, White, Ursicz, Olson, & Vámos, 1997; Delfino, Zeiger, Seltzer, Street, & McLaren, 2002) resulting in the misclassification of actual exposure. Second, as stated above, this study failed to account for several potential confounding factors such as aeroallergens (Higgins et al., 2000), several air pollutants, e.g., volatile organic compounds (Delfino, Gong, Linn, Hu, & Pellizzari, 2003), medication uses (Delfino et al., 2002; Lewis et al., 2005), indoor air pollutants (such as biomass cooking), passive smoking, and participants' underlying diseases. Finally, because pollution levels also vary by day, the corresponding comparisons may be biased because the participants may also differ in important ways depending on what day of the week it is.

Conclusion

The findings of this study confirm the evidence that poor air quality can lead to discomfort. It also shows that some pollutants have synergistic effects to some certain symptoms. Finally, this study reveals that O_3 has a negative association with some symptoms that needs further work to validate the results.

Acknowledgements: Special thanks are due to the Pollution Control Department for the air quality data and to the Northern Meteorology Center for the meteorological data. The author would like to sincerely thank all residents participating in this study and research assistants for making this study possible. This study was funded by the Thailand Research Fund (RDG5130001).

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continued on page 84

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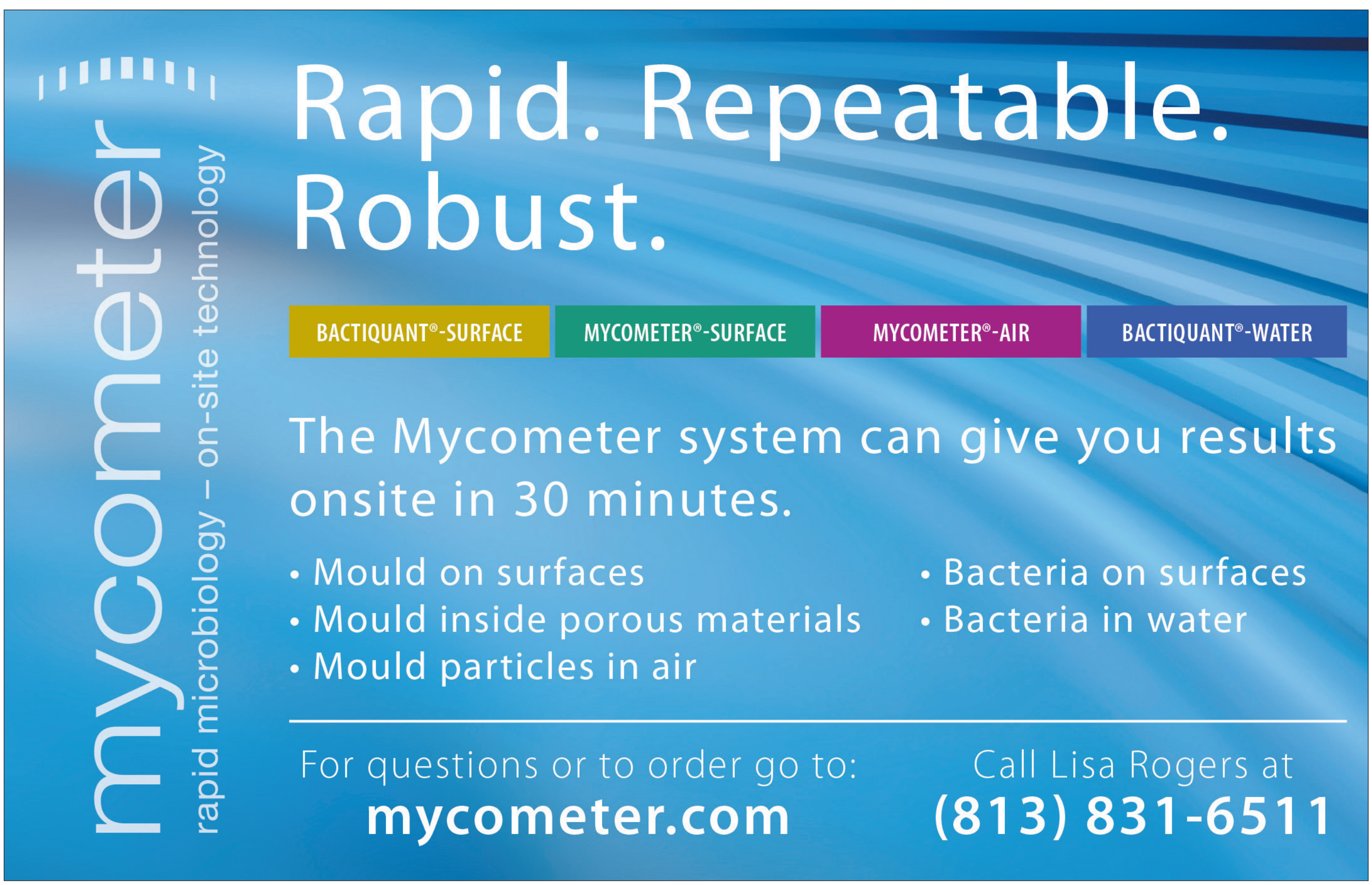
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Spatial Variation in Ambient Benzene Concentrations Over a City Park

Prepublished online May 2013,
National Environmental Health Association.

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Abstract **Passive diffusive samplers were used to collect ambient benzene for a one-week sampling period (April 27 to May 4, 2011) at 11 locations throughout a city park in the Tampa, Florida, area. Concentrations were determined through gas chromatography with mass spectrometry. Spatial variability within the park and its contribution to uncertainty in health risk estimates were studied. Measured concentrations ranged from 0.23 to 0.34 μg/m^3. The relative percentage differences for samplers collocated with a regulatory reference monitor and with a duplicate were 3% and 14%, respectively. The spatial variability over the park was small with a coefficient of variation of 11%. The concentration variation due to sampler placement contributes less to uncertainty in health risk estimates than the uncertainty associated with the inhalation unit risk parameter (39% versus 170% relative percent differences over the ranges studied). Results suggest that there is a limit to the spatial resolution needed for risk calculations.**

Introduction

Benzene is a known human carcinogen (e.g., it is classified as Group 1 by the International Agency for Research on Cancer), and a substantial contributor to estimated health risks of ambient exposures to toxic air pollutants (McCarthy, O'Brien, Charrier, & Hafner, 2009). Current regulatory monitoring networks for hazardous air pollutants, including benzene, use large and expensive systems that require electricity, restricting the location and number of measurement sites. This results in low spatial resolution of concentration data and inadequate characterization of environmental benzene exposures, particularly for children; hence, a high level of uncertainty is present in risk assessments (Möller, Schuetzle, & Autrup, 1994).

Due to continued lung development, higher breathing rates and activity levels, and more time spent outdoors, children are more susceptible to the health effects of air pollution than adults (American Academy of Pediatrics Committee on Environmental Health, 2004). High concentrations of combustion-related pollutants (including benzene) have been associated with higher incidences of acute respiratory infections in children (Romieu, Samet, Smith, & Bruce, 2002). Levels of a benzene metabolite in the urine of schoolchildren in Bangkok, Thailand, have been found to be comparable to that of adult street vendors who experienced higher ambient concentrations (Ruchirawat, Settachan, Navasumrit, Tuntawiroon, & Autrup, 2007). Toxicological support documents on benzene carcinogenicity for the U.S. Environmental Protection Agency (U.S. EPA, 1998) suggest that both the type of leukemia and susceptibility differ between children and adults; however, not enough data are available to quantify these differences. Higher resolution data would contribute to the understanding of both exposure and health effect differences.

To understand children's exposures, concentrations of benzene are needed where children spend time such as parks and school grounds. In a review, Mejía and co-authors (2011) found that most studies on air pollutant exposures at school used data from remote monitoring stations or dispersion modeling. Studies that monitored on school grounds often did not indicate the specific location of measurement. The pollutant focus of previous work has primarily been nitrogen dioxide, ozone, sulfur dioxide, and particulate matter. Work on spatial variation of benzene in urban areas has investigated impacts of roadway traffic and the relationship between indoor and outdoor concentrations. Thorsson and Eliasson (2006) and Menezes and co-authors (2009) included measurements of benzene in urban parks, in Sweden and Brazil, respectively; they found concentrations in parks to be substantially lower than levels in high traffic areas. Janssen and co-authors (2001) found decreasing outdoor benzene concentrations with distance from the motorway in a study of 24 schools in the Netherlands; indoor concentrations were higher than outdoor concentrations. Godoi and co-authors (2009) found measured benzene levels to be higher indoors than outdoors in a study of two schools in

Brazil. By examining the indoor-outdoor concentration ratio, they concluded that the indoor concentrations were primarily due to outdoor sources. Overall, more measurement data is needed on the spatial variation in benzene concentrations within areas where children may be exposed.

Studies of environmental equity investigate the distribution of environmental exposures, risks, and effects among subpopulations of different races or socioeconomic status (Brown, 1995). In studies of multiple urban areas, disadvantaged groups have been found to disproportionately live in neighborhoods with high traffic density and poor air quality (Houston, Wu, Ong, & Winer, 2004; Pastor, Sadd, & Morello-Frosch, 2002; Wheeler & Ben-Shlomo, 2005). In the Tampa, Florida, area, studies have found that census tracts and elementary schools with higher proportions of African-American, Hispanic, and low-income groups are located in areas with higher levels of nitrogen dioxide, air toxics, and traffic counts (Chakraborty, 2009; Stuart, Mudhasakul, & Sriwatanapongse, 2009; Stuart & Zeager, 2011). Using neighborhood-scale (block group) census data, Stuart and co-authors (2009) also found that these same groups disproportionately reside further from regulatory monitors.

In a study of southern California, Houston and co-authors (2004) concluded that the historic urban sprawl growth pattern resulted in minority and poor neighborhoods located in areas with high traffic density and more affluent neighborhoods located in suburban areas. Hence, it contributed to a disproportionate burden of health effects on poorer populations. Additionally, urban sprawl is considered a less sustainable form of growth than compact, dense cities, due to increased land use and inefficiency of public transportation (Camagni, Gibelli, & Rigamonti, 2002). As cities work to manage growth, high-resolution monitoring can guide approaches for maximizing exposure equality and urban sustainability.

This article presents a pilot study of small-scale variation in benzene concentrations over a city park and investigates impacts on the uncertainty of cancer risk estimates, particularly for children. We present the methods used for the sampling and analysis of benzene, method evaluation, and data analyses. Results are then discussed on ambient benzene levels, their spatial variation, and the uncertainty in cancer risk estimates introduced by sampler placement.

Methods

A city park in Hillsborough County, Florida, was chosen as the field site for our pilot study. The park is located adjacent to an elementary school and shares a structured playground with the school. We chose 11 sampling sites with available standing structures for sampler placement (a utility pole or tree), with the goal of spanning the park with approximately equidistant placement. For mapping and analysis, location coordinates were determined using a Garmin GPS device (with accuracy listed as within 10 m). Figure 1a shows the study area and sampling locations. For evaluation of measurement accuracy, we collocated an additional sampler with the only benzene regulatory monitor for the county, operated by the Environmental Protection Commission of Hillsborough County; this site is outside the park.

To characterize ambient benzene, we exposed passive diffusive samplers with activated charcoal sorbent for a one-week sampling period (April 27 to May 4, 2011) at the sites discussed above. A duplicate sampler was also exposed at site 8 for assessment of measurement precision. Two field blank cartridges accompanied the exposed samplers (one during deployment and the other during retrieval). Each field blank cartridge was uncapped and immediately resealed at one sampling location, as specified by U.S. EPA (1999a). Upon return to the laboratory, cartridges were stored at 4°C. Each cartridge was submersed for 30 minutes in 2 mL of low-benzene carbon disulfide for extraction. One laboratory blank cartridge was also used for quality control purposes. We quantified benzene via gas chromatography with mass spectrometry using a 50 m x 0.25 mm x 0.25 µm polysiloxane capillary column. A 1.0-µL aliquot of the extracted solution was injected on the column at a temperature of 240°C, with helium as the carrier gas at a flow rate of 1.2 mL/min. The total run time was 60 minutes with the following temperature program: start at 35°C for nine minutes, increase at 5°C/min. to 60°C, hold at 60°C for 46 minutes. To quantify unknown concentrations, we diluted analytical standard benzene stock to create five calibration standards ranging from 0.10 to 1.75 µg/mL. We used daily calibration curves, with a daily control chart for quality control. All samples and calibration standards were normalized by the addition of 2-fluorotoluene as an internal standard. The analysis protocol was adapted for our specific equipment from the Radiello sampler manual (Fondazione Salvatore Maugeri, 2006), previous studies (Angiuli et al., 2003; Cocheo, Boaretto, & Sacco, 1996; Godoi et al., 2009), and established standard methods (U.S. EPA, 1999a, 1999b). Quality control procedures and metrics were adapted from U.S. EPA Compendium Method TO-15 (U.S. EPA, 1999b).

We converted to concentrations in air by subtracting the mean blank value and multiplying the blank-adjusted values by a temperature-adjusted sampling rate and the exposure time recorded for each sampler. We used the sampling-period average of hourly temperatures measured at the Tampa International Airport (National Weather Service KTPA station) to adjust the sampling rate (Fondazione Salvatore Maugeri, 2006). The limit of detection was calculated as three times the standard deviation of the blank values, assuming a one-week sampling time (10,080 minutes).

To characterize spatial variations in benzene concentrations over the study area, we mapped the concentration distribution in ArcGIS, followed by spatial interpolation using kriging. Summary statistics, including the coefficient of variation (*CV*), were used to quantify variation.

To investigate the health risks associated with the concentrations measured here, we estimated excess lifetime cancer risk from inhalation exposure by multiplying the exposure concentration (C_E) by the inhalation unit risk for benzene. The exposure concentration was calculated as $C_E = C_a t_E f_E d_E / T$ (U.S. EPA, 2009), where C_a is the measured concentration of benzene, t_E is the exposure time (hours/day), f_E is the exposure frequency (days/year), d_E is the exposure duration (years), and T is the averaging time (hours). We chose exposure parameters to represent a child who is exposed due to time spent at school, with t_E of 6.5 hours/day, f_E of 180 days/year, and

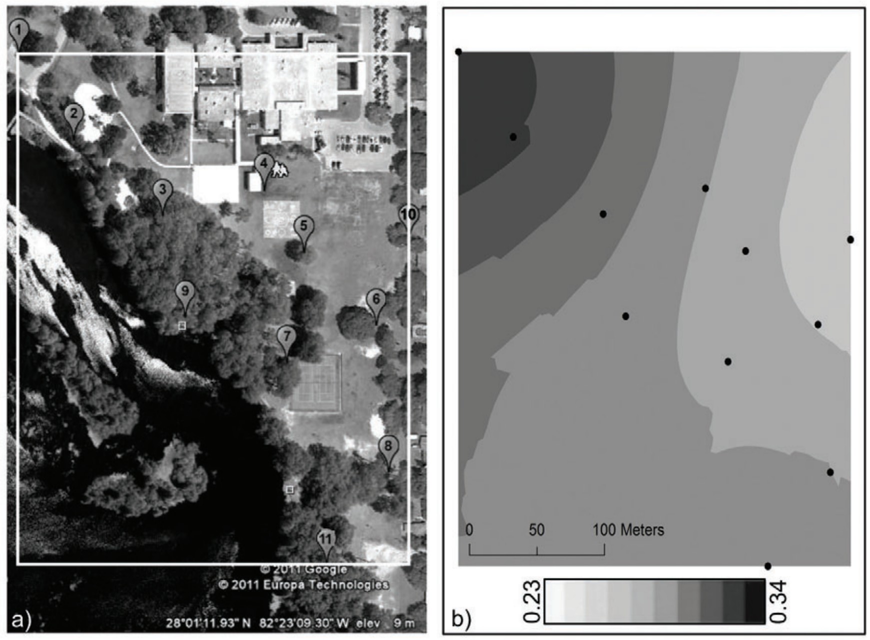
d_E of 6 years (representing kindergarten through 5th grade) (Hillsborough County Public Schools, 2010, 2011). For *T*, we applied the value for excess lifetime cancer risk of 70 years (in units of hours) (U.S. EPA, 2009). The inhalation unit risk represents the upper bound excess cancer risk per unit of exposure concentration determined through review of human and animal studies. U.S. EPA lists a range of $2.2\text{x}10^{-6}$ to $7.8\text{x}10^{-6}$ per µg/m^3 (U.S. EPA, 2010), while the California Office of Environmental Health Hazard Assessment (OEHHA) lists a value of $2.9\text{x}10^{-5}$ per µg/m^3 (OEHHA, 2009). For this work, we estimated risks using the minimum and maximum of these values, i.e., $2.2\text{x}10^{-6}$ and $2.9\text{x}10^{-5}$. These were combined with the range of measured concentrations to estimate a range of risks.

We also investigated the impact of sampler placement (and sampling resolution) on the uncertainty of health risks calculated using the standard methods discussed above. Since actual levels vary over the area represented by a single sample, the choice of sampler location leads to uncertainty in the exposure concentration. When propagated through the risk calculation, proportional uncertainty in the estimated health risk results. Uncertainty in the inhalation unit risk has a similar effect. Hence, we compared the range of measured concentrations (quantified as a relative percent difference) over the study area to that of the inhalation unit risk, in order to assess impacts of sampler placement on health risk assessment.

FIGURE **1**

a) Study Area With Sampling Locations (and Corresponding Sample IDs)

b) Contour Plot of Interpolated Concentrations (mg/m^3) (Generated in ArcGIS)



The white box in a) corresponds to the area plotted in b). Sample ID 12 (not shown) was located at the site of the county reference monitor, outside of the study area. A duplicate sampler was located at site 8.

Results and Discussion

Measured benzene concentrations and summary statistics for the study area are shown in Table 1. Results are at the low end of the range in weekly outdoor concentrations (0.3–5.0 µg/m^3) observed at schools in the Netherlands (Janssen et al., 2001) and weekly levels (0.3–1.4 µg/m^3) for an urban park in Gothenburg, Sweden (Thorsson & Eliasson, 2006). This is consistent with previous work showing low concentrations in urban parks as compared with levels in high traffic areas (Upmanis, Eliasson, & Andersson-Skold, 2001). Overall, the values measured here are near the low end of the general range of benzene measured in U.S. studies, of approximately 1–10 µg/m^3, with peak values up to about 50 µg/m^3 (Health Effects Institute Air Toxics Review Panel, 2007).

The overall range of estimated excess cancer risk for the range of concentrations measured here is from $5.7\text{x}10^{-9}$ to $1.1\text{x}10^{-7}$. These risks are quite low when compared with other studies (McCarthy et al., 2009; Payne-Sturges, Burke, Breysse, Diener-West, & Buckley, 2004; Tam & Neumann, 2004), which reported cancer risk due to benzene of greater than 10^{-6} at almost all sites. Here, we considered only the contribution to the lifetime cancer risk due to elementary school exposure, not an exposure duration of an entire lifetime. Further, we note that the concentrations here are for one week of sampling and hence may not be representative of longer-term averages or extremes.

Evaluation results indicate a precision from the two collocated duplicate samplers of 14% (as a relative percent difference), which is within the recommended guideline of less than 25% (U.S. EPA, 1999b). A 3% relative percent difference was found between the concentration measured by the reference regulatory monitor and that measured by a collocated passive sampler; this is within the precision of the passive measurement (that ultimately limits the measurable accuracy). We note that the comparison is not direct, since the sampling times are different. The reference monitor takes 24 hour canister samples every six days; the final 13 hours of one sampling period and the full second sampling period of the reference monitor overlapped with the sampling period for our study.

The spatially interpolated concentration field is provided in Figure 1b. A gradient can be observed; the highest interpolated concentrations are in the northwest corner of the sampling area, which includes the site near the playground structure (site 2). The highest point measurement value was at site 3. The

TABLE 1

Benzene Concentrations and Summary Statistics

Sample ID (or Summary Statistic)	Concentration (µg/m³)
1	0.33
2	0.33
3	0.34
4	0.28
5	0.31
6	0.27
7	0.29
8	0.29
8* (duplicate)	0.34
9	0.29
10	0.23
11	0.31
12	0.26
Reference monitor	0.26
Mean	0.30
Standard deviation	0.03
Range	0.23–0.34

Note. The limit of detection was 0.18 mg/m³. Sample ID locations are provided in Figure 1. Summary statistics are based on measurements from sites in the study area (1–11), excluding the site 8 duplicate value. The reference monitor value is the average of two 24-hr. sample values (from April 27 and May 3) obtained from the Environmental Protection Commission of Hillsborough County from every sixth day monitoring during the study period. Reference-method samples were collected in 6-L stainless steel canisters and analyzed by gas chromatography-mass spectrometry, per *Compendium Method TO-15* (U.S. Environmental Protection Agency, 1999b). All calculations were performed using full precision in Excel; hence summary values cannot necessarily be reproduced from the concentration values reported to two significant digits.

location of the high area is consistent with emissions expected from the parking lot and vehicular park entrance nearby. Wind data for the Tampa area (at the international airport) do not indicate a dominant wind direction during the study period. Only 12% of the winds were from the WNW to N quadrant, however, which may have kept concentrations low overall. It should be noted that, due to the precision of measurement, confidence in real differences is small. Overall, the coefficient of variation for the sampling area was only 11%. Although no standard metric exists for quantifying spatial heterogeneity, a value of 20% has been used to distinguish between a homogeneous and heterogeneous field in previous studies of particulate matter (Blanchard et al., 1999; Wilson, Kingham, Pearce, & Sturman, 2005). The appropriateness of this criteria for a benzene field is unknown, but is explored below.

The range of measured concentration values can be used to explore impacts of sampler placement on uncertainties in the estimated health risks of benzene. The relative percent difference between the maximum and minimum measured concentrations was 39%; this results in an equivalent percent difference in estimated excess health risks (if all other parameters are kept constant). As a comparison, a similar value (44%) can be calculated from the range of risks due to toxic air pollutant levels (driven primarily by formaldehyde and benzene) measured at four sites in a study of intra-urban differences in Pittsburgh (Logue, Small, Stern, Maranche, & Robinson, 2010). Conversely, the range of values for inhalation unit risk listed by U.S. EPA and California OEHHA is much larger, at 170% (as relative percent difference). (Note that this does not represent the full range of risks, just a range of upper bound risk estimates.) Hence, the uncertainties associated with sampler placement in our study are small compared with those from the unit risk parameter; the variation in concentration over the sampling area would need to be much larger to have a similar independent effect. This suggests a threshold to the utility of increased spatial resolution of measurement that will depend on the purpose for which the concentration measurements are used. We note, however, that the effect on overall risk uncertainty is multiplicative, so even small variations in concentrations could lead to large overall uncertainties in risk. Further, uncertainties in other parameters in the calculation are not analyzed here (e.g., personal activity patterns).

Although of limited utility for risk estimation at the very high resolution studied here, monitoring data that characterizes intra-urban variations may be helpful for epidemiological studies, development of better risk assessment values, and city planning. While considering the effects of misclassification of particulate matter exposure on inference from time-series mortality studies, Zeger and co-authors (2000) found that the largest bias was due to differences between the ambient levels measured by fixed-site monitors and average individual exposures. Spatially resolved outdoor benzene concentrations may help reduce this bias, particularly given the association between indoor and outdoor concentrations of benzene (Godoi et al., 2009). In combination with personal activity data, higher-resolution concentration data may allow better estimation of individual exposures (Nuckols, Ward, & Jarup, 2004; Whitworth, Symanski, Lai, & Coker, 2011).

The usefulness of additional increases in spatial resolution may currently be limited, however, by other uncertainties in risk estimation and the precision of passive measurements. Nonetheless, decreased misclassification of exposure in health effect studies may in turn lower the uncertainty in health risk assessment parameters. This is particularly important for children, as current U.S. EPA risk assessment methods do not differentiate risk to children for benzene carcinogenicity. Jensen and co-authors (2001) suggest that high-resolution concentration data may aid city planners to build sustainable cities and reduce environmental inequity. For these purposes, characterization of concentration variations over space may be as important as differences in individual activities and exposures.

Conclusion

Eleven sampling locations were chosen at a city park to investigate spatial variation in ambient benzene using passive samplers. Precision and accuracy were evaluated through co-location. Excess lifetime cancer risks were estimated, and the magnitude of uncertainty due to sampler placement was compared to the uncertainty due to inhalation unit risk. The precision and accuracy were measured at 14% and 3%, respectively. The spatial variation over the park was found to be low (*CV* of 11%), with an overall range, as a relative percent difference, of 39%. Comparison suggests limits to the use of these methods for very high resolution sampling, but appropriateness for study of larger-scale intra-urban concentration variations of benzene in Hillsborough County.

Compared with the range of inhalation unit risk of 170% (as a relative percent difference), these results indicate that one sampler may be sufficient to represent the excess lifetime cancer risk from benzene exposure for the sampling area, using currently recommended methods. Further research is needed to characterize the spatial variation in benzene concentrations and risks over larger intra-urban scales, including neighborhoods. The necessary monitoring resolution will depend on the purpose for which the data will be applied. High-resolution data may lower errors in epidemiological studies resulting from differences between actual and measured concentrations, may improve risk assessment parameters (particularly for less-characterized groups, like children), and may aid city design.

Acknowledgements: The authors thank Drs. Foday Jaward and Thomas Mason for helpful comments on the thesis that generated this manuscript. Dr. Foday Jaward also provided assistance with the GCMS system and Haofei Yu provided assistance with sampler deployment and mapping. This material is based upon work supported by the National Science Foundation under Grant No. 0846342.

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Transport of *E. coli* in a Sandy Soil as Impacted by Depth to Water Table

Prepublished online June 2013,
National Environmental Health Association.

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Abstract **Septic systems are considered a source of groundwater contamination. In the study described in this article, the fate of microbes applied to a sandy loam soil from North Carolina coastal plain as impacted by water table depth was studied. Soil materials were packed to a depth of 65 cm in 17 columns (15-cm diameter), and a water table was established at 30, 45, and 60 cm depths using five replications. Each day, 200 mL of an artificial septic tank effluent inoculated with *E. coli* were applied to the top of each column, a 100-mL sample was collected at the water table level and analyzed for *E. coli*, and 100 mL was drained from the bottom to maintain the water table. Two columns were used as control and received 200 mL/day of sterilized effluent. Neither 30 nor 45 cm of unsaturated soil was adequate to attenuate bacterial contamination, while 60 cm of separation appeared to be sufficient. Little bacterial contamination moved with the water table when it was lowered from 30 to 60 cm.**

Introduction

Approximately 20% of the U.S. and 50% of the North Carolina households use septic systems for managing their sewage (U.S. Census Bureau, 2004, 2011). In order to properly treat wastewater containing an array of disease-causing organisms (e.g., *Salmonella* spp., *E. coli*) (Meschke & Sobsey, 1998), the soil in the treatment area of any septic system (known as drainfield) must remain aerobic and unsaturated (Paul, Rose, Jiang, Kellogg, & Shinn, 1995). The relatively nutrient-poor unsaturated soil is an undesirable environment for anaerobic bacteria in nutrient-rich septic tank effluent. Also, aerobic soil contains a large number of indigenous bacteria, protozoa, and nematodes that can prey on bacteria present in the wastewater.

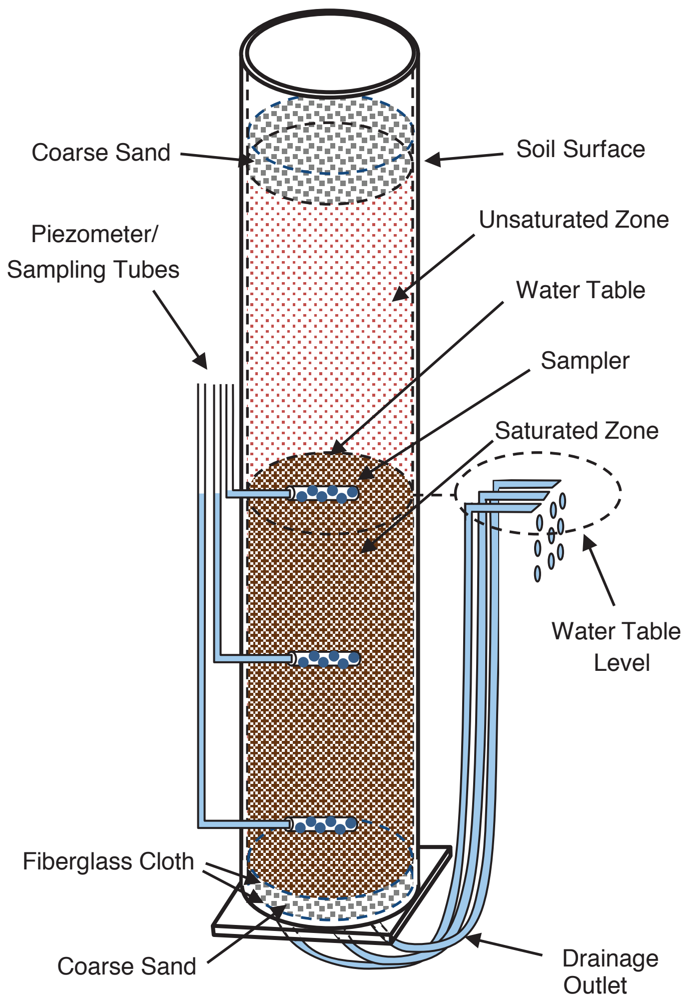
According to North Carolina regulations (North Carolina Department of Environment and Natural Resources [NCDENR], 2007), sites for individual septic systems must contain an adequate area with a minimum of 45 cm (18 in.) of suitable unsaturated soil between the bottom of the system's trenches and any unsuitable layer (based on soil structure, mineralogy, wetness, and the presence of a restrictive layer or bedrock) in sandy soils, and a minimum of 30 cm (12 in.) of suitable unsaturated soil in coarse loamy, fine loamy, and clayey soil groups. The regulations regarding this separation distance vary among states (for example, Delaware Department of Natural Resources and Environmental Control, 2005; Idaho Department of Environmental Quality, 2011; Indiana State Department of Health, 1990).

The seasonal high water table (SHWT), identified using morphological properties with a chroma 2 or less (i.e., redoximorphic features) (Vepraskas, 1992), or by direct monitoring, is considered a restrictive layer. Severson and co-authors (2008), however, showed that the SHWT may rise above the redoximorphic features of chroma 2 or less at certain times during the year. The capillary fringe (CF), the zone immediately above the water table, is nearly saturated even though it is considered to be part of the vadose (unsaturated) zone. Misinterpretation of the redoximorphic features or not recognizing the presence of CF may result in an overestimation of the actual thickness of the unsaturated soil needed for the removal of microbes in wastewater.

The fate and transport of *E. coli* and other bacteria through soil have been studied for decades (Ausland, Stevik, Hanssen, Kohler, & Jenssen, 2002; Bolster & Abit, 2012; Butler, Orlob, & McGauhey, 1954; Faust, 1982; Gagliardi & Karns, 2000; Hagedorn, McCoy, & Rahe, 1981; Jiang, Morgan, & Doyle, 2002; Karathanasis, Mueller, Boone, & Thompson, 2006; Rahe,

FIGURE 1

Schematic Diagram of the Soil Columns



Hagedorn, McCoy, & Kling, 1978; Smith, Thomas, White, & Ritonga, 1985; Stiles & Crohurst, 1923; Tate, 1978). Despite these studies, evidence regarding the critical distance for treatment and removal of enteric bacteria in soils between the bottom of the septic system trenches and the SHWT remains inconclusive. Caldwell (1937) and Caldwell and Parr (1937) recommended 150 to 200 cm of unsaturated soil for effectively removing all enteric bacteria from sewage. Cogger and co-authors (1988) found that 30 cm of unsaturated coastal plain soils of North Carolina was not adequate for removing bacteria, but 60 cm may be the critical threshold for bacterial treatment.

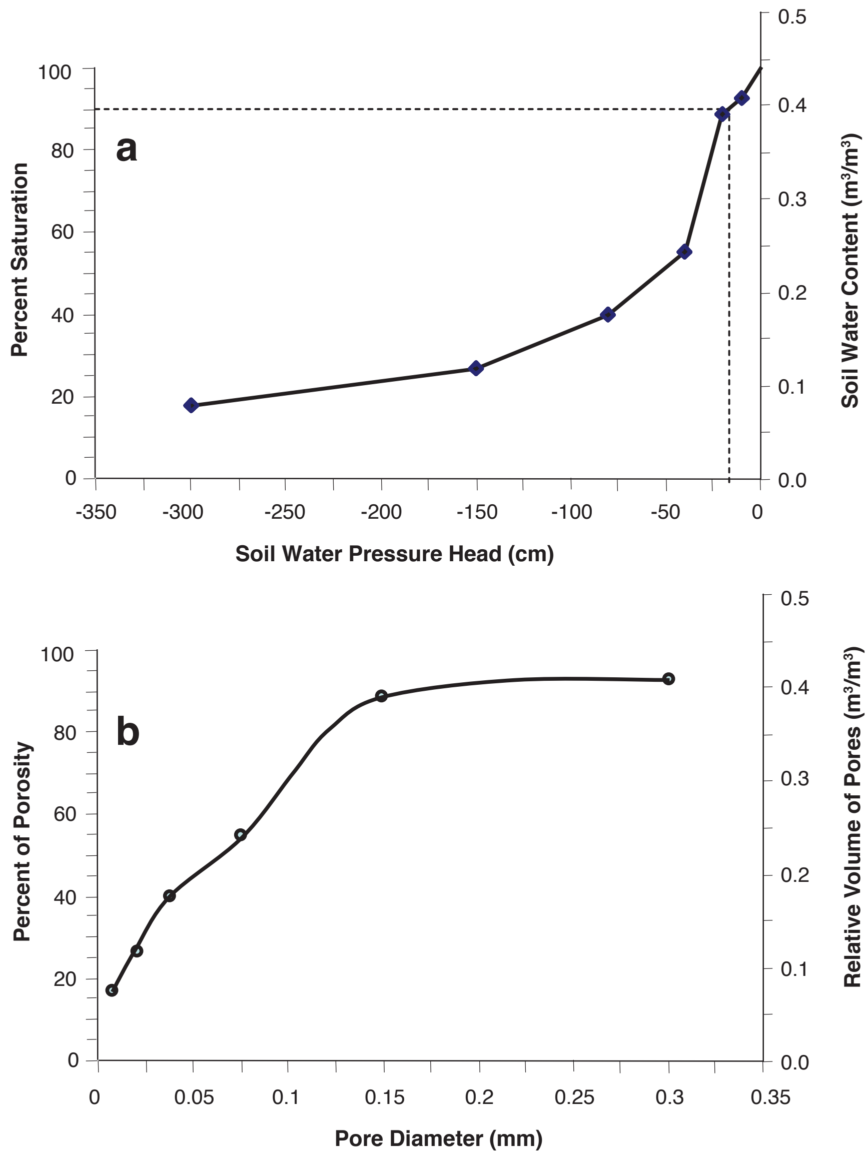
Mathematical models have been used to describe transport of bacteria through soil (Abu-Ashour, Joy, Lee, Whiteley, & Zelin, 1994; Hijnen, Brouwer-Hanzens, Charles, & Medema, 2005; Hornberger, Mills, & Herman, 1992; McGechan & Vinten, 2003). In recent years, the HYDRUS model (Simunek, van Genuchten, & Sejna, 2006) has been used to simulate water flow in the drainfield of septic systems (Finch, Radcliffe, & West, 2008; Heatwole & McCray, 2007; Radcliffe & West, 2009). These models require specific information about various soil properties that may be difficult to determine. To our knowledge, none of the mathematical models for septic systems has been tested under field conditions. Therefore, their use for designing septic systems is not recommended at this time.

One of the problems for selecting an appropriate separation distance is related to water flow from the septic system trenches. Water flow in coarse-textured soils is through interparticle pores, while in well-structured loamy and clayey soils water flow is mainly through macropores (root channels, animal borings, and interped faces) (Amoozegar, Niewoehner, & Lindbo, 2008; Vepraskas, Jongmans, Hoover, & Bouma, 1991). Also, in areas with shallow groundwater CF may play an important role in vertical and horizontal water flow (Abit, Amoozegar, Vepraskas, & Niewoehner, 2008a, 2008b; Amoozegar, Niewoehner, & Lindbo, 2006). In general, water flow under unsaturated conditions is mainly through small pores, which increases the contact with soil particles, allowing the chemical and microbial contaminant to be attenuated more effectively through adsorption or physical filtration. Conversely, under saturated conditions more water moves through the larger pores than under unsaturated flow conditions. As a result, pathogens can be transported rapidly with little attenuation or filtration below the water table (McCoy & Hagedorn, 1979). Studies have also shown that bacteria are able to survive and move long distances in a highly conductive saturated zone (Anan'ev & Demin, 1971; Butler et al., 1954; McCoy & Hagedorn, 1979; Stiles & Crohurst, 1923). In addition, saturated conditions may kill the indigenous bacteria, nematodes, and protozoa that eliminate the enteric bacteria from the septic tank effluent.

The coastal plain region of North Carolina and other eastern states contains many coarse-textured soils with SHWT. It is advantageous to know what separation distance is adequate to eliminate pathogens from septic tank effluent and if microbial contamination can travel vertically when a water table falls with drainage. The goal of our study was to assess the fate and transport of microbes present in septic tank effluent applied to a coastal plain soil with short transport distances to a SHWT. Specifically, the objectives were as follows: 1) to evaluate the depth to water table for removal of bacteria, and 2) to determine if microbial contamination moves with the fluctuating water table.

FIGURE **2**

Average Soil Water Characteristic Curve (a) and Pore Size Distribution (b) for Repacked Norfolk Sandy Loam Soil Material



The dashed line shows the water content and soil water pressure head at 90% saturation.

Materials and Methods

Soil Columns

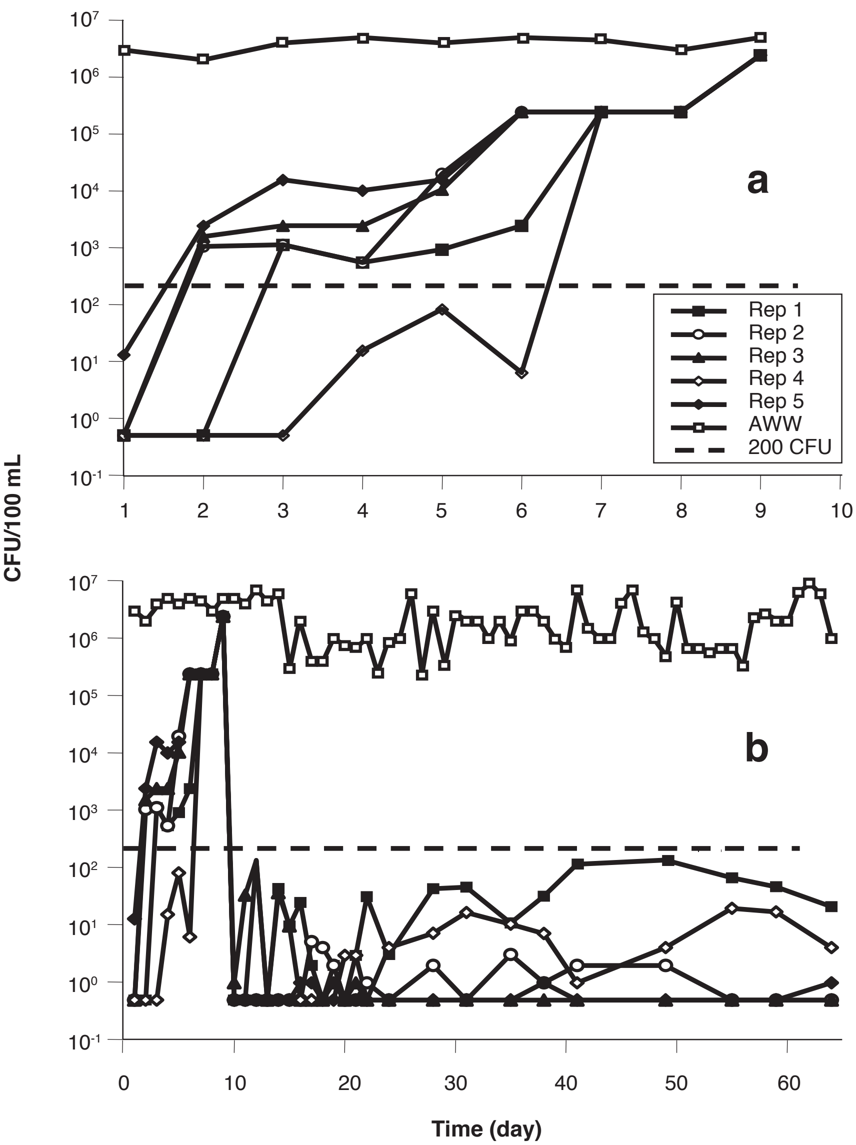
The soil was collected from a Norfolk loamy sand (thermic Typic Kandiudults) site in the lower coastal plain region of North Carolina. The Norfolk soil series is commonly found in the southeast as well as other states (Virginia, North Carolina, South Carolina, Georgia, Florida, Alabama, and Arkansas) (see https://soilseries.sc.egov.usda.gov/OSD_Docs/N/NORFOLK.html). The soil material was air dried and passed through a 2-mm (#10) sieve. Seventeen columns (15.2-cm inside diameter), constructed of 75-cm long sections of 6-inch polyvinyl chloride (PVC) pipe, were packed with 18.6 kg of soil (in 4.5-cm thick sections) to a depth of 65 cm at an average bulk density of 1.57 g/cm^3 over a 2-cm thick layer of coarse sand (Figure 1). The inside walls of the columns were covered with a sand-silicone caulk mixture to prevent wall flow. Three outlets at the bottom connected to Tygon tubing regulated the water table level in each column. A 2.5-cm thick layer of coarse sand was placed on top of the soil to prevent surface scouring during wastewater application. Three small water samplers, constructed of 7.5-cm long perforated Plexiglass tube and wrapped in cheesecloth, were installed at 30, 45, and 60 cm below the soil surface in each column (Figure 1). The top of the soil below the sand layer represented the bottom of the septic system trench.

The soil texture (sandy loam with 64% sand, 30% silt, 6% clay, and 0.3% organic matter) was determined by the hydrometer method (Gee & Or, 2002), and the soil water retention was determined (Dane & Hopmans, 2002) using eight 7.5-cm diameter, 7.5-cm long cores, packed at an average bulk density of 1.49 g/cm^3. The pore size distribution was calculated using the average water content at different pressure heads (Figure 2). Approximately 45% of the pores in the columns were smaller than 0.06 mm (60 μm) and 90% of the pores were smaller than 0.15 mm (150 μm).

The columns were initially saturated from the bottom to prevent air entrapment in the soil. The columns were then drained and the water levels in them were regulated at 30, 45, and 60 cm below the soil surface by adjusting the Tygon tubing outlets (Figure 1). Five replications were used for each water table treatment and two columns were used as control with water table at 30 cm depth. Once the water table was established at the desired depth, the drainage outlets were fixed, and the columns were left stagnant for three weeks to form anaerobic conditions.

FIGURE **3**

Concentration of *E. coli* Bacteria Detected at 30 cm Depth in Soil Columns



After passing artificial wastewater (AWW) through 30 cm of unsaturated soil for nine days (a), and at 60 cm depth for 55 days after lowering the water table from 30 cm to 60 cm (b).

Wastewater Characteristics and Application

E. coli isolated from human urine (American Type Culture Collection [ATCC] #11775) was used as a proxy for *E. coli* in actual septic tank effluent because its characteristics are considered to be similar to *E. coli* found in human waste, and it has been used as a control agent for analyzing *E. coli* in wastewater (Evenson & Strevett, 2006; U.S. EPA, 2010). In addition, the ATCC #11775 has a biosafety level of 1, indicating the elevated risk of illness only for individuals with a compromised immune system. Detailed information on this strain can be found at http://www.atcc.org/ATCCAdvancedCatalogSearch/ProductDetails/tabid/452/Default.aspx?ATCCNum=11775&Template=bacteria. Artificial wastewater (AWW), consisting of 8.5 mg/L monopotassium phosphate, 21.75 mg/L dipostassium phosphate, 17.7 mg/L disodium phosphate, 27.5 mg/L calcium chloride, 11 mg/L magnesium sulfate, 15 mg/L sodium chloride, and 60 mg/L nutrient broth (Powelson & Mills, 2001) was used because it is more homogeneous, predictable, and safer to handle than actual septic tank effluent. This AWW, spiked with ATCC #11775, closely mimicked the survival of microbes in actual septic tank effluent.

The long-term application rate (LTAR) for sandy loam textured soils in North Carolina (NCDENR, 2007) is 0.3-0.4 gal/(ft^2/day) for low-pressure pipe systems (based on drainfield area) and 0.6–0.8 gal/(ft^2/day) for conventional systems (based on trench bottom area), which is equivalent to 0.2 to 0.27 gal/(ft^2/day) based on drainfield area. In our study, 200 mL of wastewater were applied to each column daily. Based on the cross-sectional area of the columns (182 cm^2), this is equivalent to an application rate of approximately 1.1 cm/day or 0.27 gal/(ft^2/day), which is in line with the LTAR used in North Carolina and perhaps many other states.

Experimental Procedure

Seventeen 200-mL aliquots of AWW were autoclaved, and 15 of them were inoculated with three to four colonies of *E. coli* each day. The inoculated aliquots were then incubated for 24 hours at 37°C. Each 200-mL dose of AWW had between 9.4×10^4 and 9.7×10^6 CFU of bacteria/100 mL, which is typical of actual septic tank effluent (Ausland et al., 2002; Prasad, Rajput, & Chopra, 2006). The spread plate technique was used to enumerate the *E. coli* concentration in 100-µL aliquots from AWW serial dilutions (10^{-4} to 10^{-6}) on mFC agar plates. The inoculated plates were incubated in a water bath at 44.5°C for 24 hours before the dark blue colonies were counted. These plates also served as the source of *E. coli* for subsequent inoculation of AWW.

Once a day for 65 days, 200 mL of inoculated AWW were applied to the top of each of the 15 test columns, and 200 mL of sterilized AWW were applied to each of the two control columns. The sampling port at the designated water table level on the side of the respective column was then opened to allow a 100-mL soil solution sample from top of the water table to drain freely into a dark colored (light-blocking) sample bottle. To maintain a constant depth to water table, another 100-mL sample was removed from an outlet at the bottom of the column and discarded. All the applied wastewater infiltrated the soil after application, giving an average rate of 1.1 cm/day for water flow through the unsaturated zone. The soil water profile above the water table in each column fluctuated between consecutive wastewater applications, but by maintaining a water table at a given depth the average rate of vertical flow and water content fluctuations remained relatively uniform during our study.

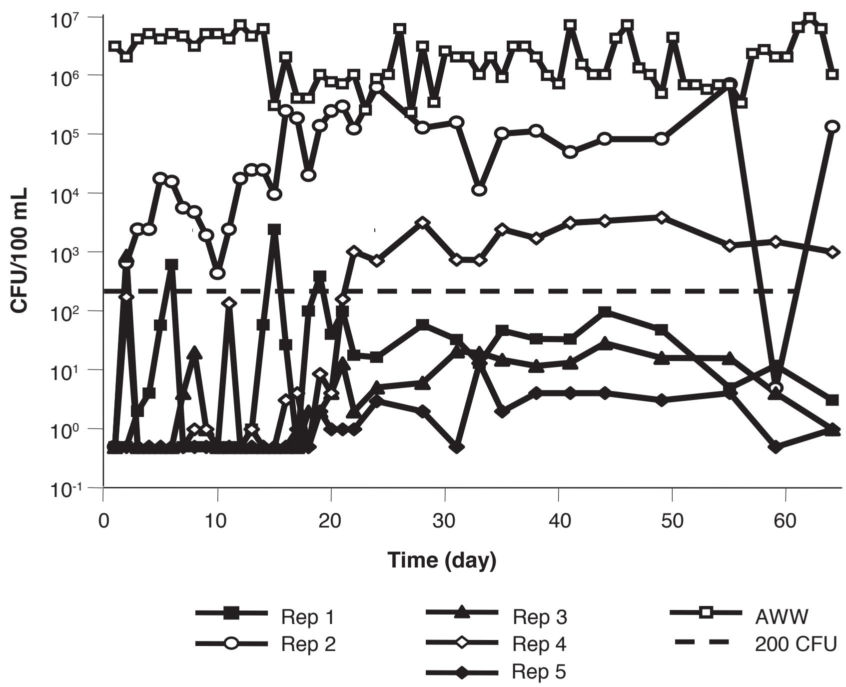
The 100-mL samples collected at the water table were enumerated for *E. coli* within 30 minutes of collection by the Colilert procedure (U.S. EPA, 1997 [U.S. EPA method# 9223B]). This method correlates well with both the membrane filter and multiple tube fermentation procedures (Eckner, 1998; Edberg, Allen, & Smith 1988, 1991; Edberg, Allen, Smith, & Kriz, 1990). According to IDEXX Laboratories, the upper detection limit for this procedure is approximately 2,420 CFU/100 mL. Any sample reaching this limit was diluted using sterilized deionized water and reanalyzed. For quality control, all sets of analysis included blank and replicate samples.

The safe drinking water standard for total coliform is zero (U.S. EPA, 2011). Since North Carolina has no septic system regulations regarding microbial concentration in groundwater, the primary contact level of 200 CFU/100 mL for class C waters (water suitable for fishing, wildlife, secondary recreation, and agriculture) was selected arbitrarily for comparison.

After nine days, the *E. coli* concentrations at the water table for all five replications for the 30-cm separation treatment were equal to the inflow concentration, indicating a complete breakthrough. The water table for this treatment was then dropped to 60 cm depth by lowering the outflow tubes 5 cm every hour. This treatment simulated a falling water table, although the rate of the water table drop was faster than what occurs under natural conditions. The second part of the experiment was continued to day 65 using the same procedure as previously described.

FIGURE **4**

Concentration of *E. coli* Bacteria Detected at 45 cm Depth in Soil Columns



After passing artificial wastewater (AWW) through 45 cm of unsaturated soil for 65 days.

Statistical analysis was performed with a nonparametric one-way Kruskal-Wallis Test (Hollander & Wolfe, 1973). This test makes no assumptions about the normality of the data. The overall comparisons were made with the NPAR1WAY procedure of SAS (SAS Institute, 1989, 1994). A follow-up procedure was used to make pairwise comparisons between two treatments at a time. The procedure that was used was a modified version of a Dunn (1964) procedure presented in Hollander and Wolfe (1973) due to the limits of the Kruskal-Wallis procedure dealing with large sample sizes.

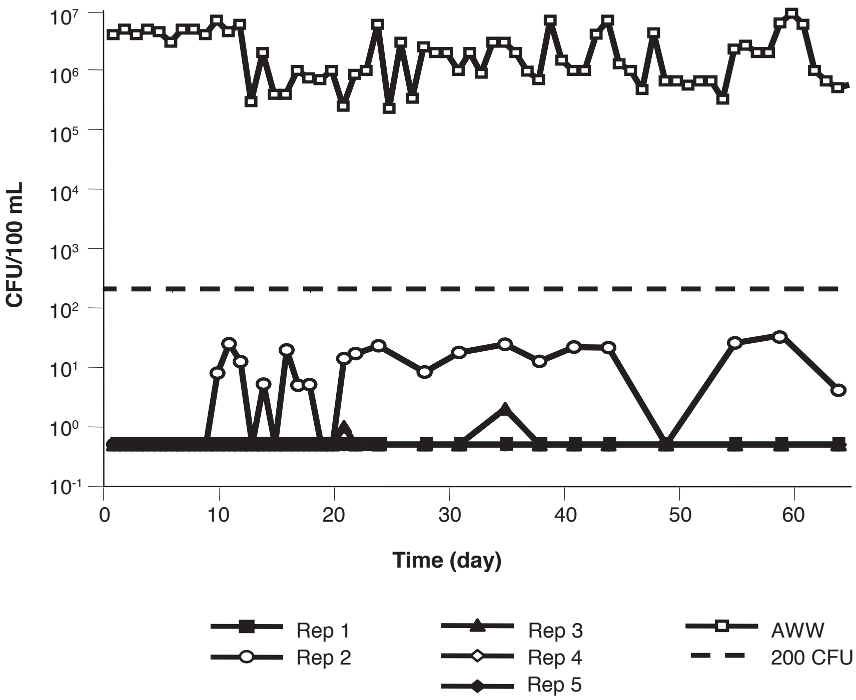
Results and Discussion

Microbial Attenuation

Three days after the wastewater application, the *E. coli* counts at the water table in four of the replications for the 30-cm water table treatment were above the 200 CFU/100 mL (Figure 3a). Within nine days, microbial counts at the water table for all five replications (2.2×10^6 CFUs/100 mL) were statistically on the same order of magnitude as the AWW inflow concentration of 3.9×10^6 CFU/100 mL (Figure 3a). With virtually no decrease in concentration of *E. coli* within 30 cm of unsaturated zone, this was deemed a treatment failure. The volume of water held against free drainage in the upper 30 cm of the column calculated using the average soil water characteristic curve (see Figure 2) was approximately 2,100 cm^3. Based on this, the cumulative volume of water applied to the columns in nine days was equivalent to 0.85 pore volume, indicating that microbes were transported to the water table with little to no retention. Also, approximately 90% of the pores of this soil were filled with water at 17 cm of tension, indicating

FIGURE 5

Concentration of *E. coli* Bacteria Detected at 60 cm Depth in Soil Columns



After passing artificial wastewater (AWW) through 60 cm of unsaturated soil for 65 days.

that the CF in this soil is approximately 17 cm thick. Since CF is almost saturated, little microbial attenuation may take place, leading to rapid contamination of the top of the saturated zone. Overall, we can say with certainty that 30 cm of unsaturated flow above a water table is insufficient for microbial treatment.

A high degree of variability occurred overall with *E. coli* concentrations ranging from 10^1 to 10^5 on day 65 for the 45 cm water table treatment (Figure 4). Even though *E. coli* concentrations for three replications did not exceed the standard, the geometric mean of *E. coli* concentration was close to or exceeded the 200 CFU/100 mL during 65 days of wastewater application. Therefore, 45 cm of unsaturated flow in this soil may not be sufficient for adequate treatment of bacterial contamination. We believe this may be due to the presence of the CF (assumed to be 17 cm thick). Considering the CF, only 28 cm of unsaturated flow was present for this treatment.

Sixty centimeters of separation may be the critical threshold for removal of enteric bacterial contamination during 65 days of wastewater application because the *E. coli* concentration at the water table never reached the 200 CFU/100 mL limit selected as standard (Figure 5). Three of the five replicates never showed *E. coli* concentration, one had a maximum of 2 CFU/100 mL, and the last one had a maximum concentration of 34 CFU/100 mL. Excluding the CF, 60 cm of separation is equivalent to approximately 43 cm of unsaturated flow in this soil.

The pairwise comparison confirmed that, at a probability level of .05, the CFU for both 30- and 45-cm water table treatments were greater than 60 cm of separation. No statistically significant difference existed, however, between 30- and 45-cm treatments, even though the numerical average concentration of *E. coli* for 30-cm treatment was substantially greater than the 45-cm separation distance. At the .075 probability level, all treatment comparisons were statistically significant.

Water Table Fluctuation

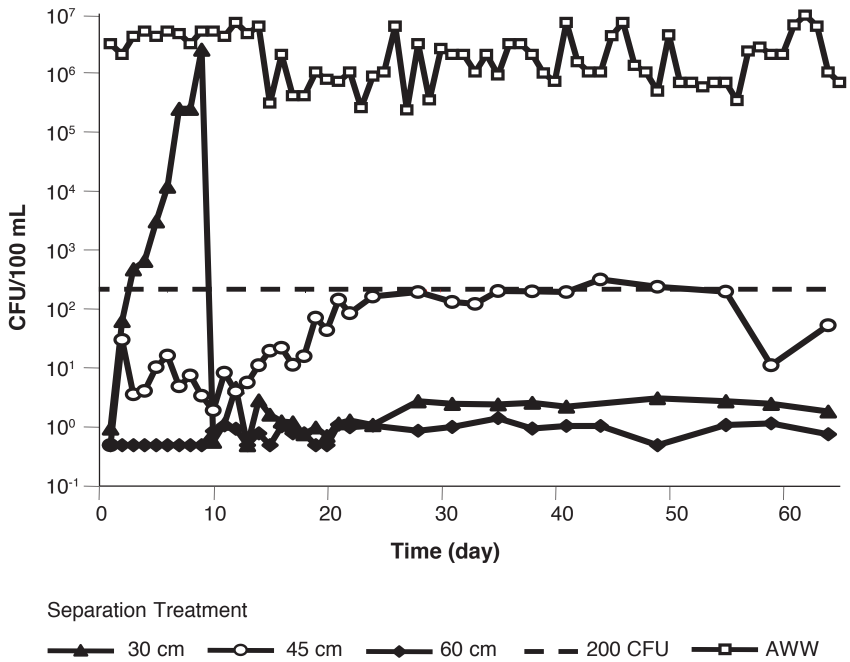
As described earlier, the water table at 30 cm depth was lowered to 60 cm to ascertain if microbial contamination survives and travels with the falling water table. As the water table receded, concentrations of *E. coli* at the water table decreased substantially, indicating that a limited number of live microbes move downward with the falling water table (Figure 3b). After lowering the water table to 60 cm, the average concentration of bacteria on day 10 was <1 CFU/100 mL for the five replications (Figure 6). The concentration of *E. coli* after day 10 increased for some of the replications (see Figure 3), but was never more than 138 CFU/100 mL. In comparison, the maximum *E. coli* concentration for the original 60 cm treatment remained below 34 CFU/100 mL as described before. Higher *E. coli* concentration at 60 cm depth for the water table dropping treatment was perhaps due to the live transport of some of the bacteria when the water table receded. Although the bacterial contamination did not move significantly with the falling water table, they may have survived long enough to be carried by the addition of wastewater on subsequent days. This finding concurs with Stiles and Crohurst (1923) that bacterial contamination in the upper part of an aquifer is stranded in the unsaturated and CF when the water table recedes.

The overall Kruskal-Wallis nonparametric statistical test and the pairwise comparison test indicated that a significant difference existed among water table treatments. Treatment that began with 30 cm of separation and was increased to 60 cm of separation (deeper water table) had higher CFU values than the treatments where water table remained at 60 cm of separation. Treatments with 45 cm of separation had greater values of CFU than both of these treatments after day 10.

The mechanisms for bacteria attenuation in soil include physical filtration, adsorption to soil particles, and die-off (Gannon, Manilal, & Alexander, 1991). In general, large-diameter pores play an important role in water flow under saturated or near-saturated conditions (as in the CF), whereas water flow in the unsaturated zone is restricted to the smaller pores. As a result, in soils containing macropores (e.g., worm hole, root channels), or large interparticle pores (e.g., in sandy soils), filtration of colloidal particles (Kretzschmar, Robarge, & Amoozegar, 1994) as well as bacteria may be limited. In addition, under saturated or near-saturated conditions, little oxygen may be present in the soil, allowing anaerobic bacteria to survive. Based on data in Figure 2, the average water content in the unsaturated zone above the water table was relatively high and only 15% of the pores were filled with air when the water table was at 30 cm depth. This allowed rapid transport of *E. coli*, providing an undesirable environment for bacterial survival. When the water table was at 60 cm depth, almost 50% of the pores were filled with water, and only pores that were smaller than 75 µm conducted water in the upper 20 cm of the column. This allowed more filtration and die-off for bacteria.

FIGURE **6**

Geometric Mean for the Microbial Count for All Treatments During 65 Days of Study



Note. AWW = artificial wastewater.

Conclusion

Sixty centimeters of unsaturated flow was most efficient at removing *E. coli*, while 30 and 45 cm of unsaturated flow were inadequate for treating microbes. Within nine days, the *E. coli* in wastewater reached the water table at 30 cm below the soil surface. Forty-five centimeters of unsaturated flow decreased the microbial counts reaching the water table, but it was very close or exceeded the 200 CFU/100 mL standard during 65 days of wastewater application. With 60 cm of unsaturated flow, *E. coli* concentrations were reduced to an acceptable level during the study period. Based on the results for individual replications of the treatments, we estimated that the probability of groundwater contamination to be 100% when the separation distance above a water table is 30 cm or less, greater than 40% when the distance is 45 cm, and less than 10% when the distance is 60 cm or more. In areas with SHWT, pretreatment of wastewater prior to distribution within the drainfield is a viable option for minimizing groundwater contamination by microbes present in septic system effluent (Duncan, Reneau, & Hagedorn, 1994).

The results of our study show that the longer the distance of the unsaturated flow under septic system trenches, the more effective the soil is in removing anaerobic bacteria from wastewater. Dropping the water table depth, which increases the length of unsaturated flow path, increases the efficacy of the soil to treat AWW for bacterial contamination. In our study, however, *E. coli* concentrations after dropping the water table from 30 to 60 cm were still greater than the concentrations of *E. coli* detected at the water table in the treatment with 60 cm of continuous unsaturated condition for the duration of the experiment. This may be a result of the previously saturated soil acting as a source of contamination. Also, indigenous aerobic bacteria may have been eliminated by the saturated and anaerobic conditions when the water table was high, making the soil ineffective in removing the incoming bacteria. Overall, it does not appear that a significant amount of microbial contamination travels with the descending water table.

Acknowledgements: This research was supported in parts by the Division of Water Quality, EPA 319 Program, NCDENR; and the North Carolina Agricultural Research Service (NCARS). The assistance of Dr. Jeff Thompson for statistical advice is greatly appreciated. The use of trade names in this publication does not imply endorsement by NCDENR or NCARS of the products used or criticism of the ones not mentioned.

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continued on page 100

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Prepublished online June 2013,
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Fecal Contamination of Food, Water, Hands, and Kitchen Utensils at the Household Level in Rural Areas of Peru

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Abstract **The study described in this article evaluated sources of contamination of children's food and drinking water in rural households in the highlands of Peru. Samples from children's meals, drinking water, kitchen utensils, and caregivers' and children's hands were analyzed for total coliforms and *E. coli* counts using Petrifilm EC. Thermotolerant coliforms in water were measured using DelAgua test kits while diarrheagenic *E. coli* was identified using polymerase chain reaction methods (PCR). Thermotolerant coliforms were found in 48% of all water samples. *E. coli* was found on 23% of hands, 16% of utensils, and 4% of meals. Kitchen cloths were the item most frequently contaminated with total coliforms (89%) and *E. coli* (42%). Diarrheagenic *E. coli* was found in 33% of drinking water, 27% of meals, and on 23% of kitchen utensils. These findings indicate a need to develop hygiene interventions that focus on specific kitchen utensils and hand washing practices, to reduce the contamination of food, water, and the kitchen environment in these rural settings.**

Introduction

Diarrheal diseases are among the leading causes of childhood illness and death in developing countries, killing an estimated 1.3 million children less than five years of age annually (Black et al., 2010).

The World Health Organization outlines several aspects critical to the prevention of diarrhea. They include improved drinking water systems and sanitation facilities, improved nutrition (through breast-feeding and better weaning practices), and good personal and domestic hygiene, among others (United Nations Children's Fund/World Health Organization [WHO], 2009). Several studies have demonstrated a high prevalence of bacterial contamination of water and foods within households (Black et al., 1989; Lanata, 2003; Wright et al., 2004), which is likely associated with incidence of infections in susceptible individuals, especially children.

A need exists for effective interventions in developing countries that can minimize food and water contamination at the household level and therefore reduce the rate of diarrhea in these environments (Hunter, 2009; Lanata, 2003). By measuring risky practices and behaviors and identifying kitchen sites, niches, and surfaces that harbor pathogenic microorganisms, we can provide a basis from which to develop effective interventions. The aim of our study was to identify those potential exposures at the household level, specifically those associated with contamination of food, drinking water, kitchen utensils and surfaces, and caregivers' and children's hands. Our study was conducted to inform a subsequent randomized trial that evaluated the health effects of an integrated home-based intervention package in a rural area of Peru. In addition, we tested for the presence of diarrheagenic *E. coli* (Nataro & Kaper, 1998) as an indicator of pathogenic *E. coli* in this setting.

Materials and Methods

Study Site

Our study was conducted in rural communities of San Marcos Province, Cajamarca, situated at 2,200 to 3,900 m above sea level in the highlands of Peru. Daily temperatures ranged from 7.6ºC–25.0ºC during the study period and relative humidity was between 59% and 73%. Agriculture and subsistence farming are the major economic activities in this area. Houses are mud brick structures with clay tile roofs supported by tree rods, earthen floors, and few open windows. A typical house consists of three rooms: a kitchen and dining room, a living and sleeping room, and a storage area. Water supply for about 61% of rural homes in San Marcos comes from a piped gravity system that transports untreated water captured from springs through individual or small-scale collective plastic piping to a tap in the courtyard. Only 9% of households have electricity, 2% have a closed sewage system, and 75% have access to a pit latrine (Instituto Nacional de Estadística e Informática, 2007).

Meals are based mainly on potatoes and other tubers and legumes, eaten with rice or boiled in a soup or a stew. Red meat and chicken are seldom consumed due to their high cost. Animals like dogs, guinea pigs, and chickens roam free in kitchens and households. The latter two are bred at home for sale or reserved for festive meals. Meals are prepared three to four times a day and eaten by adults and children alike. Leftover food is not consumed but discarded or fed to the animals. No time is set at which to start cooking the midday meal. Mothers start cooking anywhere from 8:00 a.m. to 12:00 p.m. and keep the food on the fire until lunch. Meals are served directly from pots to plates using wooden ladles. Kitchen utensils are washed with water brought from an outside faucet in a plastic basin, and a *malla,* a local kitchen cloth, also is used to clean dirty surfaces and caregivers' hands while cooking. The *malla* is kept wet after rinsing in the same washing up water, which is not changed very often.

Most households have access to tap water from a faucet installed in the yard. The gravity-based piped water supply system provides spring water to each household. The water is unfiltered, untreated, and chlorination is uncommon. Drinking water is either consumed directly from the faucet or boiled with herbs for children's consumption only. Hygiene practices include hand washing with water only; soap and detergent are rarely used.

Study Design

Households were identified in 32 communities based on home visits and enrolled by a trained field worker between April and September 2008 if they had a child aged 6 to 35 months. Field workers visited each participating household (N = 64) once, mostly at noon, to sample food, water, and kitchen environments.

Sample Collection

In each household approximately 20 g of each food served to the child was collected. If the child had already eaten, samples were taken from the pot. Between 50 and 100 mL of the child's drinking water and one sample from each of the available kitchen utensils (i.e., dish, cup, pot, cutlery, cutting board, and kitchen cloth) were also collected. For both the child and the caregiver, one hand was rinsed in buffer solution for microbiological testing. Samples were collected following standard procedures (Swanson, Busta, Peterson, & Johnson, 1992; WHO, 1997).

For kitchen surfaces, a 10 x 10 cm area of the cutting board or table and the surface of the utensil that was in contact with the child's food or drink was wiped using a cotton swab moistened with Butterfield's phosphate buffer (BPB) and then placed into a tube containing 10 mL of BPB. Kitchen cloths were collected in a new resealable plastic bag and a 10-cm^2 portion was cut and placed in a sterile plastic bag filled with 100 mL of BPB. To obtain samples from hands, caregivers and children placed one hand into a sterile plastic bag filled with 100 mL of BPB. The hand was massaged for 60 seconds, with emphasis on rubbing between fingers, around the fingernails, and the palm of the hand. All samples were kept in a Styrofoam box with cold packs for transport to the project laboratory in San Marcos City and stored at 8ºC until processing the same day.

Sample Analyses

Food, utensils, and hand samples were analyzed for total coliforms and *E. coli* using Petrifilm *E. coli*/coliform count plates, following standard procedures (Association of Analytical Communities [AOAC], 2000). A 1-mL aliquot of 10-fold dilutions was plated onto a Petrifilm EC plate. The plates were incubated at 35ºC ± 1ºC for 24 hours ± 2 hours to enumerate total coliforms and 48 hours ± 2 hours to enumerate *E. coli*. Water samples were analyzed for thermotolerant (fecal) coliforms using a membrane-filtration method, i.e., the Oxfam DelAgua water testing kit, and results were recorded as *E. coli* (CFU/100mL of water), an indicator for thermotolerant coliforms.

Colony counts were recorded by the on-duty lab microbiologist. Cultures were reread by a second microbiologist. Digital pictures taken from each sample were read by a third microbiologist to decide on a final result in case of discrepancies (more than 10% difference) between the first two counts.

For the detection of diarrheagenic *E. coli*, five colonies per sample were saved in peptone media vials for further characterization. From the Petrifilm EC plate, priority was given to typical *E. coli*-like colonies (blue colonies with gas) (AOAC, 2000); however, other coliforms were saved if less than five typical *E. coli*-like colonies were present. The peptone media vials were transported to the Enteric Diseases and Nutrition Laboratory at the Tropical Medicine Institute, Cayetano Heredia University, Lima, for analysis using a real-time polymerase chain reaction (PCR) multiplex system (Guion, Ochoa, Walker, Barletta, & Cleary, 2008), which detects virulence genes of enterotoxigenic *E. coli* (ETEC), enteroinvasive *E. coli* (EIEC), enteropathogenic *E. coli* (EPEC), Shiga-toxin–producing *E. coli* (STEC), enteroaggregative *E. coli* (EAEC) and diffuse-adherent *E. coli* (DAEC). The multiplex PCR was done in a five-colony pool per sample (Barletta et al., 2009).

TABLE **1**

Total Coliforms and *E. coli* in Food, Water, Utensils, and Hands From Rural Households of Peru

Sample Type	Total Coliforms			*E. coli*		
	% (*n/N*)	Geometric Mean	Ranges	% (*n/N*)	Geometric Mean	Ranges
Child meals		**CFU/g**	**CFU/g**		**CFU/g**	**CFU/g**
Salad	67 (2/3)	4.4 x 10	10^2	0 (0/3)		0
Dairy	44 (4/9)	8.1 x 10	10^2–10^9	22 (2/9)	4.2	10^0–10^7
Tuber cooked/fried	21 (3/14)	1.6	10^1–10^2	0 (0/14)		0
Rice	18 (2/11)	1.2	10^1–10^2	0 (0/11)		0
Soup	17 (2/12)	1.7	10^2–10^3	8 (1/12)	1.0	10^3
Toasted bread	11 (1/9)	0.8	10^1	0 (0/9)		0
Oat	9 (1/11)	1.4	10^4	0 (0/11)		0
Stew	0 (0/8)		0	0 (0/8)		0
All child meals	19 (15/77)		0–10^9	4 (3/77)		0 –10^7
Drinking water	N/A	N/A	N/A	48 (10/21)	2.6*	10^0 –10^2*
Kitchen utensils		**CFU/utensil†**	**CFU/utensil†**		**CFU/utensil†**	**CFU/utensil†**
Kitchen cloth	89 (17/19)	1.2 x 10^4‡	10^0–10^7‡	42 (8/19)	1.2 x 10‡	10^0–10^5‡
Washing basin	70 (7/10)	2.1 x 10	10^1–10^3	10 (1/10)	1.0	10^2
Water jar	69 (9/13)	1.3 x 10^2	10^1–10^9	15 (2/13)	1.2	10^1–10^2
Pot	64 (7/11)	6.3 x 10^2	10^1–10^9	18 (2/11)	1.4	10^0–10^3
Spoon	64 (9/14)	2.9 x 10	10^1–10^3	21 (3/14)	1.3	10^1–10^2
Dish	58 (7/12)	1.2 x 10^2	10^1–10^9	8 (1/12)	0.6	10^1
Cup	50 (6/12)	2.5 x 10	10^0–10^7	8 (1/12)	0.5	10^0
Bottle's nipple	45 (5/11)	2.4 x 10	10^1–10^9	9 (1/11)	1.1	10^3
Cutting board	43 (6/14)	2.0 x 10‡	10^0–10^5	14 (2/14)	0.8‡	10^0‡
Ladle	28 (5/18)	2.2	10^1–10^3	6 (1/18)	0.6	10^1
All kitchen utensils	58 (78/13)		10^0–10^9	16 (22/134)		10^0–10^5
Hands		**CFU/hands**	**CFU/hands**		**CFU/hands**	**CFU/hands**
Caregiver	76 (16/21)	2.8 x 10^2	10^1–10^5	29 (6/21)	4.8	10^1–10^4
Child	55 (12/22)	2.2 x 10	10^1–10^4	18 (4/22)	1.4	10^1–10^3
All hands	65 (28/43)		10^1–10^5	23 (10/43)		10^1–10^4

*Thermotolerant (fecal) coliform CFU/100 mL.
†Area of utensil in contact with food/drink.
‡CFU/100 cm^2.

Data Analysis

Geometric means of the colony counts (total coliforms and *E. coli*) for each type of sample were calculated. A value of 0.5 was assigned to all samples with zero colony counts to allow for calculations. Proportional differences were analyzed by Chi-square tests with Yates's correction or by two-tailed Fisher's exact test using Epi Info version 6 statistical package.

Results

A total of 275 samples (134 from kitchen utensils, 77 from children's meals, 43 from hands, and 21 from children's drinking water) from 64 households were analyzed. The frequency of contamination with total coliforms and *E. coli* by type of sample is presented in Table 1. Total coliforms were significantly more present on hands (65%) and on kitchen utensils (58%) than in children's meals (19%); $p < .01$. Kitchen cloths (89%, 17/19) and caregivers' hands (76%, 17/19) were the individual samples most frequently contaminated with total coliforms. The frequency of *E. coli* in drinking water (48%) was significantly higher than that of kitchen utensils (16%, $p = .002$) and children's meals (4%, $p < .0001$). No statistical difference was observed, however, when comparing drinking water and all hands ($p = .09$). Kitchen cloths were most frequently contaminated with *E. coli* (42%), with a geometric mean of 1.2 x 10^4 CFU/100 cm^2.

A total of 108 samples were tested for diarrheagenic *E. coli*. DAEC was the most frequent type identified (9/108), followed by ETEC (8/108), EIEC (4/108), STEC (3/108), and EAEC (1/108). Overall, at least one type of diarrheagenic *E coli* was detected in 20% of all tested samples, including in 33% (2/6) of children's drinking water, 27% (3/11) of children's meals, 23% (14/60) of kitchen utensils, and 10% (3/31) of hands.

Discussion

Our study describes the high frequency of microbiological contamination of water and food consumed by children in parts of rural Peru, and indicates an important potential cause of diarrhea. A high percentage (48%) of the water consumed by children was often boiled with herbs and subsequently kept in jars or pots, but contained thermotolerant coliforms. Dairy products and boiled soups also had remarkably high *E. coli* counts (up to 10^7 CFU/mL in dairy). The source of these contaminants likely originates from contaminated kitchen utensils including plates, spoons, pots, or jars, as well as *mallas*, the local kitchen cloths. Children's and caregivers' hands were also contaminated with *E. coli* due to poor hygiene practices.

Our study had some limitations. Sampling was conducted during the dry season (April to September), and not during the rainy season (December through March). Hence, seasonal variations in water and food contamination were not captured. Study conditions allowed for only a small number of convenience samples from each type of food or kitchen utensil, which is sufficient for descriptive purposes, but limited for giving precise estimates. Sampling centered on the midday meal for logistical reasons. It is possible that meals prepared in the early morning or in the evening may have had different levels of contamination, influenced by cooler temperatures at those times. Future studies would need to sample children's meals over a 24-hour period and ideally, repeatedly, in order to fully describe the level and variability of food contamination in these households. Study conditions did not allow for serial sample collection before and after food preparation and at the time of serving to children, which would have allowed us to identify the critical control points to minimize or eliminate the risk of contamination in a hazard analysis and critical control point system (Bryan, 1981).

Few studies (Adachi, Mathewson, Jiang, Ericsson, & DuPont, 2002; Vigil et al., 2009) have attempted to identify diarrheagenic *E. coli*—the strains of *E. coli*—in environmental samples (food, water, and utensils), using molecular and specific PCR methods. We tested for these groups of pathogens by PCR, based on a presumptive identification of *E. coli*-like colonies and coliforms. We found only a small number of colonies with diarrheagenic *E. coli* strains. It is unclear whether the lower isolation rates found are real or are due to low sensitivity in our selection of *E. coli*-like colonies. These results suggest that risk estimates based on total coliform or *E. coli* counts overestimated the true risk of diarrheal diseases from food and water due to pathogenic *E. coli*.

Despite these limitations, the results of our study are comparable to others from developing country settings, where weaning food and water in households were frequently found to be contaminated with fecal matter (Clasen et al., 2003; Kung'u et al., 2009; Rufener, Mäusezahl, Mosler, & Weingartner, 2010). In a study conducted in peri-urban Lima, Peru (Black et al., 1989), weaning food was found to be contaminated with *Salmonella* spp., *Vibrio cholerae* non-O1, and ETEC originating from secondary contamination of kitchen utensils after food preparation. Foodborne illnesses are associated with food preparation too far in advance of consumption (allowing growth of pathogens present in the food to levels exceeding the minimal infectious dose), improper cooling, and inadequate reheating (Lanata, 2003). In our study communities, food stuffs and leftovers were not stored for second servings, since cooking was done three to four times per day; however, food samples collected at eating time directly after cooking were found to be contaminated. This could be explained by the high frequency of contamination found on kitchen surfaces and utensils, most likely due to the washing up process: washing up in a plastic basin with untreated and unchanged water leaves food residuals behind as a source for bacterial growth. Other studies have shown how common cross contamination is in the kitchen through contaminated water used to clean dishes (Beumer & Kusumaningrum, 2003).

Our study indicates that kitchen cloths may present a significant yet underrecognized source of contamination of kitchen utensils, since cloths are used all over the kitchen to wipe dirty surfaces as well as hands and remain wet after rinsing in the same washing-up water. In other settings, kitchen cloths were identified as vehicles for pathogens that were able to survive for extended periods of time (Kusumaningrum, van Putten, Rombouts, & Beumer, 2002; Mattick et al., 2003). Food safety interventions in these communities should focus on kitchen hygiene practices, hand washing, safe food preparation, and safe handling of cooked food.

Conclusion

The prevalence of fecal contamination of food and drinking water given to children highlights the need for improving domestic hygienic practices, like hand washing and cleaning kitchen utensils, to prevent diarrheal diseases transmitted through the fecal-oral route. Effective interventions to reduce contamination of the kitchen environment should be developed. Further studies are needed on the correlation between diarrheagenic *E coli* identification as detected by PCR and the traditional culture method for detecting fecal coliforms in food and water. In a related study, we will evaluate the impact on the rate of diarrheal diseases in young children of an intervention designed to improve water availability in the kitchen environment through kitchen sink installation, using point-of-use water disinfection by solar exposure. Further effects of promoting hand washing with soap or detergent and improving hygiene practices in the kitchen will also be studied.

Acknowledgements: We would like to extend our thanks to the families who participated in the study for their contributions; to the field workers for their dedication to obtaining the samples; to Mrs. Selenne Flores, the study field supervisor, for her care and diligence in getting this study done; to Dr. Jan Hattendorf, for his help revising the manuscript; and to Amena Briet for language editing. We would also like to express our appreciation to the local authorities and the Cajamarca Health Region for their support throughout the study. A grant from UBS Optimus Foundation, Zurich, Switzerland, supported this work. The ethical review boards of the Instituto de Investigacion Nutricional, Lima, Peru, and the cantonal ethical review board of Basel, Switzerland, approved the study.

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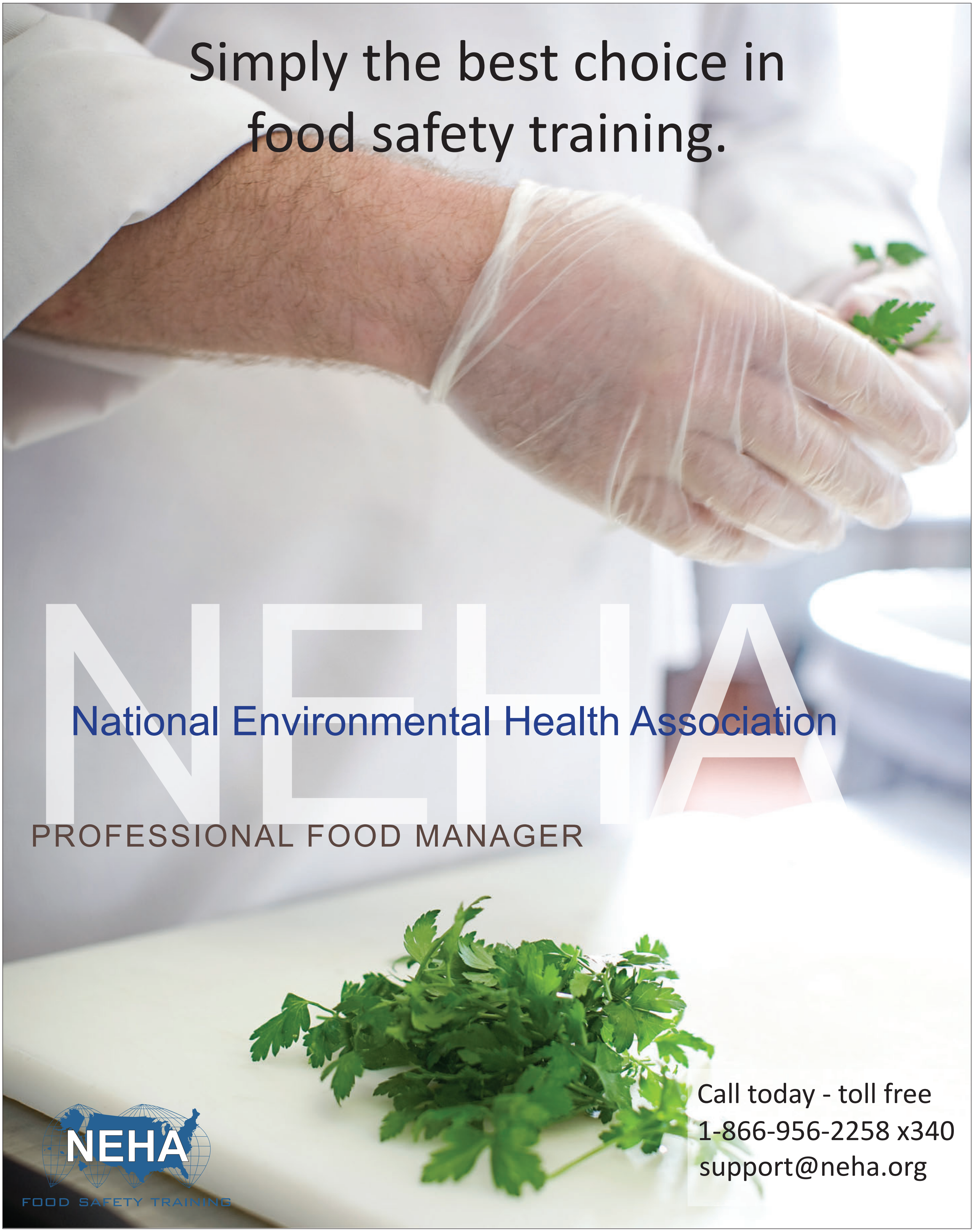
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Assessment of Nonzoonotic Soil-Transmitted Helminth Levels in Soils in Yenagoa Metropolis, Niger Delta

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Abstract

In order to assess the prevalence of nonzoonotic soil-transmitted helminths in the Yenagoa Metropolis, 480 soil samples were collected from five communities for 12 months. The soil samples were collected along two transects from the waterfront and community playgrounds. Analysis was by standard methods. The results obtained from the study described in this article showed that 44.79% (95% confidence interval [*CI*] = 40.34%–49.24%) of the soil samples tested positive for nonzoonotic soil-transmitted helminths. *Ascaris lumbricoides* was the most common helminth with a prevalence rate of 35% (95% *CI* = 30.73%–39.27%). Mixed occurrence of nonzoonotic soil-transmitted helminths was 10.21%. Although the community playgrounds had a higher prevalence of nonzoonotic soil-transmitted helminths than the waterfront ($p > .05$), more cases of mixed occurrence of nonzoonotic soil-transmitted helminths occurred in the waterfront than the community playgrounds ($p > .05$). The wet season had a higher prevalence rate of nonzoonotic soil-transmitted helminths than the dry season ($p < .05$). The observed high prevalence of nonzoonotic soil-transmitted helminths in soil is considered a potential public health risk to swimmers and children playing outdoors in the Yenagoa metropolis.

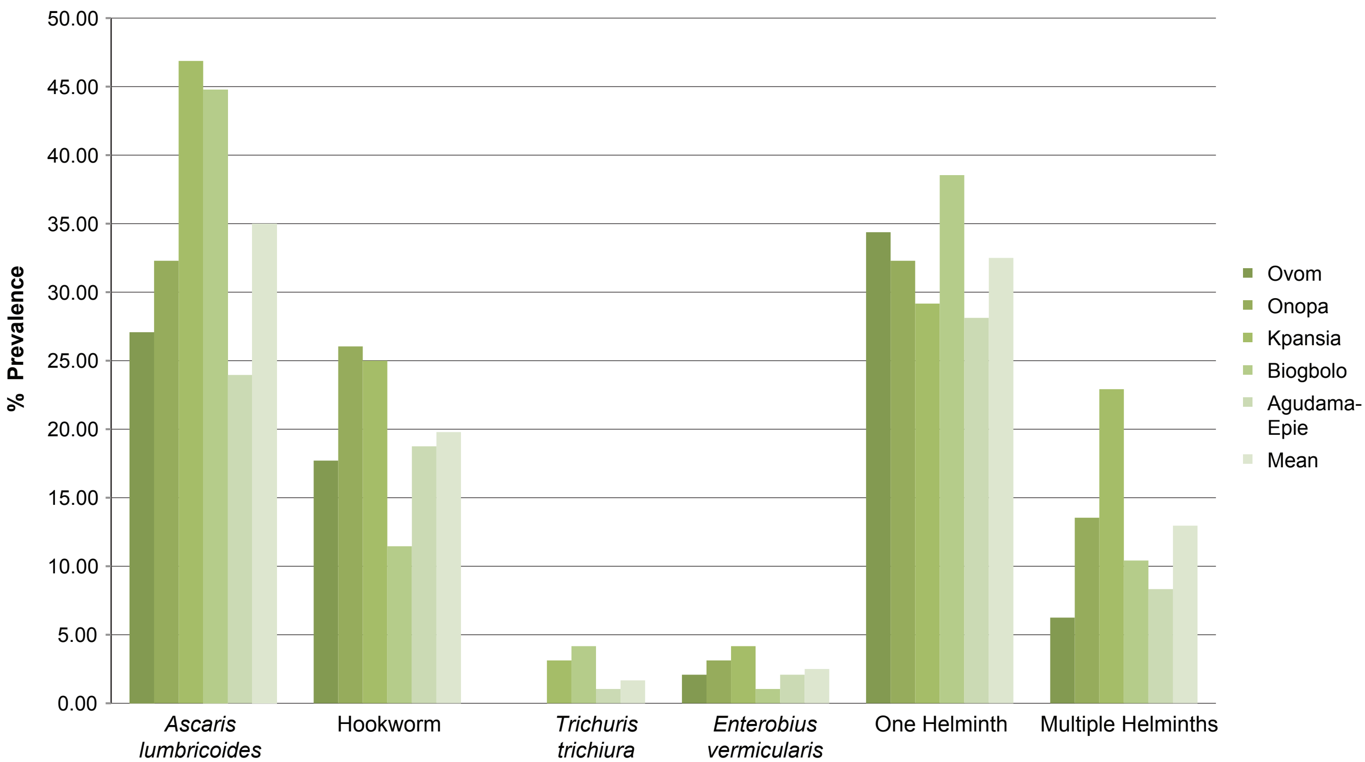
Introduction

Soil-transmitted helminths are known worldwide as a public health hazard, particularly in developing countries. The World Health Organization (WHO, 2002) estimated that more than one billion of the world's population are chronically infected with soil-transmitted helminths and another two billion are at risk (Saka, Abdulraheem, Akanbi, & Musa, 2006). Soil polluted with fecal material is instrumental in the transmission of soil-transmitted helminths (Baig, Rana, Zaki, & Khan, 2007; Ulukanligil, Seyrek, Aslan, Ozbilge, & Atay, 2001). Soil pollution and soil-transmitted helminths are facilitated where poor peri-domestic/environmental sanitation and personal hygiene, poverty, and low levels of education exist, among other factors (Egwunyenga & Ataikiru, 2005; Ozumba, Ozumba, & Anya, 2005; Saka et al., 2006; United Nations Development Programme [UNDP], 2006). These conditions are commonplace in slums and squatter settlements as well as in poorly planned urban areas in both developed and developing countries. According to the United Nations Development Programme report (UNDP, 2006), some 1.1 billion people in developing countries have inadequate access to water and another 2.6 billion lack basic sanitary facilities. Close to half of all people in developing countries suffer at any given time from a health problem caused by water and sanitation deficits.

Yenagoa, the capital city of Bayelsa State in the Niger Delta, lies between latitude 4º50´ and 5º05´ N and longitude 6º15´ and 6º30´ E. It is within the freshwater swamp forest ecozone (Capital City Development Authority [CCDA], 2007) and experiences equatorial climatic conditions. Rainfall

FIGURE **1**

Prevalence of Nonzoonotic Soil-Transmitted Helminths in Soils in the Yenagoa Metropolis



is usually about 2,000–4,000 mm per year and relative humidity varies from 65% in the dry season to 80% in the wet season (Oyegun, 1999). The average daily temperature of the area is about 29°C. The prevalence rate of human intestinal soil-transmitted helminths in primary school pupils has been reported to be 54.24% (Bariweni, Ekweozor, & Bassey, 2009).

Yenagoa, like other urban centers in developing countries, is characterized by high levels of unemployment, shortage of housing, and poor management of waste. Many houses were built without regard to minimum building codes or standards prior to its state capital status. Because of poverty and low levels of environmental awareness, many of the residents dispose of their bodily wastes and other domestic wastes directly into the creeks (Bariweni, Izonfuo, & Amadi, 2002). Thus the greatest problem with housing in the city is the lack or inadequate provision of toilet facilities and potable water. The *Yenagoa Master Plan* (Bayelsa State of Nigeria, 2004) reported that 20.6% of the houses in the city had no toilets. It is therefore a usual practice for fecal wastes to be thrown into the nearby bushes or river. Such practices are known to result in very poor sanitary conditions.

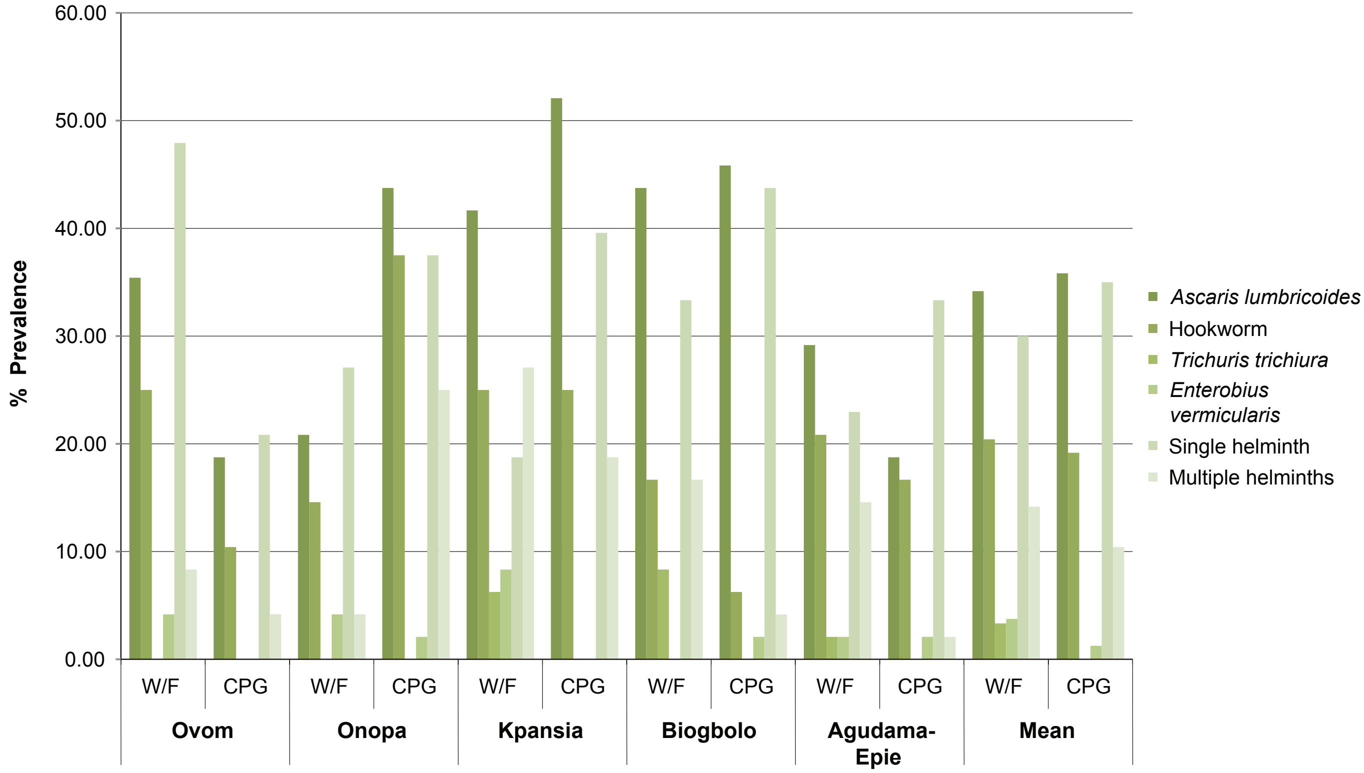
Another basic problem in the city is the high water table, which is almost at the land surface at the peak of the rainy season. About 70% of Bayelsa State is usually flooded during the rainy season (CCDA, 2007), therefore many of the built-up areas can be better described as back swamps. In the worst areas, the septic tanks of many buildings often become inundated with groundwater, causing fecal waste to overflow to the land surface at the peak of the rainy season. Special sanitation problems then arise after heavy rain when runoff water spreads animal wastes, street debris, town chemicals, and other fecal wastes since no drainages exist to channel water to appropriate destinations. The poor sanitary conditions combined with the prevailing climate favor the survival of parasitic organisms such as the nonzoonotic (human intestinal) soil-transmitted helminths. Our study was therefore carried out to assess the prevalence of soil-transmitted helminths in soils and the potential health risks to the human population in Yenagoa.

Methods

In order to assess the distribution and prevalence of nonzoonotic soil-transmitted helminths in Yenagoa, 480 soil samples were collected from five communities: Ovom, Onopa, Kpansia, Biogbolo, and Agudama-Epie. Forty soil samples were collected randomly with a hand auger from the uppermost layers (0–15 cm) of soils monthly for 12 months from five communities in the

FIGURE **2**

Distribution of Nonzoonotic Soil-Transmitted Helminths in Waterfronts (W/F) and Community Playgrounds (CPG) in the Yenagoa Metropolis



Yenagoa Metropolis. Eight soil samples each (four from the community playgrounds and four from the waterfront) were collected along two transects from the five communities, labeled, and immediately transported to the laboratory for microscopic analyses. All soil samples were analyzed by the flotation technique (Zenner, Gounel, & Chauve, 2002) using zinc sulfate solution (density 1.18–2.00) as the flotation medium. Results were reported as proportions (in percentages) of the samples that were found positive for soil-transmitted helminths and confidence intervals determined. Proportions were also assessed for significance by the *z*-test for proportions (Bluman, 2004) and *p*-values <.05 were regarded as significant.

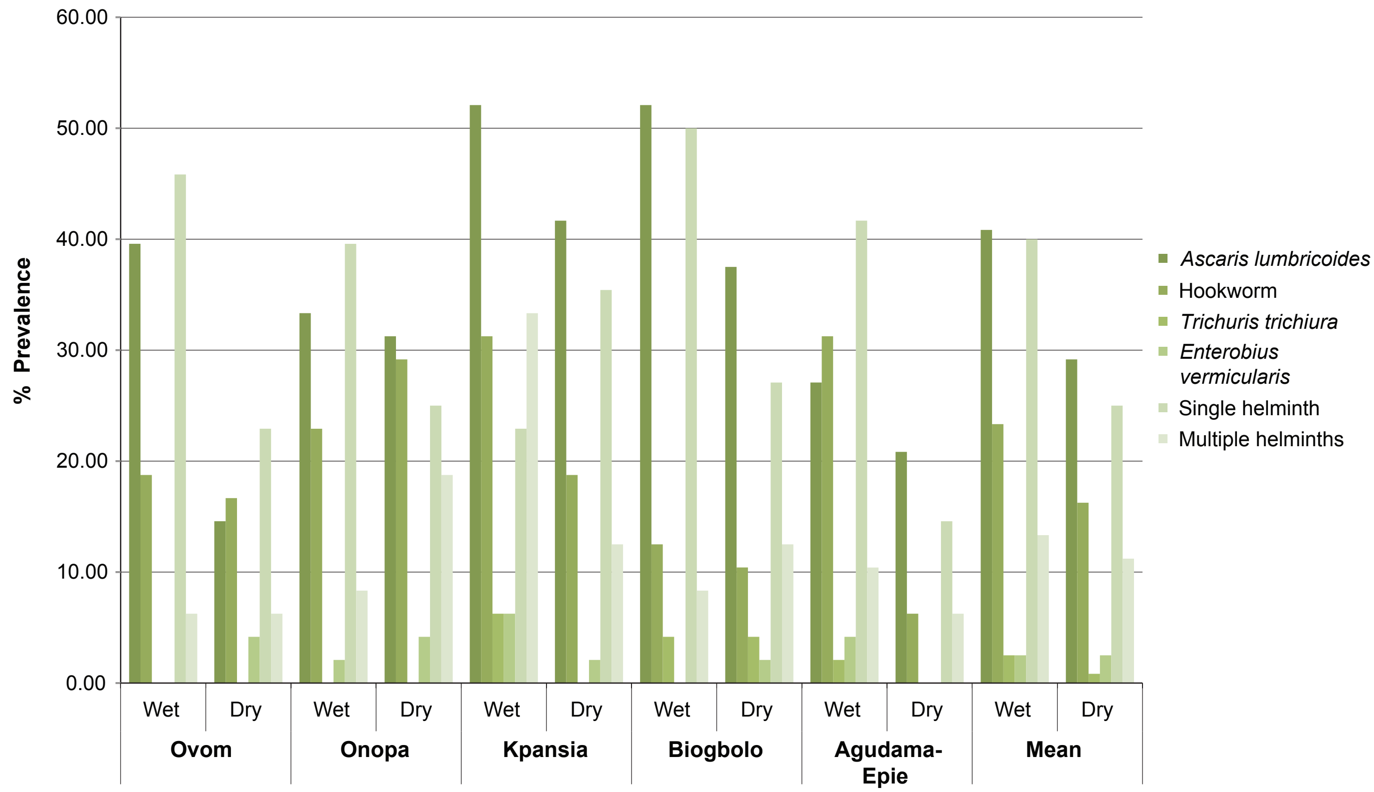
Results and Discussion

Results from the study (Figure 1) showed that 44.79% (95% confidence interval [*CI*] = 40.34%–49.24%) of the soil samples tested positive for nonzoonotic soil-transmitted helminths. This ranged from a prevalence rate of 36.46% in Agudama-Epie to 52.08% in Kpansia. The observed prevalence rate in the various communities was not significantly different from the prevalence level of 44.79% for the entire Yenagoa Metropolis ($p > .05$) except in Kpansia and Ovom (Figure 1). *Ascaris lumbricoides* was the most common helminth with a prevalence of 35% (95% *CI* = 30.73%–39.27%). This was followed by hookworm (19.79%; 95% *CI* = 16.23%–23.35%), *Enterobius vermicularis* (2.5%; 95% *CI* = 2.11%–2.89%), and *Trichuris trichiura* (1.67%; 95% *CI* = 1.34%–2%). The fact that *A. lumbricoides* was the most common is not surprising. Although hookworm is the most dependent on soil (Alexander & Strete, 2001), *A. lumbricoides* is the most prodigious in terms of egg production and most resistant to environmental stress (Schmidt & Roberts, 2000) among the helminths identified in our study. *A. lumbricoides* was also the most common helminth (35.25%) in stool samples collected from primary school children in Yenagoa (Bariweni et al., 2009).

The mean occurrence rate of soil-transmitted helminths in the soil samples was 32.5% (95% *CI* = 28.31%–36.69%) for a single helminth and 13.96% (95% *CI* = 7.5%–12.92%) for more than one helminth. These results are at variance with the findings of Ulukanligil and co-authors (2001) in which a higher prevalence of multiple occurrences was present than single occurrences of soil-transmitted helminths. Hookworm, which is usually transmitted where people walk barefooted because of its mode of transmission by skin penetration (Rottier & Ince, 2007), was highest in Onopa (26.04%) and closely followed by Kpansia

FIGURE 3

Seasonal Distribution of Nonzoonotic Soil-Transmitted Helminths in Soils in the Yenagoa Metropolis



(25%). The prevalence of multiple occurrences of soil-transmitted helminths in soil samples ranged from 6.25% in Ovom to 25% in Kpansia for multiple helminths. The observed levels of soil-transmitted helminths in the soil samples were found to be lower than the 84.4% levels found in soils in Sanliurfa, Turkey (Ulukanligil et al., 2001). The prevalence rate observed in the soil samples was also lower than the prevalence rate in primary school children (54.24%) in Yenagoa (Bariweni et al., 2009) but this was not significant ($p > .05$).

With respect to the distribution of nonzoonotic soil-transmitted helminths in the waterfront and community playgrounds in Yenagoa Metropolis (Figure 2), results showed that the mean prevalence of nonzoonotic soil-transmitted helminths was 34.17% in the waterfront and 35.42% in the community playgrounds. This ranged from 31.25% in Onopa to 56.25% in Ovom in the waterfront and from 25% in Ovom to 60.42% in Onopa in the community playgrounds. The trend observed was found to be similar to the findings of Ulukanligil and co-authors (2001) in which the community playground had a higher prevalence (76.6%) than the river bank (70%).

Although the community playgrounds had a higher mean prevalence of soil-transmitted helminths than the waterfront ($p > .05$), all the soil-transmitted helminths identified in our study showed higher prevalence in the waterfront than the community playgrounds except *A. lumbricoides,* which had a higher prevalence in the community playgrounds than the waterfront ($p > .05$). The waterfront also had more cases of multiple occurrences of soil-transmitted helminths than the community playgrounds ($p > .05$). This may be explained by the fact that the waterfront was wetter and therefore provided a more favorable environmental condition for the helminth eggs to survive than the community playgrounds, which were drier. *T. trichiura*, which is the least prevalent helminth, was not found in any of the community playgrounds. This statistic compared well with the findings of Ulukanligil and co-authors (2001) in which *T. trichiura* was also not found in the soils. *T. trichiura* eggs are less resistant to environmental stress such as drying or direct exposure to sunlight (Klass, 1987), which to a reasonable extent explains their absence in the soils of the community playgrounds.

Results regarding the seasonal distribution of nonzoonotic soil-transmitted helminths in Yenagoa Metropolis (Figure 3) showed that the prevalence of soil-transmitted helminths ranged from 47.92% in Onopa to 58.33% in Biogbolo during the wet season (May to October), and from 20.83% in Agudama-Epie to 47.92% in Kpansia during the dry season (November to April). The wet season generally had a higher mean prevalence of soil-transmitted helminths (53.33%) than the dry season (36.25%) ($p < .05$). A cursory look at the results also revealed that all the soil-transmitted helminths identified in our study had a higher distribution in the wet season than the dry season ($p > .05$), except *E. vermicularis* for which no difference existed between the wet and dry season ($p < .05$). A higher prevalence of multiple occurrences of soil-transmitted helminths also occurred in the wet season than the dry season ($p > .05$). This can be explained by the fact that the wet season provided a more favorable environmental condition in addition to the effect of runoff water, which enhanced the spread of ova of helminths during the rainy season compared to the dry season.

Conclusion

It may be concluded that the high prevalence of soil-transmitted helminths in soil in Yenagoa Metropolis is a high potential health risk to the population, especially children who play in the community playgrounds and to swimmers and tourists who use the waterfront for recreation. Furthermore, it may be concluded that the risk of helminthic infection is generally higher in the rainy season than in the dry season. According to the World Health Organization (2002), children are especially at risk due to their direct exposure to soil and other helminth-carrying material. The high prevalence of soil-transmitted helminths in soil may be responsible for the heavy worm burden (54.24%) found in primary school children in the city (Bariweni et al., 2009). Another cause for concern is the prolonged existence of soil-transmitted helminths, especially *A. lumbricoides* in the environment. As stated by Schmidt and Roberts (2000), because of the longevity of *A. lumbricoides*, it is impossible to prevent reinfection when the peridomestic environments have been liberally seeded with eggs, even when proper sanitation habits are initiated later. The cycle of infection and reinfection may therefore continue to remain a public health concern for a long time.

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INTERNATIONAL PERSPECTIVES

Prepublished online September 2013, National Environmental Health Association.

Risk Assessment of Rooftop-Collected Rainwater for Individual Household and Community Use in Central Kerala, India

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Abstract **Water quality of rooftop-collected rainwater is an issue of increased interest particularly in developing countries where the collected water is used as a source of drinking water. Bacteriological and chemical parameters of 25 samples of rooftop-harvested rainwater stored in ferrocement tanks were analyzed in the study described in this article. Except for the pH and lower dissolved oxygen levels, all other physicochemical parameters were within World Health Organization guidelines. Bacteriological results revealed that the rooftop-harvested rainwater stored in tanks does not often meet the bacteriological quality standards prescribed for drinking water. Fifty percent of samples of harvested rainwater for rural and urban community use and 20% of the samples for individual household use showed the presence of *E. coli*. Fecal coliform/fecal streptococci ratios revealed nonhuman animal sources of fecal pollution. Risk assessment of bacterial isolates from the harvested rainwater showed high resistance to ampicillin, erythromycin, penicillin, and vancomycin. Multiple antibiotic resistance (MAR) indexing of the isolates and elucidation of the resistance patterns revealed that 73% of the isolates exhibited MAR.**

Introduction

Though 70% of the earth's geographical area is covered by water, only 1% of it is potable; the rest is unsafe for consumption. Lack of investment, growing water demand, overexploitation of existing sources, pollution, and maintenance problems make the supply of potable water in developing countries extremely difficult to obtain. The availability of an adequate supply of safe water is fundamental to the development process in all sectors with benefits such as improved labor productivity (Gadgil, 1998). The microbial quality of many drinking water sources in India, both groundwater and surface water, is affected by human activities (Pushpangadan, 2003) and it is reported that nearly 44 million people in India are directly affected by water quality problems, either due to bacteriological or chemical pollution (Nigam, Gujja, Bandyopadhyay, & Talbot, 1998).

Due to decreasing supply and the ubiquitous contamination of surface and groundwater resources by microbial and chemical contaminants, rainwater harvesting has become more relevant now in areas that enjoy high rainfall. Rainwater harvesting can provide a renewable supply of natural, soft, clear, and odorless water that could be used for a range of purposes and could represent the primary source of household water in some areas. Governmental agencies across the world are now introducing policies to promote increased use of rainwater. In India, awareness is growing

Rainwater-Harvesting Tank in Study Area

of the potential of rainwater harvesting to meet the demand of safe water throughout the country, especially in rural locations. Several state governments including in Kerala have introduced legislation that makes it obligatory to incorporate rooftop harvesting systems in newly constructed buildings in urban areas. Governments are also providing subsidies to promote the use of rainwater harvesting systems.

The National Sample Survey Organization (1999) has reported that the most significant risk to human health related to drinking water quality is from microbiological sources through fecal contamination. In general the quality of rainwater is not treated for bacteriological quality and is assessed at the household level because of the presence of leaves and other materials such as mosquito larvae, insects, rodents, frogs, etc. The World Health Organization (WHO) proposed appropriate treatment techniques to use harvested rainwater as a safe drinking water source (WHO, 2006). Researchers revealed the value of solar disinfection (SODIS) as a low-cost, sustainable, and simple method of treating contaminated water in developing countries (Acra, Jurdy, Muallem, Karahagopian, & Raffoul, 1989; Acra, Raffoul, & Karahagopian, 1984; Sommer et al., 1997).

Though Kerala receives adequate annual rainfall, many parts of the state are experiencing severe drinking water shortages due to poor water management. As a proactive measure, the government of Kerala is promoting rainwater harvesting in rural and urban areas for household use and community use in schools. Monitoring of this rooftop-collected water, however, is not carried out. Hence our study had an objective to determine the bacteriological and nutrient quality of rooftop-harvested and stored rainwater for individual household use as well as for community use in rural and urban settings. General bacterial flora of rooftop-collected water were characterized and the risk assessment of these strains was carried out by drug resistance analysis.

Methods

Collection of Samples

The rooftop-harvested rainwater stored in ferrocement tanks (see photo above) was collected from rural and urban areas of Kottayam, Alappuzha, and Eranakulam districts of Kerala. Ferrocement tanks are made of a thin sheet of cement mortar reinforced with a cage of wire mesh and steel bars. The rainwater from the rooftop is directed to the tank by gutters and polyvinyl chloride pipes to the inlet of ferrocement tanks. Except the inlet and outlet portions, all the sides are closed to prevent groundwater entry to the tanks. The samples were aseptically collected in sterile bottles from the outlet pipe of the tanks and transported to the laboratory in an ice box. Nine storage tanks for individual household use and eight storage tanks each for rural and urban community use were selected at random for sample collection.

Analysis of Physicochemical Parameters

Temperature was recorded with the help of a Celsius thermometer and the pH was monitored with the help of an electronic pH meter. Turbidity was measured using nephelometric turbidity units and total dissolved solids (TDS) and electrical conductivity was measured using a conductivity meter. Acidity, alkalinity, chlorinity, hardness, dissolved oxygen (DO), phosphate, nitrate, and nitrite were measured using standard methods (Clesceri, Greenberg, & Eaton, 1998).

Bacteriological Analysis

Total Heterotrophic Bacterial Count

Water samples were serially diluted aseptically up to 10^{-2} using sterile distilled water. Aliquots of 0.2-mL samples from each dilution were spread plated in triplicate on R2A medium for the enumeration of total aerobic heterotrophic bacteria, which is expressed as total viable count (TVC). The plates were then incubated at room temperature (around 30°C) for 48–72 hours. After incubation, plates with 30–300 CFUs were selected for counting and isolation of bacterial cultures.

Isolation and Identification of Bacterial Isolates

After recording the morphological characters and pigmentation, representative types constituting at least 10–20 numbers of colonies on plates were selected and restreaked onto R2A plates to ensure purity. All the purified isolates were maintained on R2A slants for further characterization and identified to generic level using the taxonomic key for identification by Barrow and Feltham (1993), Holt and co-authors (2000), and Harley and Prescott (2002). Further characterization of the members of the family Enterobacteriaceae was not carried out; however, it was considered as a single genus for counting the number of genera present in the samples. The isolates were characterized based on Gram staining, spore staining, motility test, Kovac's oxidase test, oxidation fermentation test, and catalase test (Jeena, Deepa, Mujeeb Rahiman, Shanthi, & Hatha, 2006).

Analysis of Fecal Coliform (Most Probable Number [MPN] Method)

The MPN load of fecal coliform (FC) bacteria was determined by three-tube dilution method using *E. coli* (EC) broth as the medium (Hatha, Chandran, & Mujeeb Rahiman, 2004). Ten mL, 1 mL, and 0.1 mL of water samples were inoculated into respective dilution tubes containing inverted Durham's tubes. Ten-mL samples were inoculated into 10 mL double strength EC broth; 1-mL and 0.1-mL samples were inoculated into single strength EC broth of 10 mL each. Inoculated tubes were incubated at 44.5ºC for 24 hours and observed for growth and gas production. Tubes that showed growth and gas production were recorded as FC positive and used for calculating the MPN index. The density of FC was expressed as MPN per 100 mL of water.

TABLE **1**

Physicochemical Characteristics of the Rooftop-Collected and Stored Rainwater for Individual Household and Community Use

Parameter Analyzed	Source of Harvested Rainwater		
	Rural Individual	Rural	Urban
	Household Use (n = 9)*	Community Use (n = 8)*	Community Use (n = 8)*
pH	9.03	9.06	8.51
Temperature (ºC)	28.78	28.93	29.03
Conductivity (mS[a])	0.09	0.21	0.18
Total dissolved solids (ppm[a])	52.85	101.99	88
Turbidity (NTU[a])	0.67	0.89	0.5
Acidity (mg/L)	0.56	0.00	0.63
Alkalinity (mg/L)	47.22	60.63	55.63
Chloride (mg/L)	9.16	9.05	15.62
Salinity (mg/L)	16.54	16.36	28.21
Hardness (mg/L)	40.78	40.13	56.38
Dissolved oxygen (mg/L)	3.83	4.16	3.76
Nitrite (mg/L)	0.0023	0.0741	0.0329
Nitrate (mg/L)	1.4185	1.5593	6.9100
Phosphate (mg/L)	0.0512	0.0142	0.0232

[a]mS = milli Siemens; ppm = parts per million; NTU = nephelometric turbidity units.
*Mean values of parameters.

Isolation of E. coli

In order to confirm the presence of *E. coli*, positive FC tubes in the presumptive MPN tests were streaked on eosin methylene blue (EMB) agar, and incubated at 37ºC for 24 hours. After incubation the plates were observed for typical *E. coli*–like colonies (colonies with green metallic sheen). Whenever present at least two colonies per plate were picked up and restreaked to ensure purity and stored on nutrient agar vials for further biochemical characterization using indole, methyl red, Voges-Proskauer, and citrate (IMViC) test. The cultures giving + + - - reaction for IMViC test were confirmed as *E. coli* (Hatha et al., 2004).

Analysis of Fecal Streptococci (MPN Method)

The MPN load of fecal streptococci (FS) bacteria was determined by the three-tube dilution method using azide dextrose broth (ADB) as medium (Hatha et al., 2004). Ten-mL, 1-mL, and 0.1-mL samples were inoculated into respective dilution tubes. Ten-mL samples were inoculated into 10 mL double strength ADB; 1-mL and 0.1-mL samples were inoculated into single strength ADB of 10 mL each. Inoculated tubes were incubated at 37ºC for 24 hours and observed for turbidity in the medium. Tubes showing turbidity in the medium were recorded as FS positive and used for calculating the MPN index. The density of FS was expressed with MPN per 100 mL of water.

Risk Assessment of Bacterial Isolates From Harvested Rainwater

Risk assessment of bacterial isolates from harvested rainwater was determined by drug resistance analysis. A total of 100 bacterial isolates from various rainwater samples collected from the different rainwater harvesting tanks were tested against 12 antibiotics by using disc diffusion method (Bauer, Kirby, Sherris, & Turck, 1966). The following are the antibiotics and their concentrations used for our study: ampicillin (A, 10 mcg); amikacin (Ak, 30 mcg); chloramphenicol (C, 30 mcg); ciprofloxacin (Cf, 5 mcg); erythromycin (E, 10 mcg); gentamicin (G, 10 mcg); kanamycin (K, 30 mcg); nalidixic acid (Na, 30 mcg); penicillin G (P, 10 units); streptomycin (S, 10 mcg); tetracycline (T, 30 mcg); and vancomycin (Va, 30 mcg). The isolates were inoculated into tryptic soy broth and enriched by incubating at 30ºC for 6–8 hours. Using a sterile cotton swab, the enriched cultures were swabbed onto the surface of sterile Muller-Hinton agar plates. After a prediffusion time of 15 minutes antibiotic disks were applied on the seeded agar surface by using a disc dispenser. The plates were incubated for 24 hours at 30ºC. After incubation, the diameter of the zone of inhibition was measured and compared with the zone interpretive chart and classified as resistant and sensitive. Percentage multiple antibiotic resistance (% MAR) of bacterial isolate was also carried out by using standard formula and a value greater than 20 indicated that the isolate was MAR.

Statistical Analysis

The data were analyzed by two-factor analysis of variance using the statistical tool package of Microsoft Office Excel 2007 software. Wherever the treatments were found to be significant, least significant

TABLE **2**

Bacteriological Parameters of Harvested and Stored Rainwater for Individual Household and Community Use

Sample #	Source of Water	TVC[b] (CFU/mL)	FC[b] (CFU/100 mL)	FS[b] (CFU/100 mL)	FC/FS Ratio	Presence of *E. coli*
1	RI[a]-1	2.05×10^3	1.10×10^3	4.60×10^2	0.24	+
2	RI-2	2.89×10^3	4.60×10^2	1.10×10^3	0.42	+
3	RI-3	5.90×10^3	9.30×10^1	2.40×10^2	0.39	–
4	RI-4	8.40×10^4	2.40×10^2	0.00	-	–
5	RI-5	1.74×10^4	0.00	2.30×10^1	0.00	–
6	RI-6	1.28×10^4	0.00	1.50×10^2	0.00	–
7	RI-7	2.88×10^4	0.40×10^1	1.50×10^2	0.03	–
8	RI-8	1.54×10^4	0.00	4.30×10^1	0.00	–
9	RI-9	8.20×10^3	0.00	4.60×10^2	0.00	–
10	RC[a]-1	2.51×10^3	4.60×10^2	4.60×10^2	1.00	–
11	RC-2	5.15×10^2	2.40×10^2	9.30×10^1	2.58	+
12	RC-3	4.93×10^3	2.10×10^1	4.30×10^1	0.49	+
13	RC-4	2.44×10^4	1.50×10^3	4.60×10^2	3.26	+
14	RC-5	1.74×10^4	4.30×10^1	7.50×10^1	0.57	+
15	RC-6	4.90×10^2	0.00	3.90×10^1	0.00	–
16	RC-7	3.00×10^2	0.00	0.40×10^1	0.00	–
17	RC-8	4.40×10^4	0.40×10^1	9.30×10^1	0.04	–
18	UC[a]-1	2.28×10^4	1.50×10^2	4.60×10^2	0.33	+
19	UC-2	2.04×10^4	2.10×10^3	2.40×10^2	8.75	–
20	UC-3	1.74×10^3	0.70×10^1	0.40×10^1	1.75	+
21	UC-4	5.80×10^3	0.00	0.70×10^1	0.00	–
22	UC-5	3.20×10^4	4.60×10^2	2.40×10^2	1.92	+
23	UC-6	1.31×10^4	0.00	2.40×10^2	0.00	–
24	UC-7	5.52×10^4	0.90×10^1	2.10×10^2	0.04	+
25	UC-8	2.48×10^4	0.00	2.40×10^2	0.00	–

[a]RI = rural individual household use; RC = rural community use; UC = urban community use.
[b]TVC = total viable count; FC = fecal coliform; FS = fecal streptococci.

were calculated and significant treatments were identified. Whenever necessary the results were presented as the average and standard deviation.

Results

Physicochemical Parameters and Nutrient Concentration of Harvested Rainwater

Mean values of the different physicochemical parameters of the rooftop-collected and stored rainwater for use in rural household, rural community, and urban community use are presented in Table 1. All the physicochemical parameters except pH were within the guidelines for these parameters. pH of the harvested and stored rainwater samples was highly alkaline in nature with a pH up to 11.32 in the rainwater samples from one of the tanks. Our study also showed the DO content of all the rooftop-harvested rainwater samples analyzed were in the range of 3.25–4.87 mg/L. Result also showed significantly higher concentration of nitrate ($p < .001$) than nitrite and phosphate, though no significant difference existed between samples.

Bacteriological Quality of the Rooftop Harvested and Stored Rainwater

The bacteriological quality of harvested rainwater is presented in Table 2. Total heterotrophic bacterial count ranged from 3.00×10^2 to 8.40×10^4 CFU/mL. While FC load ranged between zero to 2.10×10^3, FS load was from zero to 1.10×10^3 cells/100 mL. Our study revealed that 64% of harvested rainwater samples had FC and 96% of harvested rainwater samples had FS. The FC/FS ratio was less than 0.7 in 76% of the water samples indicating pollution from nonhuman sources. While *E. coli* was isolated from two of the nine rooftop-harvested rainwater samples for household use, water from 50% (4/8) of the storage tanks for community use in rural and urban areas showed the presence of this organism.

A total of 100 bacterial cultures from the harvested rainwater were characterized up to generic level and the percentage occurrence of different genera of bacteria in the rooftop-harvested rainwater is presented in Table 3. The harvested rainwater for urban community use had more diverse bacterial flora when compared to that of rural areas. Nine genera of bacteria were isolated with various frequencies. *Alcaligenes* and members of the family Enterobacteriaceae were isolated from all the samples while *Moraxella* and *Micrococcus* were isolated from only one sample. *Bacillus* was the predominant genera from rural household sample, while Enterobacteriaceae and *Alcaligenes* formed the dominant genera in water for rural community use and urban community use, respectively.

Antibiotic Resistance of the Bacterial Isolates From Harvested and Stored Rainwater

Antibiotic resistance among the heterotrophic bacteria from the harvested and stored rainwater is presented in Figure 1. Overall antibiotic resistance was higher among isolates from urban samples; the only exceptions were erythromycin and tetracycline. Resistance to antibiotics such as ampicillin, erythromycin, penicillin, and vancomycin was frequent among the bacterial isolates from harvested rainwater. More than 70% of bacterial isolates were sensitive, however, to amikacin, chloramphenicol, ciprofloxacin, and gentamicin. Percentage MAR and resistance profiles (data not shown) of viable bacterial isolates from harvested rainwater showed that 73% of bacteria had % MAR greater than 20. The highest % MAR was shown by *Alcaligenes* (83.3%) followed by *Bacillus* and Enterobacteriaceae (75%).

Discussion

Physicochemical Parameters and Nutrient Concentration of Harvested Rainwater

Except for pH, other parameters were within the drinking water quality guidelines and were in agreement with the observations of other researchers from different parts of the world (Chang, McBroom, & Beasley, 2004; Radaideh, Al-Zboon, Al-Harahsheh, & Al-Adamat, 2009; Simmons, Hope, Lewis, Whitmore, & Gao, 2001). The pH of rainwater usually ranges from 4.5 to 6.5 but increases slightly after falling on the roof and during storage in tanks (Meera & Mansoor, 2006). In our observations, the pH of the freshly collected rainwater was 6.01. The alkaline nature of the stored rainwater is most likely due to the insufficient curing of the storage tanks before usage. The storage tanks were made of ferrocement, which have been reported to cause alkalinity of stored rainwater (Handia, 2005; Simmons et al., 2001). The tanks from which the samples were collected during our study were newly constructed because the rainwater harvesting started only recently in the study area. It is argued, however, that the pH value declines with age of tank and period of storage.

DO is the most important parameter in potable water systems for its effect on other chemicals in the water as it oxidizes both organic and inorganic compounds and alters their chemical and physical states (Nduka, Orisakwe, & Ezenweke, 2008). DO levels of stored rainwater in urban and rural areas were above 3 mg/L. Relatively low DO may be due to the lack of replenishment of oxygen from atmospheric mixing/photosynthesis (that are possible in well water) as well as due to the consumption of oxygen by the microbial community for oxidation of organic material present, if any. DO of water is dependent on temperature, turbulence at the surface, surface area exposed to the atmosphere, atmospheric pressure, and amount of oxygen in the surrounding air, which are not favorable for stored water in the harvesting tanks.

TABLE 3

Percentage Occurrence of Different Genera of Heterotrophic Bacteria in the Rooftop-Collected Rainwater for Individual Household and Community Use

Genera	% Occurrence in Rooftop-Collected Rainwater		
	RI[a] (*n* = 24)	RC[a] (*n* = 32)	UC[a] (*n* = 44)
Gram negative			
Acinetobacter	8.33	–	11.36
Alcaligenes	8.33	21.88	47.72
Enterobacteriaceae	33.34	65.63	15.91
Moraxella	–	–	2.27
Pseudomonas	–	3.12	6.82
Vibrio	–	6.25	4.55
Gram positive			
Bacillus	50.00	3.12	–
Micrococcus	–	–	6.82
Unidentified	–	–	4.55
Total	100.00	100.00	100.00

[a]RI = rural individual household use; RC = rural community use; UC = urban community use.

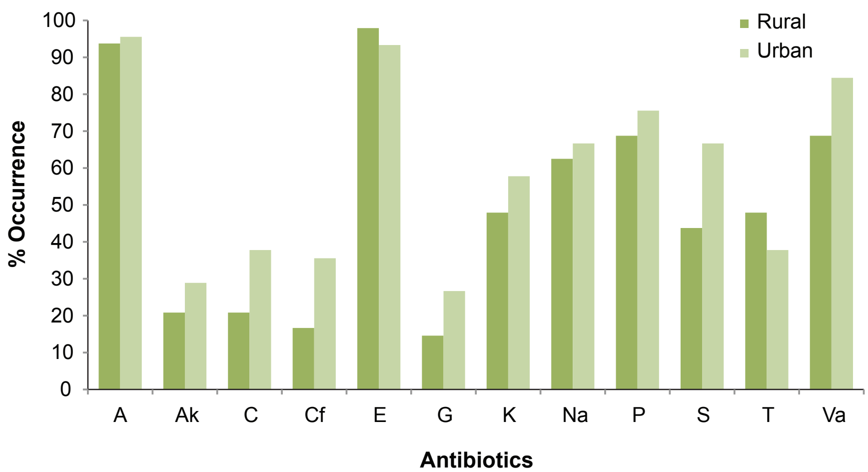
Wide variations in the concentrations of major ions like calcium, magnesium, sodium, potassium, chlorides, sulfates, and nitrates due to differences in roofing material and its treatment, orientation and slope of roof, air quality of region, characteristics of precipitation, etc., were reported by various researchers (Chang et al., 2004; Forster, 1996; Wu, Huizhen, Junqi, Hong, & Guanghui, 2001). Chloride ions are essential for life and in small concentrations they are not harmful to humans in drinking water. WHO (2004) permitted up to 200 mg/L chloride in potable water. Unlike well water, rainwater is adversely affected by local air pollution and debris in the rainwater catchment and conveyance areas. The result of our study indirectly points to good ambient air quality in the study area. While rainwater is considered pure, a large number of human-made atmospheric pollutants exist such as sulfur dioxide, nitrogen oxides, and various hydrocarbons, which together are the principal causes of acid rain. Such water can be unsafe to drink, especially in areas of heavy pollution such as industrialized urban areas. The nitrite and nitrate concentrations of the rainwater samples of our study were less than those reported by Radaideh and co-authors (2009) from the harvested rainwater from different regions.

Bacteriological Quality of the Rainwater

The TVC load of the samples analyzed in our study were much lower than those reported in earlier studies (Simmons et al., 2001; Uba & Aghogho, 2000). FC and FS load were also relatively lower, though *E. coli* was isolated from nearly 50% of samples for community use and 20% samples for household use. Previous studies (Simmons et al., 2001) reported that 56% of domestic roof-collected rainwater supplies exceeded the microbiological criteria of <1 FC/100 mL and recorded the presence of total coliforms, FC, and *E. coli* in a considerable number of samples (Radaideh et al., 2009). In Kerala, around 40% of the communicable diseases have been attributed to waterborne infections and in some of the coastal villages of Kerala diarrheal disease was ranked as the second most important cause of illness (Remani, 2004). Previous reports on microbiological quality of rainwater (Handia, 2005; Radaideh et al., 2009) revealed that rainwater harvesting systems do not often meet the microbiological quality standards for drinking water. In our study FC/FS ratios were worked out to track the source of fecal contamination. The major limitation of the FC/FS ratio is the difficulty

FIGURE **1**

Antibiotic Resistance Among the Heterotrophic Bacteria From Harvested Rainwater for Rural and Urban Use



A = ampicillin; Ak = amikacin; C = chloramphenicol; Cf = ciprofloxacin; E = erythromycin; G = gentamicin; K = kanamycin; Na = nalidixic acid; P = penicillin G; S = streptomycin; T = tetracycline; Va = vancomycin.

to use it effectively when mixed pollution sources are present. Since the rooftops are not protected from the entry of mongooses and birds the possibility of mixed sources of contamination is present. The results of our analysis, however, revealed nonhuman sources of fecal contamination. Though a difference of opinion exists on this method to identify the source, the results agree with previous observations (Appan, 1997).

Since the earliest microbiological investigations of drinking water quality, the detection of fecal indicator bacteria in drinking water has been used as a means of predicting the possible presence of pathogenic bacteria (WHO, 2004). Tank rainwater is usually harvested from a roof catchment area that is open to the environment and can be accessed by birds, insects, and animals. Fecal droppings from birds, lizards, mice, rats, and possums that can access the roof catchment may contain pathogenic microorganisms that are harmful to health when ingested. Pacific Islands Applied Geo-Science Commission's report (Mosley, 2005) had recommended several measures such as trimming of tree branches overhanging the roof, prevention of entry of mongooses to the rooftop, and regular cleaning as these may act as potential sources of entry of pathogenic bacteria into the rooftop-collected rainwater.

The results of our study revealed relatively better quality of rooftop-collected rainwater in the study area when compared to reports from other developing countries (Mbaka, 2004; Sazakli, Alexopoulos, & Leotsinidis, 2007). The FC and FS values of the harvested rainwater samples analyzed ranged between zero to 2.10 x 10^3 and zero to 1.10 x 10^3 CFU/100 mL, respectively, which was comparable to observations made by Fewtrell and Kay (2007). Detection of specific pathogens was also reported in harvested rainwater (Albrechtsen, 2002; Simmons et al., 2001; Uba & Aghogho, 2000). Since the FS and FC loads were comparatively low, we did not analyze the samples for specific pathogens.

Antibiotic Resistance of the Bacterial Isolates

Our study revealed that most bacterial isolates from harvested rainwater were resistant to ampicillin, erythromycin, penicillin, and vancomycin. Specific reports on drug resistance among the bacterial isolates from harvested rainwater are not available for comparison, though presence of drug-resistant bacteria in bottled mineral water/drinking water were available (Zeenat, Hatha, Viola, & Vipra, 2009). Ours may be the first report of drug-resistant bacteria in rooftop-harvested rainwater. MAR indexing of the isolates and elucidation of the resistance patterns (results not shown) revealed that 73% of the isolates were MAR in our study. Apart from variations among different genera, wide variations in the MAR index and resistance patterns were also noticed within the different strains of the same genera, indicating the diversity of the strains. One interesting observation was the higher frequency of resistance among bacterial isolates from urban area samples. Also resistance to most antibiotics was relatively high when compared with bacterial isolates from polluted household water, especially to chloramphenicol, ciprofloxacin, and streptomycin. This is an interesting observation, as we feel most of the urban flora might have originated from humans as a result of spillage of oral microflora or by sneezing or cough. These antibiotics are specific to human use and the resistant forms may be of human origin. The occurrence of MAR bacteria in the environment is certainly a well-known phenomenon. Many investigators believe that these drug-resistant organisms have become more common recently due to the extensive use of antibiotics in medicine and agriculture throughout the world.

The situation in India is more serious where the misuse of antibiotics is prevalent. Antibiotics are readily available over the counter without any prescription, and the general public is not aware of the consequences of improper and frequent use of antibiotics. This concern is particularly relevant in light of the discovery that resistance characteristics can be transferred to nonresistant recipient cells via R-factor plasmid vectors (Mitsuhashi, 1977). The microflora that could be present in the rainwater are most likely the transient aerobic microflora, which could be mostly of human origin in a highly populated country like India. The sampling locations in our study were located among dense rural and urban community and most of the drug-resistant forms that are encountered might have possibly originated from a human source.

Conclusion

The rooftop-collected water in the study area for individual household and community use was found to be within the WHO chemical quality for drinking except for the pH and lower DO levels. The bacteriological quality did not often meet the standards prescribed for drinking water, however, and was found to contain FS and FC in several samples. The multidrug-resistant nature of the bacteria from the samples is a matter of concern. The gastroenteric pathogen *E. coli* was detected in 40% of the harvested rainwater. *Alcaligenes*, Enterobacteriaceae, *Pseudomonas*, *Vibrio,* etc. were also isolated from the rainwater harvesting tanks. Our study suggests that some form of low-cost treatment of harvested rainwater such as SODIS is necessary before it can be used as a source of drinking water. To reduce the probability of coliform in rainwater harvesting systems it is recommended to keep the rain catchment area clean and free of debris and should be cleared of overhanging branches of trees.

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Prepublished online October 2013,
National Environmental Health Association.

Concentration Gradient Patterns of Traffic and Non-Traffic-Generated Fine and Coarse Aerosol Particles

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Abstract **The research project described in this article was undertaken to establish baseline information for a Health Impact Assessment (HIA) project of Interstate 75 road construction in Cincinnati, Ohio. The objective of the authors' study was to evaluate the concentrations of elemental and organic carbon (EC and OC), as well as characterize particle number concentrations using devices that measure the fine fraction in the range of 0.02–1 µm and the coarse fraction up to 20 µm. The measurements were conducted at two sites located in the proximity of an interstate highway (at 124 and 277 m) as well as at a remote control site (at >2000 m from any interstate highway). Samples were collected for 24 hours over 12 days in each season (i.e., summer, fall, and winter). Wind data were obtained from the area weather station. Data were analyzed using mixed linear models. Significant increases in concentrations of EC, OC, and fine particles as well as in EC/OC ratios were observed with decreased distance to the highway; this difference was more pronounced in the fall. These results suggest that residents and workers in areas near high-traffic highways may be exposed to elevated levels of airborne fine particles. The results can be used as a baseline for future HIAs of road construction in the area.**

Introduction

The Health Impact Assessment (HIA) is a tool that provides decision makers at the city, county, state, and federal levels with information on how a policy will potentially affect the health of the population. The "recommendations" of HIA projects are evidence based and geared towards maximizing positive health impacts by removing or minimizing the negative health impacts on the population (Taylor & Quigley, 2002). Health effects occupy the focal point in decision-making policies outside of the health sector.

Highway traffic in urban areas is a significant contributor to the total airborne particulate concentration. Several studies have suggested that exposure to traffic-generated aerosols exacerbates asthma in patients living near highways (Holguin, 2008). Children are particularly susceptible to these aerosols due to their developing respiratory system and could be adversely affected if they reside close to highways (Gauderman et al., 2007). Even short-term exposure to traffic-related particles has been shown to reduce lung function in atopic schoolchildren (Barraza-Villarreal et al., 2011).

$PM_{2.5}$ is defined as airborne particulate matter with an aerodynamic diameter less than or equal to 2.5 µm. It is most often produced via combustion (U.S. Environmental Protection Agency [U.S. EPA], 2008). Due to their small size, these particles penetrate deep into the respiratory tract, creating the potential for adverse health effects (U.S. EPA, 2008). $PM_{2.5}$ mass concentrations do not vary greatly with differing distances from highways (Martuzevicius et al., 2004; Roorda-Knape et al., 1999) and are only slightly affected by traffic density (Martuzevicius et al., 2005). A more clear effect of traffic sources has been observed for fine particles (<1 µm in diameter) and ultrafine particles (<0.1 µm) (Reponen et al., 2003; Zhu et al., 2002, 2009).

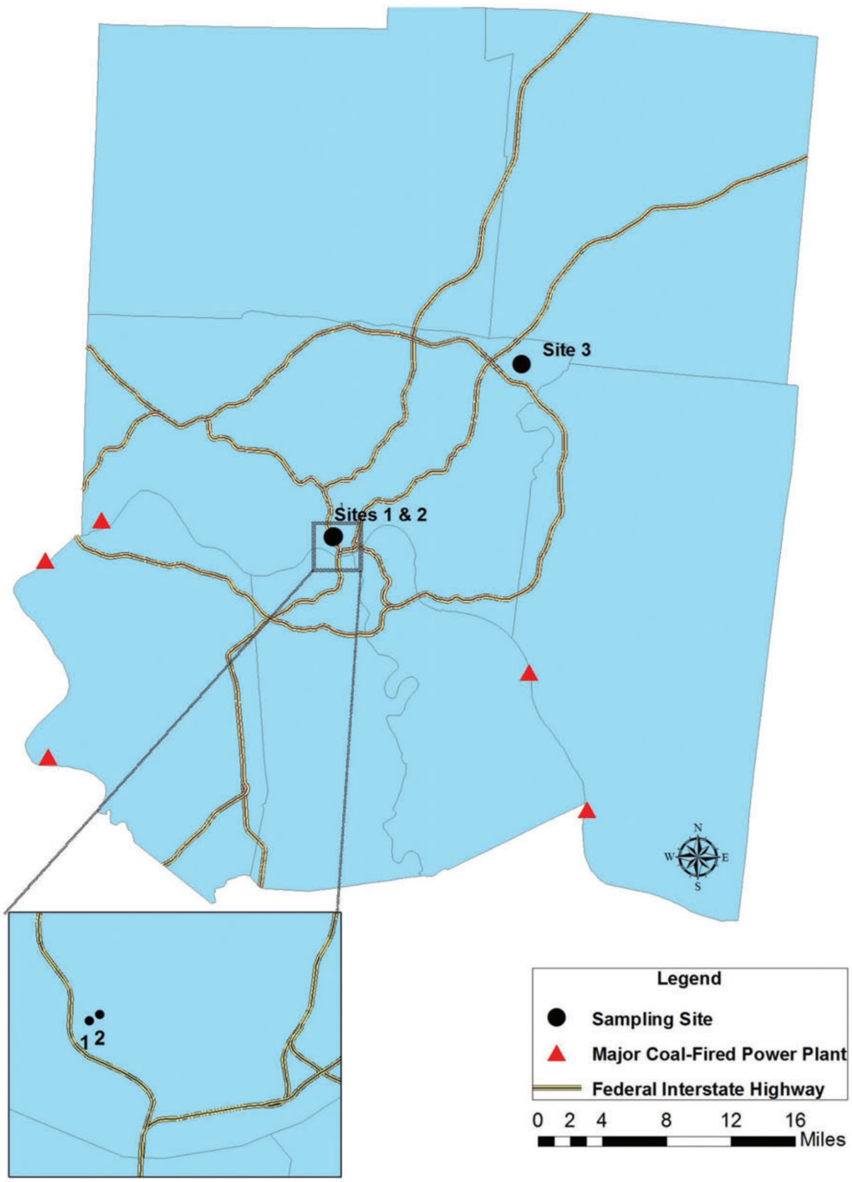
bon (EC/OC) provides an estimate of the overall percentage of the total carbon (EC + OC) that can be attributed to combustion of diesel. Where traffic exhaust is the primary source of diesel and combustion exhaust, this ratio is used to indicate the fraction of the total carbon attributable to diesel consuming vehicles (Maykut, Lewtas, Kim, & Larson, 2003).

Interstate highways, the major traffic arteries in the U.S., undergo various improvements, especially in major metropolitan areas known for traffic congestion. Widening highways by adding lanes allows for higher traffic volume, which may increase the traffic aerosol emission. An improvement of Interstate 75 (I-75), a major north-south transportation corridor, is currently in the planning phase in the greater Cincinnati area. This will include adding one lane in both directions, which may result in potential health implications for residents in the construction area—a mostly low-income population. In this light, an HIA of the construction site was initiated to obtain baseline air quality information, followed by assessment of air quality during the construction and after its completion.

The distance traveled by highway-generated airborne particles of different sizes is unknown for highways in the greater Cincinnati area. Furthermore, the effect of future road construction on the local air quality is not clear. Our case study investigated particle number and mass concentrations of fine and coarse particles as well as EC, OC, and $PM_{2.5}$ at different distances from an interstate highway in order to create a baseline data set for future HIAs.

FIGURE 1

Location of Sampling Sites in Relation to Highways and Major Coal-Fired Power Plants



Kim and co-authors (2004) reported that the concentrations of black carbon (organic carbon [OC] and elemental carbon [EC]) were higher in areas within 300 m of highways compared to background. While OC is produced by all combustion sources, EC is primarily generated by traffic sources, particularly diesel-burning vehicles (Birch & Cary, 1996). As such, EC is frequently used as a surrogate for traffic-generated aerosols (Holguin, 2008; Ryan et al., 2009). It is reported that EC concentrations are greater in areas near highways and increase with increased truck traffic (Kinney, Aggarwal, Northridge, Janssen, & Shepard, 2000; Lena et al., 2002; Martuzevicius et al., 2004). The ratio of elemental carbon to organic car-

Methods

Site Selection

Three sampling sites were selected based on direction and distance from a high-traffic highway in Cincinnati, Ohio, and are further referred to as site 1, site 2, and site 3 (Figure 1). Sites 1 and 2 were located at 124 m and 277 m, respectively, from I-75 (site 2 was initially chosen at 283 m but moved 6 m closer after the first sampling period). Site 3 served as a background station located at >2,000 m from any interstate highway in the metropolitan area. The sites were selected northeast and downwind of the closest highway (based on the predominant wind direction in Cincinnati [Martuzevicius et al., 2004]) and far away from major coal-fired power plants. Stations 1 and 2 were placed on the roofs of buildings (at the heights of 7.6 and 19.5 m, respectively) and the background station 3 was placed on the ground. The traffic volume on the highway nearby sites 1 and 2 was 142,500 cars and 19,000 trucks per day. The respective numbers on the highway nearest to site 3 were 82,400 and 9,100.

Aerosol Sample Collection

Ambient air sampling was conducted during three seasons (summer and fall of 2010 and winter of 2011). A total of four Harvard $PM_{2.5}$ Impactors (MS&T area sampler) were utilized on the three sites. Twelve 24-hour samples were collected at a flow rate of 20 L/min during each season. The sampling was carried out on days with limited or no rainfall. At each of the three sites, particulate matter was collected onto 37-mm quartz filters that were analyzed for EC and OC concentrations using evolved gas analysis by a thermal optical analyzer as performed by a commercial laboratory. An additional (the fourth) $PM_{2.5}$ sampler was deployed at site 1 to collect samples onto 37-mm Teflon filters that were analyzed gravimetrically.

Each station was equipped with two real-time particle measurement devices: a P-Trak condensation nuclei particle counter and an ARTI optical particle counter. The P-Trak measures the total number concentration of airborne particles in the size range of 0.02–1 μm (fine particles), whereas the ARTI measures the particle number concentration size selectively in the size range of 0.7–20 μm (mostly coarse particles). The real-time data generated by both instruments were recorded as three-minute averages. The instruments were operated from 8:00 a.m. to 6:00 p.m. on each day when the filter samples were collected; however, in some cases only a portion of this 10-hour window was found useful (the limitation was due to technical problems such as rapid evaporation of isopropanol in a condensation nuclei counter, especially in summer, and malfunctioning of pumps during long-term sampling). As a result, the real-time data obtained from 9:30 a.m. to 12:30 p.m. were utilized for analysis because these measurements were consistent at all three sites and analysis of variance demonstrated that the average concentrations calculated from the data collected from 9:30 a.m. to 12:30 p.m. did not differ from the overall average values determined for the entire 10-hour period

FIGURE **2**

Calculation of Wind Index

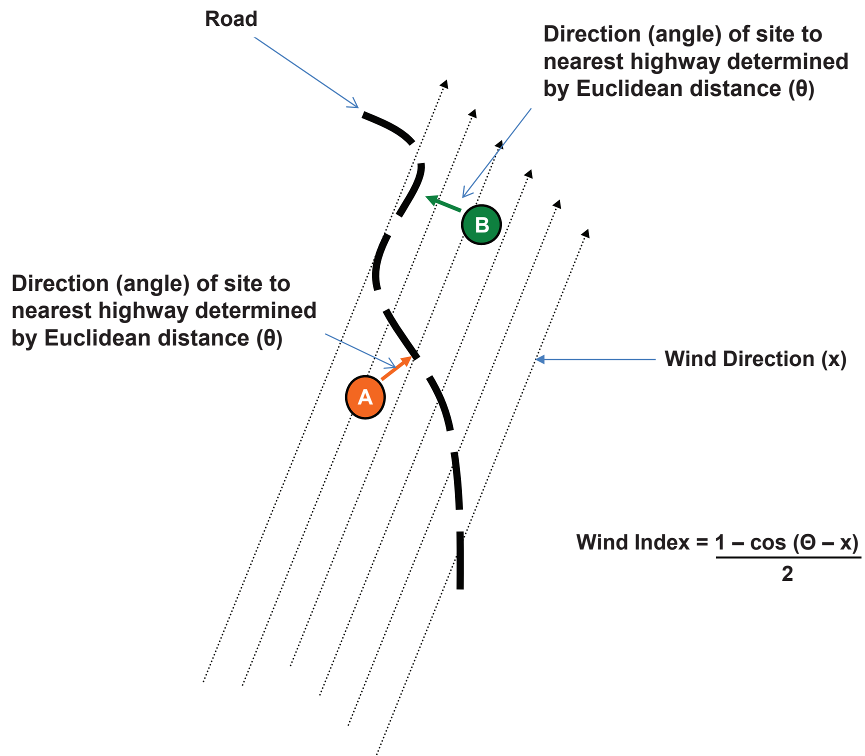


TABLE **1**

Geometric Means and 95% Confidence Intervals

Measured Parameter	Geometric Mean	95% Confidence Interval
$PM_{2.5}$[a] (μg/m³)	15.4	13.3–17.8
EC[a] (μg/m³)	0.53	0.46–0.61
OC[a] (μg/m³)	3.53	3.28–3.81
EC/OC	0.15	0.14–0.17
Number concentration of fine particles measured by P-Trak (1/cm³)	12628	10579–15074
Number concentration of course particles measured by ARTI (1/cm³)	1267	943–1702

[a]$PM_{2.5}$ = particulate matter ≤2.5 μm; EC = elemental carbon; OC = organic carbon.

TABLE 2

Arithmetic Means (Standard Deviations) of Wind Speed and Wind Index Per Season

Season	Wind Speed, mph	Wind Index			*p*-Value[a]
		Site 1	Site 2	Site 3	
Summer	5.82 (1.89)	0.56 (0.39)	0.59 (0.34)	0.33 (0.20)	.006
Fall	8.28 (3.23)	0.50 (0.39)	0.50 (0.38)	0.54 (0.26)	.872
Winter	9.42 (3.58)	0.66 (0.35)	0.65 (0.35)	0.52 (0.35)	.152
p-Value for the difference between seasons	<.001	.695	.636	.426	

[a]For the difference in wind index between sites.

TABLE 3

Comparison of Particle Concentrations Between Sites by Analysis of Variance Mixed Model for Fixed Effects

Site Comparison	Differences of Least Squares Means of Natural Log Transformed Concentration Values (*p*-Value)				
	EC[a]	OC[a]	EC/OC	Fine Particle Concentration (P-Trak)	Coarse Particle Concentration (ARTI)
1 vs. 2	**0.278 (<.001)**	**0.197 (<.001)**	**0.085 (.016)**	**0.370 (.007)**	-0.032 (.866)
1 vs. 3	**1.052 (<.001)**	**0.502 (<.001)**	**0.559 (<.001)**	**0.971 (<.001)**	-0.191 (.677)
2 vs. 3	**0.774 (<.001)**	**0.306 (<.001)**	**0.474 (<.001)**	**0.601 (.002)**	-0.159 (.712)

Note. Significant differences between pair-wise comparisons are bolded (α = .05). Model is adjusted for wind speed and wind index.
[a]EC = elemental carbon; OC = organic carbon.

(p > .05). Average concentrations were used instead of hourly data because cumulative exposure values are more relevant for the future HIA.

Data Analysis

The statistical modeling was performed for EC, OC, EC/OC ratio, $PM_{2.5}$ mass, and number concentrations of fine and coarse particles. Data were found to be normally distributed when log transformed. Geometric means and 95% confidence intervals were calculated for all particle concentrations. The arithmetic mean was also calculated for $PM_{2.5}$, so that data could be compared with the National Ambient Air Quality Standard (NAAQS) (U.S. EPA, 2011).

Wind speed and direction were obtained from data gathered at the nearest National Weather Service sampling location, 8 miles from sites 1 and 2 and 24 miles from Site 3. Daily averages of the available hourly values were determined for each 24-hour filter collection period, from 8:00 a.m. to 8:00 a.m. the following day. Additionally, averages of the hourly values between 9:00 a.m. and 1:00 p.m. were determined to relate to real-time samples. A wind index was calculated for each site as follows:

$$Wind\ Index = \frac{1 - cos\ (\theta - x)}{2}$$

where θ = the angle (θ) of the site to the nearest highway and x = wind direction (Figure 2) (Ryan et al., 2008).

The wind index is a rescaling of the difference in the angle to nearest major traffic source and predominant wind direction to a scale of zero to one. The wind index is a continuous variable; sites directly upwind of the nearest traffic source had a wind index equal to zero, sites directly downwind of the nearest traffic source had a wind index equal to one, and sites perpendicular to the wind direction had an index of 0.5.

For all analyses, sample days missing data from any three sites were excluded. Each data set was then analyzed for spatial and seasonal variation using a linear mixed model (SAS v. 9.2 software), adjusting for wind speed and calculated wind index. The mixed model was also used to compare wind speeds and wind indexes between the sampling seasons. A p-value < .05 was considered statistically significant.

Results

The geometric means (GM) and 95% confidence intervals for $PM_{2.5}$, EC, OC, EC/OC and the number concentrations of fine and coarse particles are presented in Table 1. $PM_{2.5}$ concentrations varied from 5.4 to 34.4 µg/m^3 having a geometric mean of 15.4 µg/m^3 and an overall arithmetic mean of 17.0 µg/m^3. Daily average concentrations of EC varied from 0.06 to 2.91 µg/m^3 (GM = 0.53 µg/m^3) whereas OC concentrations were higher, varying from 0.73 to 10.35 µg/m^3 (GM = 3.53 µg/m^3). The EC/OC ratios varied from 0.04 to 0.48 (GM = 0.15). The number concentrations of fine particles, as measured with the P-Trak, ranged from 2,991 to 42,749/cm^3 (GM = 12,628/cm^3). Considerably lower particle number concentrations were measured with the ARTI for large particles: 268–8,872/cm^3 (GM = 1,267/cm^3).

Table 2 presents wind speeds and indexes. On average, the sampling sites were neither up nor downwind of the source based upon the wind index and were not significantly different between seasons. A significant difference in the wind index between sites, however, was observed in the summer (p = .006, Table 2). Therefore, statistical analyses presented in Table 3 and 4 on the concentration data were adjusted for wind data. Wind speed was strongest in the winter and weakest in the summer (p < .001). Given that the original wind data were obtained from a single weather station, the same wind speed was used for each sampling site.

The geometric means of the EC and OC concentrations and EC/OC ratios at each site are presented in Figure 3 and the results of analyses in Tables 3 and 4. EC concentrations decreased with increasing distance from the nearest interstate highway and the differences were significant between sites (p < .001; Table 3). As expected, site 1 had the highest EC concentration and site 3 had the lowest EC concentration for all seasons. The EC concentrations were lowest in winter and highest in fall (p = .004; Table 4). Similar to EC, OC concentrations decreased consistently in all sampling seasons with increasing distance from the nearest interstate highway (Table 3). In all seasons, site 1 exhibited the highest OC concentration and site 3 had the lowest OC concentration (p < .001). No significant differences were found in OC concentrations between the different seasons (Table 4). EC/OC ratio was highest at site 1 and lowest at site 3 (Table 3). The highest EC/OC values were identified in the fall. The seasonal differences were significant between fall and summer (p = .031; Table 4) and between fall and winter (p < .001).

Sampling for $PM_{2.5}$ was only conducted at site 1. $PM_{2.5}$ concentration was significantly lower in the fall than in the winter (p = .027; Table 4).

The geometric means for number concentrations of fine and coarse particles measured at each of the three sites during the three sampling seasons are shown in Figure 4. Some lack of consistency in performance of the real-time particle monitoring instruments resulted in varied sample numbers between seasons. The concentrations of fine particles were significantly different between sites. The particle number concentration at site 1 was greater than at site 2 (p = .007) and site 3 (p < .001), and the concentration at site 2 was greater than at site 3 (p = .002) (Table 3). Fine particle concentration was lower in the sum-

FIGURE **3**

Geometric Means and 95% Confidence Intervals for the Concentrations of Elemental Carbon (EC) and Organic Carbon (OC) and EC/OC Ratio (*N* = 12)

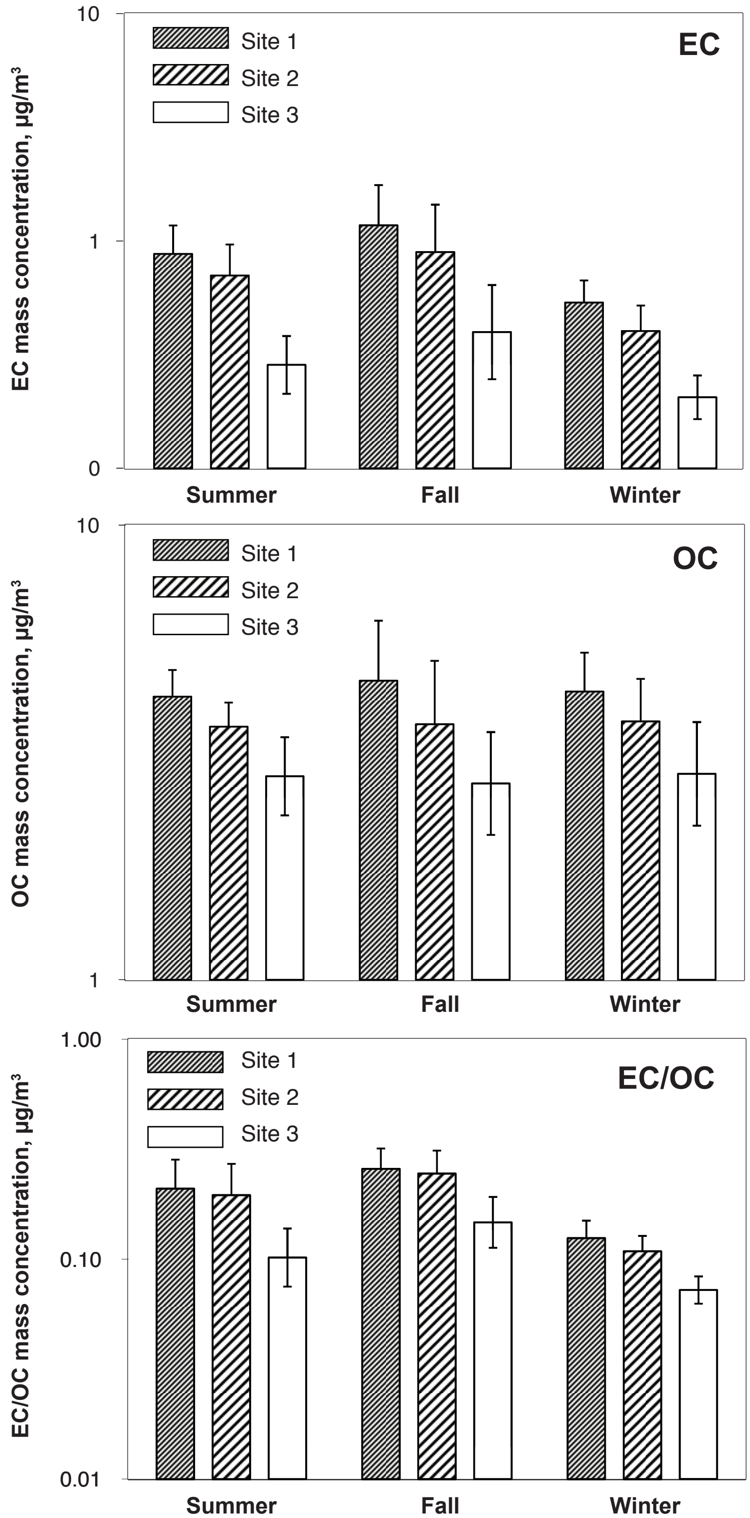


TABLE 4

Comparison of Particle Concentrations Between Seasons by Analysis of Variance Mixed Model for Fixed Effects

Seasons	Differences of Least Squares Means of Natural Log Transformed Concentration Values (*p*-Value)					
	$PM_{2.5}$[a]	EC[a]	OC[a]	EC/OC	Fine Particle Concentration (P-Trak)	Coarse Particle Concentration (ARTI)
Summer vs. fall	0.250 (.352)	-0.473 (.056)	-0.084 (.841)	**-0.364 (.031)**	**-0.610 (.025)**	-0.404 (.811)
Summer vs. winter	-0.210 (.498)	0.168 (.696)	-0.161 (.558)	0.333 (.065)	-0.854 (.051)	-0.529 (.785)
Fall vs. winter	**-0.460 (.027)**	**0.641 (.004)**	-0.077 (.849)	**0.697 (<.001)**	-0.244 (.690)	-0.125 (.967)

Note. Significant differences between pair-wise comparisons are bolded (α = .05). Model is adjusted for wind speed and wind index.
[a]$PM_{2.5}$ = particulate matter ≤2.5 µm; EC = elemental carbon; OC = organic carbon.

mer than in the fall (p = .025; Table 4). The average concentrations of coarse particles were highest at site 3 in summer and winter, but this difference was not statistically significant (Table 3). Seasonal variation of large particles was not significant either (Table 4).

Discussion

We found that the concentrations of EC, OC, and the EC/OC ratio were significantly greater at locations nearest the highway, suggesting that traffic is a major contributor to these ambient aerosols. Our results support data reported by Kim and co-authors (2004), which revealed that the concentration of traffic-related air pollution decreases downwind from the highway.

A similar decreasing trend was observed for the number concentrations of fine particles, suggesting a concentration gradient also exists for fine particles with respect to distance. It should be noted that the sampling stations were located at different heights. Hitchins and co-authors (1999) have shown, however, that the sampling height did not affect the concentration of fine particles at distances of 80 and 210 m from the highway. While the measured particles are not necessarily all traffic related, our data suggest that the aerosol concentration in the size range of 0.02–1 µm is greater in areas near a highway with intense traffic. Investigations in other cities have indicated that number concentrations of fine particles decrease to the background level at a distance of about 300 m from highways (Zhu et al., 2002, 2009). Reponen and co-authors (2003) reported that the spatial variation between 400 m and 1600 m from a highway was not significant. Our study shows significant differences between sites 124 m and 277 m from the nearest source and significant differences between both of these sites versus the background site located more than 2,000 m away from highways. While the spatial variation reported in this study appears to be somewhat different from the one reported by Reponen and co-authors (2003), the number concentrations of fine particles, which ranged approximately from 1.5×10^4 to 2.0×10^4 1/cm^3, were similar.

The number concentration of coarse particles was higher at site 3 compared to sites 1 and 2, though this difference was not statistically significant. This may be attributable to the increase in landscaping activities taking place at site 3. During both seasons, lawn care companies were in the area surrounding site 3 up to five days a week. The location of this sampling site on the ground may have increased the contribution of local sources, which is a limitation of our study.

Limited information is available on the horizontal and vertical variation of coarse particles (Cheung et al., 2010, 2011) and therefore the impact of the differing sampling height is difficult to estimate. It is notable, however, that the effect of landscaping activities was not seen in OC concentrations. It was apparent only for the number concentrations of coarse particles measured with an optical particle counter. Our results are consistent with those presented by Pabkin and co-authors (2010), who concluded that traffic is the major source for both fine and coarse particles near highways, whereas natural sources such as windblown dust dominate in more rural areas. Future HIA studies should include chemical speciation of the coarse particle size fraction. This would allow more clear differentiation of the effects of traffic and road construction.

A clear seasonal variation was observed for most of the studied particle types. Concentrations of EC and fine particles as well as EC/OC ratio were highest in the fall. In contrast, concentrations of $PM_{2.5}$ were lowest in the fall. Martuzevicius and co-authors (2004) have reported similar seasonal variation suggesting that it is the result of coal-powered power plants being the primary $PM_{2.5}$ contributor in the greater Cincinnati area and increased energy usage during summer and winter. The fact that samples were only collected on days with limited or no rainfall and low wind speeds largely limits the influence of weather-related phenomena. The results indicate that the effect of traffic on the aerosol concentrations is greater in the fall than in the summer and winter, when other aerosol sources appear to be more dominant.

The highest measured concentration was close to the 24-hour fine particle threshold listed in the NAAQS of 35 µg/m^3 (U.S. EPA, 2011). Furthermore, the overall average of the 24-hour $PM_{2.5}$ samples (17.0 µg/m^3, n = 36) exceeded the annual $PM_{2.5}$ NAAQS of 15.0 µg/m^3.

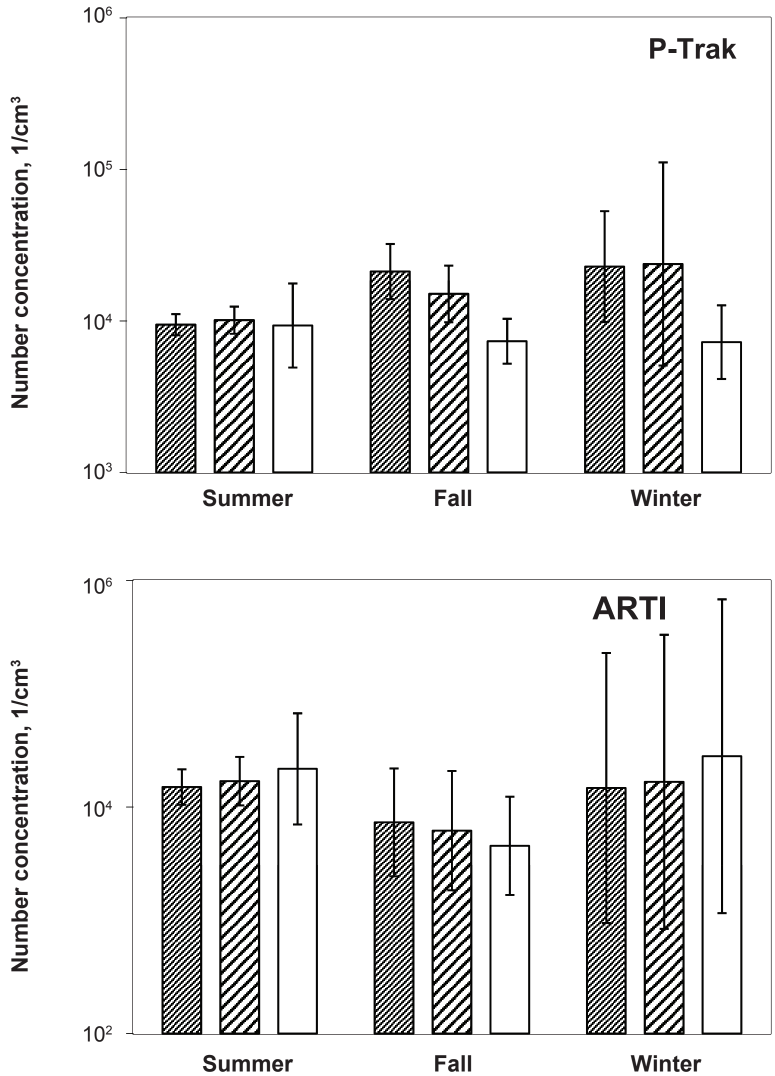
Our results suggest that residents and workers in areas near high-traffic highways may be at increased risk of experiencing negative effects from traffic-related aerosols. High background levels of $PM_{2.5}$ add to overall particle exposure. Future road construction will likely lead to increases in concentrations of fine and coarse airborne particles as a result of congested traffic (Keuken et al., 2010), changing traffic patterns, and the construction activity itself. It is also possible that the concentrations observed after construction may decrease due to more efficient traffic patterns. By contrast, the above positive outcome may be diminished if the increase in road space will increase the traffic density over time. To determine the true trend of air pollution associated with the highway improvement, it seems useful to conduct similar sets of measurements during and after construction.

The road construction/demolition can also indirectly affect the health of citizens (Wernham, 2011). New traffic patterns may increase the risk of traffic-related injuries. Furthermore, the roadway might unintentionally cut off an important walking route to and from a transit stop or local school, making it harder for adults and children to get enough exercise.

These are significant health concerns. It is estimated that health problems associated with our current transportation system, such as injuries, asthma, cardiovascular disease, and premature mortality may result in over $300 billion in additional costs every year. This amount includes accidents and medical expenses as well as lost wages and lost productivity (Wernham, 2011). One way to reduce the negative impacts of transportation is the HIA, which is a powerful tool being used worldwide to identify unintended health risks and unnecessary costs.

FIGURE **4**

Geometric Means and 95% Confidence Intervals for Fine and Coarse Particle Concentration



Measured with a P-Trak (summer: $n = 4$; fall: $n = 8$; winter: $n = 3$) for fine particle concentration and measured with an ARTI (summer: $n = 5$; fall: $n = 4$; winter: $n = 3$) for coarse particle concentration.

Conclusion

In summary, the concentrations of EC, OC, fine particles and the EC/OC ratio were significantly greater at locations nearest the highway, suggesting that traffic is a major contributor to these ambient aerosols. The concentrations of EC and number of fine particles as well as EC/OC ratio were highest in the fall, whereas the concentration of $PM_{2.5}$ was lowest in the fall; these findings suggest that the effect of traffic on the aerosol concentrations in this area is more pronounced in the fall.

Our study was undertaken to provide decision makers with a tool to assess the exposure and consequently the health impact of the future infrastructure improvement to I-75 in the greater Cincinnati area. The main outcome of our study is a baseline aerosol database to be used in a follow-up HIA evaluation.

Acknowledgements: Our study was partially funded by the National Institute for Occupational Safety and Health through the University of Cincinnati Education and Research Center grant #T42/OH008432 and the Cincinnati Childhood Allergy and Air Pollution Study through the National Institute of Environmental Health Sciences grant #R01 ES11170.

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Environmental Toxicity and Poor Cognitive Outcomes in Children and Adults

Prepublished online October 2013,
National Environmental Health Association.

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Abstract **Extensive literature has already documented the deleterious effects of heavy metal toxins on the human brain and nervous system. These toxins, however, represent only a fraction of the environmental hazards that may pose harm to cognitive ability in humans. Lead and mercury exposure, air pollution, and organic compounds all have the potential to damage brain functioning yet remain understudied. In order to provide comprehensive and effective public health and health care initiatives for prevention and treatment, we must first fully understand the potential risks, mechanisms of action, and outcomes surrounding exposure to these elements in the context of neurocognitive ability. This article provides a review of the negative effects on cognitive ability of these lesser-studied environmental toxins, with an emphasis on delineating effects observed in child versus adult populations. Possible differential effects across sociodemographic populations (e.g., urban versus rural residents; ethnic minorities) are discussed as important contributors to risk assessment and the development of prevention measures. The public health and clinical implications are significant and offer ample opportunities for clinicians and researchers to help combat this growing problem.**

Introduction

The effect of environmental toxins on the brain and nervous system has been studied and documented extensively. A majority of the existing research has focused on the detrimental effects of heavy metals, however, while it appears that exposure to heavy metals represents only a portion of the total environmental health risks. For example, the damaging effects of air pollution on the brain, and particularly on intelligence, are being increasingly viewed as a significant concern. Another important concern is exposure to organic chemical compounds (such as polychlorinated biphenyls and dioxins) with toxic effects. What is particularly troubling about these compounds is that they can accumulate in animals and therefore can be delivered to humans via food products (Tuomisto, Vartianen, & Tuomisto, 2011).

Pollution is a complex issue affecting underdeveloped as well as developed countries. Socioeconomic issues are also relevant; people living in urban or impoverished areas may face greater exposure. Combatting pollution is a unique challenge, as it can require significant participation and economic support across all levels of government, from local municipalities all the way up to national authorities. All of these factors make pollution a broad research subject. As a result, this article will focus on some of the more recent developments and concerns regarding a few noteworthy pollutants. First, some of the major findings of the effects of the heavy metals lead and mercury on cognitive function will be reviewed. Although lead and mercury have been studied previously and extensively as mentioned earlier, they were included in this article due to recent developments regarding exposure. For example, the acceptable blood lead levels (BLLs) in children have decreased steadily over time. Mercury has reemerged as a threat due to its persistence in the food supply, particularly seafood.

Second, this article will also explore the potential effects of air pollutants and organic chemical pollutants on cognition. Special attention is given to the effects of pollution on specific population groups as well as more universal effects throughout each section of the article. The discussion on air pollution was included in this article because it is being increasingly viewed as a potential problem and its mechanisms of action are still not completely understood. Discussion on the effects of organic compounds was also included in this article due to growing concerns about exposure through the food supply. Finally, public health and clinical implications for health care providers are discussed.

Lead and Mercury

Children

The potential health effects of high exposure to lead have been studied comprehensively. Over the past few decades a debate has persisted, however, about how much exposure is necessary to produce negative health effects, especially in children. Indeed, the Centers for Disease Control and Prevention (CDC) have steadily lowered the threshold for BLLs in children considered dangerous by 88%, from 60 µg/dL to 10 µg/dL over the last 40 years (CDC, 2006).

More recently, increasing data have also indicated that even children with blood lead concentrations <10 µg/dL are at significant risk for reduced cognitive development and functioning, including IQ deficits and poor academic performance (Bellinger, 2008; Jusko, 2008; Lanphear et al., 2005; Liu, Li, Wang, Yan, & Liu, 2013). In fact, no threshold has been determined for this effect. Accordingly, the CDC recently eliminated the terminology "level of concern." Rather, children with elevated blood lead concentrations will be identified using a reference value based on the 97.5th percentile of the National Health and Nutrition Examination Survey–generated blood lead concentration distribution in children aged 1–5 years old; currently, this value is 5 µg/dL (CDC, 2012). Prenatal exposure to lead via vertical transmission from mother to child, including at levels below 10 µg/dL, is also a concern and can have detrimental effects on cognitive function and IQ in infants and children (Jedrychowski et al., 2009).

Mercury is another heavy metal that poses a significant health threat to children. It is of significant concern that mercury (in the form of methylmercury) is found in fish and shellfish, which are readily available for consumption as part of a child's diet. In fact, the Food and Drug Administration (FDA) currently advises that young children, pregnant women, and nursing women avoid the four fish (shark, swordfish, king mackerel, and tilefish) that contain high levels of mercury (FDA, 2004). FDA also advises that these individuals consume no more than 12 ounces of fish and shellfish lower in mercury per week (FDA, 2004). These cautionary guidelines are due to growing evidence from the scientific community about the potential negative health effects of accumulating mercury via diet.

In a study of preschool children in Spain, elevated hair mercury levels were associated with a higher frequency of oily fish consumption as well as a delay in cognitive development (Freire et al., 2010). In a study on adolescents, however, higher hair mercury levels were associated with improved visual-spatial capabilities, which the authors posit may be due to general benefits of fish consumption (e.g., increased protein, increased omega-3 fatty acids) outweighing the risks of mercury accumulation (Torrente, Colomina, & Domingo, 2005). It is possible that mercury's impact is more pronounced when children are younger, when their developing neurocognitive systems are more vulnerable to harm. Some prenatal studies have distinguished between the effect of higher mercury levels and the effect of higher maternal fish consumption. In a study that examined mercury levels due to atmospheric exposure instead of exposure due to diet, higher mercury levels in young children were associated with decreased IQ (Trasande, Schecter, Haynes, & Landrigan, 2006). Despite the consensus that higher mercury levels are associated with negative effects on cognition (Oken et al., 2008), higher maternal fish consumption has been associated with a beneficial impact on cognition, even though increased fish consumption would suggest elevated mercury levels (Oken et al., 2008). This may be due to the variety of fish consumed, as different species of fish contain differing amounts of mercury, as discussed previously (FDA, 2004).

Adults

Lead also has negative effects on adults. Several studies have found an association between higher lead levels and declines in cognitive performance (Shih, Hu, Weisskopf, & Schwartz, 2007; Weuve et al., 2009), which may or may not be reversible (Winker, Ponocny-Seliger, Rudiger, & Barth, 2006). A concern also exists that low-level accumulation of lead over a long period of time, such as in the elderly, may also be harmful to cognitive performance (Stewart & Schwartz, 2007). Stewart and Schwartz even posit that the manifestation of "normal" cognitive decline due to age may be in fact due to an accumulation of neurotoxins, such as lead, over time. A concern is also that exposure to lead may later lead to the emergence of dementia disorders, particularly Alzheimer's disease (AD) (Basha & Reddy, 2010). Other researchers, however, have found no link between elevated lead levels and decreased cognition in adults (Nordberg, Winblad, Fratiglioni, & Basun, 2000).

Although evidence suggests an accumulation of low amounts of mercury appears to have a negative impact on children's cognition, less evidence exists to support a similar relationship in adults. A study of individuals occupationally exposed to mercury found that effects were small and difficult to detect on a case-by-case basis (Rohling & Demakis, 2006). Another study examined individuals who received mercury alloy amalgam-based dental implants and found no relation with cognition (Sundstrom, Bergdahl, Nyberg, Bergdahl, & Nilsson, 2010). Similarly, the link between elevated mercury and cognition in older adults is unconvincing (Weil et al., 2005). The same is true for the connection between mercury exposure and AD; associations between the two exist but they have not been found to be significant (Mutter, Naumann, Schneider, & Walach, 2007). It is possible that individuals with higher mercury levels may be benefitting from higher fish consumption, which could mitigate mercury's harmful effects. Indeed, the potential interaction between mercury and selenium has been under investigation, where the increased levels of selenium found in fish may mitigate the effects of mercury (Mozaffarian & Rimm, 2006).

Potential Mechanisms of Action

The mechanism for the toxic health effects seen from lead and mercury exposure is thought to primarily involve the brain. Lead is believed to raise oxidative stress, stimulate apoptosis of neurons, and affect neurotransmitter release (Shih et al., 2007). Lead can circulate in the blood or accumulate in bones, where it can escape into the blood during periods of bone turnover (Shih et al., 2007). When in the blood, lead is capable of substituting for calcium, allowing it to cross the blood-brain barrier and disrupt other calcium-based properties (Shih et al., 2007). It may be through these mechanisms that lead can directly affect cognitive performance. Regarding the potential link between lead and AD, research has focused on the possible epigenetic effects of lead. In studies of primates, early exposure to lead resulted in increased expression of genes associated with AD (Wu et al., 2008). Separate brain imaging studies in rodents and monkeys have found increases in amyloid plaques in the frontal lobe as well as changes in structural plasticity in the hippocampus (White et al., 2007; Wu et al., 2008).

Mercury is also believed to target the brain. According to the Agency for Toxic Substances and Disease Registry (ATSDR), inorganic, or elemental, mercury is converted into a divalent mercury ion, which can bind to thiol or sulfhydryl groups in proteins, resulting in disruption of protein structure and function (ATSDR, 1999). This may induce oxidative stress, lipid peroxidation, and mitochondrial dysfunction (ATSDR, 1999), which may promote the decline in cognitive performance. Methylmercury obtained via diet can accumulate in both the adult and fetal brain and is slowly transformed into inorganic mercury, which can act via the aforementioned mechanisms (National Academies of Science, 2000). A brain imaging study found that the areas of the brain most vulnerable to the toxic effects of mercury are the calcarine region, pre- and postcentral gyri, and the temporal transverse gyrus (Taber & Hurley, 2008). Granule cells in the cerebellum are also susceptible to harm (Taber & Hurley, 2008).

Air Pollution

Children

One major focus of research on the effects of air pollution is its potential impact on children. Due to the fact that children are continuously undergoing neurological and physical changes, they may be more susceptible to the harmful effects of toxins (Pinkerton & Joad, 2000). This is in contrast to adults, whose nervous systems are more mature and developed, and therefore more resistant to injury. Much of the research on children is centered on the effects of nitrogen dioxide (NO_2) and its influence on the respiratory system (U.S. Environmental Protection Agency [U.S. EPA], 2010). NO_2 principally acts as an irritant on the mucosa of the eye, nose, and throat as well as the rest of the respiratory tract and can decrease lung function in those with pulmonary disease (U.S. EPA, 2010).

Researchers are become increasingly worried, however, about the neurological effects of NO_2. In a study of schoolchildren in China, researchers found a significant association between air pollution, particularly levels of NO_2, and poorer results on neurobehavioral tests designed to measure the children's sensory, motor, and psychomotor functions (Wang et al., 2009). A separate study examined the effects of indoor NO_2 exposure from gas appliances in children and found a dose-response effect between NO_2 levels and cognitive outcomes such as overall cognitive function, verbal abilities, and executive functioning (Morales et al., 2009). Children with greater exposure to NO_2 also had an increased risk of developing attention-deficit hyperactivity disorder (Morales et al., 2009).

Other studies have not focused solely on the harmful effects of nitrogen dioxide but on air pollution's effects as a whole. Using black carbon as a marker for air pollution in an urban birth cohort study, Suglia and co-authors (2008) found that increased amounts of black carbon were predictive of decreased cognitive function based on assessments with the Kaufman Brief Intelligence Test as well as on the Wide Range Assessment of Memory and Learning. In addition, a separate study of air pollution comparing children living in Mexico City to those living in less polluted areas of Mexico found significant deficits in the areas of fluid cognition, memory, and executive functions among the Mexico City children (Calderón-Garcidueñas et al., 2008).

Adults

Air pollution may be equally detrimental to the developed and mature nervous systems in adults. In a study using national data on pollution levels and neurobehavioral test results from the Third National Health and Nutrition Examination Survey, Chen and Schwartz (2009) found a significant association between ozone levels and decreased test results after adjusting for sociodemographic factors. These findings are limited, however, by the authors' use of older data, indicating the need for further examination in a longitudinal study (Chen & Schwartz, 2009). The elderly may be particularly at risk for the potential health effects of air pollution. In a study conducted on elderly residents of urban China, increases in air pollution were associated with difficulties in cognitive functioning as well as performing activities of daily living (Sun & Gu, 2008). Among elderly women in Germany, an association as well as a dose-response relationship was found between long-term exposure to traffic-related air pollution and mild cognitive impairment (Ranft, Schikowski, Sugiri, Krutmann, & Kramer, 2009). Increased levels of ambient traffic-related air pollution (as marked by black carbon) was also associated with decreased cognitive function in a separate study of older men in the U.S. (Power, Weisskopf, Coull, Spiro, & Schwartz, 2011). Some researchers have expressed concern that these declines in cognition due to air pollution could be precursors to neurodegenerative diseases such as dementia and AD (Block & Calderón-Garcidueñas, 2009).

Potential Mechanisms of Action

Regarding the causal mechanism between air pollution and its potential negative health effects, human studies have suggested the brain as a potential target for injury, but the cross-sectional design often utilized in these studies makes it difficult to determine whether exposure to polluted air occurs before or after the negative cognitive effects are present. One of the first studies focused on findings in feral dogs exposed to polluted urban environments (Block & Calderón-Garcidueñas, 2009), and found damage to parts of the brain linked to the nasal pathway, such as the olfactory bulb, implicating the nasal pathway as a potential entry way for pollutants (Calderón-Garcidueñas et al., 2010). In the future, imaging studies in the brain and nervous system may provide clues as to which areas are most vulnerable. A study in mice exposed to air pollution identified the dorsal vagal complex as a potential target (Villarreal-Calderon et al., 2010). In addition, examination by brain MRI of children and dogs in Mexico City exposed to air pollution showed the prefrontal cortex as a potential target as white matter hyperintense lesions were found there (Calderón-Garcidueñas et al., 2008).

A study using animal as well as human models identified multiple potential pathways for damage to the brain and central nervous system (Block & Calderón-Garcidueñas, 2009), including neuroinflamma-

TABLE **1**

Summary of Sources and Potential Mechanisms of Action for Pollutants

Pollutant	Source of Exposure	Potential Mechanisms of Toxicity
Lead	Paint from old buildings Soil near heavily used streets and roads or factories	Increased oxidative stress Stimulation of apoptosis Impairment of neurotransmitter release
Mercury	Consumption of fish and shellfish (particularly shark, swordfish, king mackerel, and tilefish)	Disruption of protein structure and function Increased oxidative stress, lipid peroxidation, and mitochondrial dysfunction
Air pollution	Factories, automobiles, and other machinery	Neuroinflammation Altered immune system response Aggregation of proteins Direct toxic effects
Polychlorinated biphenyls	Consumption of fish and animal fats	Impairment of neurotransmitter release Disruption of intracellular signaling Hormonal imbalance Altered neuronal connectivity
Dioxins	Consumption of animal fats	Interaction with aryl hydrocarbon hydroxylase receptor

tion, altered immune system responses, aggregation of proteins, and the direct toxic effects of the pollutants themselves (Block & Calderón-Garcidueñas, 2009). Air pollutants are thought to increase pro-inflammatory signals in the body as a whole, eventually leading to inflammation in the brain and thereby also affecting the immune system (Block & Calderón-Garcidueñas, 2009). Inflammation and changes in immune system cells, such as microglia, can result in direct damage to brain tissue or damage to the blood-brain barrier, potentially making it easier for pollutants to enter and accumulate in the brain (Block & Calderón-Garcidueñas, 2009). In addition, these pollutants can have direct toxic effects on brain tissue. For example, ozone is a reactive oxygen species that can damage the brain by inducing oxidative stress (Block & Calderón-Garcidueñas, 2009). The presence of multiple mechanisms makes it difficult to determine which effect is most responsible for the damaging effects and underscores the need for further research in this area.

Organic Compounds

Children

Concern about the class of organic chemical compounds known as polychlorinated biphenyls (PCBs) is not new. Although PCBs are commonly used as dielectrics in electrical equipment, fears about their toxicity and persistence in the environment led to a ban by the U.S. Congress in 1979 and by the Stockholm Convention on Persistent Organic Pollutants in 2001 (Porta & Zumeta, 2002). Due to their strong chemical and biological stability, however, concerns about the effects of PCBs still linger. PCBs can accumulate in food chains and are found in animal tissues (Ribas-Fitó, Sala, Kogevinas, & Sunyer, 2001). Currently, the greatest source of PCB exposure in humans is food, particularly fish and animal fats (Ribas-Fitó et al., 2001).

A literature review conducted approximately a decade ago reported an association between PCB exposure and adverse cognitive effects (Ribas-Fitó et al., 2001). A more recent study also found the same relationship (Korrick & Sagiv, 2008). Prenatal exposure to PCBs also appears to have deleterious effects (Jacobson & Jacobson, 2002; Park et al., 2010). One of these studies, however, also found that in children prenatally exposed to PCBs who were also breast-fed, no negative effects were evident, suggesting a possible role of the beneficial nutrients in breast milk offsetting the effects of PCBs (Jacobson & Jacobson, 2002).

Dioxins are another set of organic chemical pollutants that have growing health concerns. Also known as polychlorinated dibenzo-*p*-dioxins (PCDDs), dioxins are persistent environmental contaminants produced as byproducts of industrial processes, such as incineration, smelting, and refining (Tuomisto et al., 2011). Like PCBs, dioxins are highly stable and can accumulate in animals, particularly in their fat stores (Tuomisto et al., 2011), where exposure to humans can occur via diet in addition to industrial exposure.

A few studies have linked dioxins to cognitive or neurobehavioral deficits. Perinatal dioxin exposure was associated with decreases in cognitive and motor abilities (Schellart & Reits, 2008). Prenatal dioxin exposure has also been implicated as a potential risk factor (Vreugdenhil, Lanting, Mulder, Boersma, & Weisglas-Kuperus, 2002). A study conducted in Japan warned about the potential neurobehavioral effects of dioxins to the offspring of heavy fish consumers or residents living near a solid waste incinerator (Yoshida, Ikeda, & Nakanishi, 2000). A separate study, however, found no decreases in the cognitive test performance of children potentially exposed to dioxins by breast-feeding (Patandin et al., 1999), again raising the possibility that breast-feeding confers specific protective benefits to counteract the effects of the dioxins.

Adults

In addition to the potential effects on children, PCBs and dioxins may also have harmful effects on adults. Fewer studies, however, have explored these links. Concern is growing about the links between PCB exposure and impairments in cognitive functions in adults and older adults, with potential links to dementia (Lin, Huang, Yeh, Kuo, & Ke, 2010). It has been suggested that neurocognitive deficits occur only in women with PCB exposure, not exposed men (Lin, Huang, Yeh, Kuo, & Ke, 2008). Dioxin exposure in adults is also a concern. Air Force veterans exposed to high dioxin levels while serving in the Vietnam War have demonstrated decreases in several measures of memory functioning (Barrett, Morris, Akhtar, & Michalek, 2001). The amount of literature on this topic is limited, however, due to the focus on children.

Clearly, further research is needed to evaluate the significance of these associations.

Potential Mechanisms of Action

The potential mechanism by which PCBs or dioxins exert their effects is unclear. A comprehensive review by Kodavanti (2005) examined three possible mechanisms: decreased release of neurotransmitters, disruption of intracellular signaling, and hormonal imbalances. A study by Lein and co-authors (2007) investigated the effect of PCB exposure on neuronal connectivity in rats and found PCBs altered connectivity in critical regions in the brain such as the hippocampus, cerebellum, and cortex, but similar evidence in humans is lacking. Hormonal imbalance is another concern. In a study of rats, PCBs interfered with the neuroendocrine system, resulting in impaired sexual differentiation of the female hypothalamus (Dickerson, Cunningham, Patisaul, Woller, & Gore, 2011). PCBs may also alter the brain capillary endothelium, as a study in mice found (Seelbach et al., 2010).

The biological mode of action for dioxin's effects is also unknown. One possibility is dioxin's interaction with the aryl hydrocarbon hydroxylase (Ah) receptor (Charnley & Kimbrough, 2006). Binding to the Ah receptor up-regulates enzymes such as the P450 cytochromes, which are responsible for both activating and deactivating toxins (Charnley & Kimbrough, 2006). It is currently unclear how exactly the deficits in neurocognition may result, however. A summary of the sources and potential mechanisms of toxicity for the pollutants mentioned can be found in Table 1.

Sociodemographic Issues

An important consideration of the negative impact of pollution on humans is whether it may affect certain socioeconomic groups more than others. For example, in terms of lead exposure, blood levels higher than the CDC's previous threshold of 10 µg/dL are much more common among minority children, children in low-income families, and children living in older homes (Bellinger, 2008). Similar patterns are seen for mercury (Friere et al., 2010). Although air pollution is generally thought to affect individuals more equally, this may not be the case. In a study of more than 150,000 public schoolchildren in Orange County, Florida, it was found that Hispanic and African-American children had greater exposure to air pollution than Caucasian children, and African-American children had greater exposure than Hispanic children (Chakraborty & Zandbergen, 2007). PCB exposure follows similar trends. Korrick and Sagiv (2008) concluded that a sociodemographic disadvantage may enhance PCB toxicity while a sociodemographic advantage may mitigate PCB toxicity.

Sociodemographic findings are important for identifying individuals with potentially greater risk of exposure and subsequent negative outcomes. For example, living in urban areas, where exposure is greater and where minority groups make up a larger percentage of residents, may be responsible for these findings (Chen & Schwartz, 2009). Living in rural areas presents its own unique risks due to greater exposure to agricultural pesticides such as organophosphates (Lizardi, O'Rourke, & Morris, 2008). Questions about unequal exposure to pollutants are important areas for further study and may provide significant insight into developing strategies to mitigate the risks of exposure.

Discussion

An important consideration discussed by Winneke (2011) is that when studying pollutants, IQ is generally used to quantify the effects on neurological function. Although this is useful in determining regulatory cutoffs, it is not as helpful in clinical settings (Winneke, 2011). Instead, assessment tools that focus on particular areas such as executive function measures or outcome measures can be used to assess specific effects rather than global effects. For example, use of specific screening instruments for mild cognitive impairment, a precursor stage that often leads to dementia, can be used to investigate the role of lead and mercury in the development of neurodegenerative disorders. Overall, these tools were created on a neuropsychological basis and relate to specific neurologic disorders and are therefore better equipped to identify problem areas and lead to the development of appropriate interventions (Winneke, 2011).

The findings from the above studies suggest how exposure to these toxic pollutants can result in neurological defects. One major issue with these studies, however, is that it is often difficult to determine if exposure to the pollutants is the true cause of these defects. For example, exposure to the pollutants may have occurred after neurological effects were present. Or perhaps a known or unknown confounding variable may be responsible for the effects. For example, socioeconomic or genetic risk factors may play a role. Another issue is that the people in the study may have differing levels of exposure, making it difficult to quantify a potential dose-response relationship.

In vitro studies may only be able to fill in some of the gaps left by population studies. Using cell cultures, researchers may be able to deduce the mechanisms of action of toxins and pollutants on a cellular and biochemical level. As pointed out by Cannon and Greenamyre (2011), however, these studies cannot reproduce the complexity of the human nervous system, and therefore are not able to model the pathogenesis of these neurological defects. Animal studies may potentially serve as the best way to study pollution's effects until more superior research methods are developed. In animal models, exposure can be quantified and assessed methodically. This is especially important for air pollution, which is particularly difficult to quantify exposure in human studies. Animal models may also serve as a way to model the mechanisms of action of pollutants on the nervous system as a whole, rather than just on a cellular level.

One potential future area of study is the difference in impact on neurologic function between prenatal exposure and postnatal exposure. As discussed earlier, prenatal exposure to lead, mercury, PCBs, and dioxins has been linked to neurological effects. One critical difference between these exposure periods is the route of exposure. During the prenatal period, the developing fetus is exposed to pollutants via the bloodstream through the placenta and umbilical cord. Postnatal exposure may occur through different routes such as inhalation, ingestion, absorption, etc. In particular, it would be interesting to study if air pollution exposure (which occurs through an inhalation route after birth) has neurodevelopmental effects in fetuses exposed via the bloodstream through umbilical cords. Although many have studied the effects of smoking during the prenatal period, the effects of air pollution as a whole during this same time period are unknown.

Conclusion

Pollution is a significant and complex problem that affects infants, children, adults, and seniors. Airborne and organic chemical pollutants have received less attention in the literature than heavy metals but are a growing cause for concern. Even though the deleterious effects of heavy metals such as lead and mercury have been well studied, there is a growing debate over newly proposed methods of exposure as well as what amount of exposure is unsafe. In addition, significant concern exists that air pollution can significantly affect the developing nervous systems of children as well as the mature nervous systems of adults. Researchers have thus far been unable to pinpoint the exact mechanisms of injury. Air pollution may act via multiple pathways, imparting wide-reaching effects on the brain and central nervous system. Organic chemical pollutants such as PCBs and dioxins also are increasingly viewed as emerging threats due to their prolonged persistence in the environment and ability to accumulate in food chains. Future studies must explore the effects of PCBs and dioxins on adult populations. Evidence from the study of sociodemographic factors provides important and compelling data to prompt further consideration of pollution exposure and effects across different populations. Careful study of exposure levels for different socioeconomic groups is an important study for future research in order to identify at-risk groups.

A popular technique in assessing the neurological effects of pollutants is the use of global measures such as IQ, which is useful for determining cutoffs and acceptable levels of exposure. The use of specific scales and outcome measures in future studies may be more useful clinically. Another potential area of study in the future is comparing the effects of pollutants in prenatal versus postnatal time periods. Differing routes of exposures as well as different developmental stages may lead to different effects.

The difficulty with studying the effects of pollutants on the nervous system via human studies is that exposure is hard to quantify. In vitro cell culture studies may elicit the mechanisms of action of pollutants on a cellular level but cannot reproduce the complexity of the nervous system. Animal studies allow exposure to be quantified in a more complex model, but results may not be comparable to human studies. Improvements in brain imaging studies may provide more helpful clues in the future. Treatment and prevention become a reality only when the underlying mechanisms are fully understood.

Combatting these effects of pollution requires an interdisciplinary approach that encompasses multiple professions and fields of study. For example, public health officials can use public outreach programs to educate vulnerable populations on how to reduce exposure. Multidisciplinary research in the fields of biology, chemistry, ecology, neurology, and more can address gaps in the science and better identify the mechanisms and potential effects of different pollutants on the brain while potentially shaping public health policy, contributing to legislation and regulation, and stimulating research and development of environmental technologies.

Acknowledgement: This work was supported by an NIH/NIEHS grant: K02-ES019878-01 (US).

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continued on page 136

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continued from page 135

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continued on page 138

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▸ GUEST COMMENTARY

Prepublished online November 2013,
National Environmental Health Association.

The Cell Phone Problem/Solution

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Part 1: The Problem

Cell phone usage in the U.S. has ballooned over the past 15 years, from 33.8 million subscribers in 1995 to an estimated 285.6 million in 2010 (Cellular Telephone Industries Association, 2012). Global measures of cell phone adoption are even more impressive; the five billionth subscriber was recorded on July 8, 2010 (Ericsson, 2010). While cell phone popularity is rising, efforts to recycle cell phones remain unpopular worldwide. The average cell phone is expected to last five years, and most Americans replace their cell phone every 12 months, usually because their contract has expired or they desire a phone with new features (Pasternak, 2009). In 2009, an estimated 141 million cell phones were discarded. Of these, only about 8%, or 11.7 million cell phones, were recycled (U.S. Environmental Protection Agency [U.S. EPA], 2011a). It would seem the general public in the U.S. is either unaware of the harm done to the environment by disposing of phones incorrectly, or they think that the inconvenience of recycling outweighs the benefits.

We believe that cell phone recycling is particularly important because (1) cell phones contain materials that are toxic when disposed of improperly, and (2) reclaiming reusable materials would reduce the demand for a trade in rare minerals that is fraught with unsafe and abusive labor practices. If phone recycling remains unpopular, the growing volume of electronic waste (e-waste) will produce an increasingly serious threat to the health of humans, animals, and the ecosystems they share.

The Environmental Impact of Discarded Mobile Devices

When mobile devices, such as cell phones, are discarded in landfills, three main components—the circuit board, liquid crystal display, and rechargeable battery—leach toxic metals into the environment (U.S. EPA, 2011b), which can cause adverse impacts on human health and the environment. The components of this waste are plastics (about 40% of the waste), metals (about 40%), and ceramics/trace materials (about 20%), all of which are known to contain "persistent toxins" that remain in the environment, often for centuries (U.S. EPA, 2011b).

While some plastics are relatively stable, brominated flame retardants (BFRs) used in plastics to prevent electronics from catching on fire are known to be persistent, bioaccumulative, and toxic to both humans and the environment (U.S. EPA, 2011c). Several studies have shown that PentaBDE, a specific isomer of a group of BFRs, has deleterious morphological, hormonal, and immunological effects on tested mammals. OctaBDE, another isomer of the group of BFRs, is associated with detrimental mammalian fetal effects that result in reduced ossification, bent ribs, and limp bones (Darnerud, 2003). Although the U.S. Environmental Protection Agency (U.S. EPA) has expressed an intention to begin regulating BFRs, some reports indicate that levels were continuing to increase as of 2009 (U.S. EPA, 2011c).

Cell phones also contain metal components, such as lead, nickel, and beryllium. Similar to BFRs, various lead-based compounds are highly persistent in the environment for prolonged periods of time. A blood concentration of less than 10 µg/dL can affect neurological development in humans (Bellinger, 2008a, 2008b; Chen, Cai, Dietrich, Radcliffe, & Rogan, 2007; Gould, 2009; Lanphear, Burgoon, Rust, Eberly, & Galke, 1998). Unfortunately, many people suffer lead poisoning without identifying the cause of their symptoms because ingestion of small amounts of lead has few early physical manifestations and the effects become more pronounced only after long periods of time. As a by-product of e-waste, lead has negatively impacted the health of workers at poorly regulated e-waste recycling plants. For example, in Guiyu, China, where 60%–80% of families work at e-waste recycling plants with unsafe recovery processes, 81.8% of children were shown to have blood lead levels higher than 10 µg/dL (Huo et al., 2007). By comparison, the national average of children with lead poisoning in the U.S. is only 1.4%, significantly lower than the prevalence in Guiyu (Iowa Department of Public Health, 2010). Funneling old phones into programs that maintain safe recycling practices could reduce the amount of e-waste that is shunted towards under-regulated recycling plants, or force underperforming plants to implement competitive safety measures to decrease the rates of lead poisoning in such settings.

While nickel and beryllium poisoning is less common than lead poisoning, their effects can be equally problematic in terms of human health. The most commonly reported adverse health effect from nickel exposure is contact dermatitis resulting from an allergic reaction (Agency for Toxic Substances and Disease Registry [ATSDR], 2005). Prolonged exposure to nickel can result in a spectrum of respiratory diseases as a consequence of lung inflammation, edema, and fibrosis (Phillips, Green, Davies, & Murray, 2010). Furthermore, studies have confirmed a positive association between lung cancer and exposure to nickel compounds such as nickel subsulfide and nickel oxide (Beveridge, Pintos, Parent, Asselin, & Siemiatycki, 2010; Landolph, Verma, Ramnath, & Clemens, 2002). These carcinogenic effects are most pronounced among occupationally exposed workers in nickel mining and stainless steel production, but the general public may be at risk as well (ATSDR, 2005).

Similarly, beryllium poisoning is most often associated with occupational exposure. Acute lung damage can occur through beryllium inhalation if it is present at sufficient levels in the air (greater than 1,000 µg/m^3). Furthermore, those who become sensitized to beryllium may develop chronic beryllium disease, an inflammatory reaction in the respiratory tract that results in shortness of breath, chest and joint pain, dry coughs, and fatigue. As with nickel, beryllium is associated with an increased risk of lung cancer (ATSDR, 2010; University of California San Francisco Medical Center, 2009).

While the environmental impact of toxic compounds in discarded mobile devices is readily apparent, the circumstances under which e-waste accumulation directly contributes to toxic levels of exposure to BFRs, lead, nickel, and beryllium in that waste have not been extensively investigated. With the increasing global burden of e-waste, including cell phone waste, human health impact is an area that requires further research.

The Human Cost of Minerals Used in Electronics

Mobile device recycling programs also have the potential to minimize the human and environmental costs of mineral mining in regions with ineffectual human rights protections. Since 1996, when a Rwandan invasion of the Democratic Republic of Congo (DRC) triggered a widespread political splintering, a number of armed factions have struggled to maintain control of the enormous mineral wealth in the eastern mountains of the country. Using violence to seize land and coerce civilians into mining, often under extremely hazardous conditions, these groups have funded their military pursuits for 14 years on so-called "conflict minerals (Polgreen, 2008)."

In many cases, the economic wealth gleaned from these operations—between $144 million and $218 million per mine per year—has been so substantial that the original political agendas have been abandoned in favor of securing absolute control over mining and laborers (Schure, 2010). To this end, some of the most appalling human rights abuses have been committed, including rape, torture, murder of unarmed individuals, looting, recruiting children as soldiers, and the forced relocation of hundreds of thousands of people (Global Witness, 2009; United Nations News Centre, 2009).

The term "conflict minerals" refers to several highly lucrative and commonly mined metallic ores including coltan, wolframite, and cassiterite, all of which are refined into precious metals (tantalum, tungsten, and tin, respectively) that form essential components of cell phones (Mineral Information Institute, 2010). Most manufacturing companies acquire these rare minerals through middlemen in East Asia via Rwanda or Burundi, indirectly funding the warring parties in the DRC and maintaining demand for their ill-gotten goods (Litvinsky, 2009). Instead of obtaining virgin precious metals, a much greater portion of the precious metals used in mobile device manufacturing could be reclaimed from old cell phones through appropriate recycling processes (Mooallem, 2008). Thus, the casual disposal of cell phones indirectly enables illicit mining operations to thrive by depriving the market of precious metals that could otherwise be reused for manufacturing new phones.

The Need for Mobile Device Recycling

Given the grave toxicities and human rights abuses that result from improperly processed e-waste, a stronger impetus than ever exists to develop safe, appealing, and effective recycling programs. While public entities such as U.S. EPA certainly have a role to play in regulating e-waste management, a possibility exists for private entities to treat discarded phones as an opportunity and to initiate recycling programs that drive consumer behavior directly. Currently, most major cell phone companies have recognized the need for cell phone recycling and are accepting unwanted cell phones at many of their retail locations nationwide (U.S. EPA, 2013). Despite these efforts, however, only about 10% of discarded cell phones in the U.S. by weight are recycled (U.S. EPA, 2013). These disappointing statistics may stem from being unaware of the environmental implications of e-waste, the perceived inconvenience of recycling, the belief that old or broken phones are devoid of value, or the lack of obvious benefits for the phone recycler.

In the second part of this article, we discuss an initiative that aims to increase recycling rates by linking phone recycling to a charity that uses mobile phones to strengthen health services in the developing world. Through such initiatives, a discarded mobile device may be reimagined as an opportunity by those who embrace the ethos of green responsibility and the integrity of human life.

Part 2: A Solution

In part 1 of this commentary, we detailed the environmental health, human health, and human rights implications of cell phone electronic waste (e-waste) recycling practices in the U.S. Here we describe a novel recycling campaign called Hope Phones as a case study that aims to address many shortcomings of status quo recycling programs (of which there are more than 500; U.S. EPA, 2010). Although hundreds of charities exist with thousands of drop-off points throughout the U.S., cell phone recycling rates linger at 8% (U.S. EPA, 2011). The subtleties of why certain people choose to recycle and others do not is difficult to characterize, but experts agree that most people do not recycle because they are unaware of the environmental consequences of improper disposal; they do not know how to go about recycling phones; or the knowledge of environmental harms does not outweigh the perceived inconvenience of recycling (U.S. EPA, 2010).

The Hope Phones campaign launched in August of 2009 with the slogan, "Old phones save lives; donate yours to a clinic in the developing world today." This national effort was designed to improve on current recycling programs by 1) making a clear commitment to researching and communicating the harm of e-waste, 2) guaranteeing the safe recycling of cell phones, 3) ensuring a simple and painless recycling process, 4) supplying phones to support mobile health (mHealth) initiatives in the developing world, and 5) framing cell phone recycling as a donation attached to valued social benefits.

Awareness

One explanation for low cell phone recycling rates is that the public lacks awareness about recycling programs and how they work, a barrier identified by the U.S. EPA Office of Solid Waste (U.S. EPA, 2012). In the U.S., the corporate relationships with cell phone consumers play a role in this problem. Cell phones are generally manufactured by companies such as LG, Motorola, and Samsung, which stand to benefit the most from recycling programs, since reclaimed metals can lower the production costs of the next generation of cell phones. Consumers acquire those cell phones and services through carriers such as AT&T, Verizon, and Sprint. Carrier companies, unlike manufacturers, have little financial investment in whether consumers recycle or not, and so their recycling programs generally fall by the wayside with minimal advertisement (U.S. EPA, 2010). Manufacturing companies have difficulty reaching out to consumers about recycling because their interaction with them is almost entirely mediated through carrier organizations (U.S. EPA, 2010).

As a nonprofit campaign that has no stake in either the cell phone manufacturing process or the sale of service contracts to consumers, Hope Phones has the freedom to conduct unbiased independent research and advertisement campaigns. To increase awareness of cell phone recycling, the Hope Phones campaign built a Web site, www.hopephones.org. Additionally, Hope Phones engages potential followers via social media outlets including Twitter and Facebook, and has attracted tens of thousands of visitors to the Web site through donated online advertisements.

The Process of Phone Recycling

As discussed in part 1 of this commentary, nonexistent safety regulations in developing world e-waste recycling plants place workers at significant health risk. In fact, unsafe recycling practices are associated with among the highest levels of toxicity currently reported due to the metals in electronics (Huo et al., 2007). Keenly aware of these practices, Hope Phones has partnered with an organization, The Wireless Source, that is committed to both safe and efficient recycling practices. Approximately 40% of recycled cell phones cannot be refurbished for reuse and instead must be processed through a refining process known as "above-ground mining" (GRC Wireless Recycling, 2013). At The Wireless Source, such cell phones are shredded and smelted in a refiner so that valuable metals including gold and copper can be reclaimed. Simultaneously, a gas-cleaning-and-filtration system captures and neutralizes any toxic chemicals that are produced (Mooallem, 2008). By weight, 99.5% of the cell phones taken in are converted into a usable form (Mooallem, 2008). Each processed cell phone contains about one dollar's worth of precious metals, which are then sold to be used in the manufacturing of new electronics. Ultimately, recycling cell phones by these methods protects workers at the recycling plant, saves energy spent on mining new metals, provides components to manufacture new cell phones, and keeps potentially toxic-yet-usable materials out of landfills.

(In)convenience

For the average cell phone user, perhaps the greatest barrier to recycling is the perceived inconvenience of the process. The typical phone user in the U.S. replaces a phone about once per year (Pasternak, 2010), but unlike bottles, plastic bags, and other recyclable goods that are discarded frequently and can be picked up at people's homes on a regular schedule, phones are discarded at low volumes without home pickup. As a result, phone recycling is viewed to be less convenient than other forms of recycling. Even those who are aware of phone recycling are often unwilling to find a company that will accept a one-phone shipment, procure and address a package, and pay for shipping. Hope Phones addresses this problem by simplifying the recycling pipeline. Anyone with old phones can specify how many they wish to contribute on the campaign Web site where they are then prompted to print out a pre-paid shipping label. After placing the phones in a box and affixing the shipping label, everything can be shipped to The Wireless Source at no cost to the recycler. For donations of 20 or more phones, clients can have a box sent to their home or workplace free of charge.

While the availability of online shipping labels promises to bring many people who are on the margin into the recycling fold, this system does not provide a tangible reminder of recycling in the way that conventional bins do. To simultaneously increase convenience and visual awareness, the Hope Phones campaign also developed drop boxes that can be sent to larger donation sites. To date, more than 1,000 collection boxes have been distributed throughout college campuses and large workplaces.

Hope Phones and Medic Mobile

The centerpiece of the Hope Phones campaign is the conversion of cell phone recycling into a tangible social good through its relationship with the nonprofit Medic Mobile. In the developing world, lack of infrastructure prevents community health workers (CHWs) from delivering efficient health care to rural areas. These CHWs are responsible for coordinating care between physicians and patients who are located at a distance from central clinics. Unfortunately, as CHWs travel from clinics to reach isolated patients (often on foot) they are as disconnected from physicians as the patients they are trying to serve. Medic Mobile helps clinics overcome these barriers by utilizing free and open source software applications that enable instantaneous two-way communication between CHWs and clinical workers. These tools rely on short message service (SMS) messages and can operate wherever a cell phone signal is present. By equipping CHWs with cell phones and clinics with a laptop running the necessary software, Medic Mobile has allowed clinics to triage emergency medical care, remotely monitor patient medication adherence, and drastically cut back on transportation and fuel costs, among other benefits (Mahmud, Rodriguez, & Nesbit, 2010).

Hope Phones funds the purchase of mobile phones and hardware for Medic Mobile through proceeds from the recycling and resale of used cell phones. The average donated phone can be exchanged for about two basic cell phones which, when put in the hands of health workers, can each give 50 families access to emergency medical care, health information, transport services, and clinic resources that would otherwise be less accessible. Since Hope Phones' inception in 2009, over 30,000 cell phones have been recycled through the campaign (for a monetized value of more than $50,000), but given the rate of expansion of Medic Mobile and its planned field sites, the organization will need far more (Hope Phones, 2010). Based on current figures, if even 1% of the cell phones discarded annually in the U.S. were recycled, one million CHWs worldwide could be given cell phones. The Hope Phones campaign looks to bridge this gap by moving the public into action through demonstrated efficacy, tangible social returns, and the conviction that one phone donated to Hope Phones will have a greater impact than one recycled conventionally.

The Perceived Value of a Recycled Phone

The collaboration between Hope Phones and Medic Mobile serves the dual purpose of generating funding for mHealth programs and changing the way that individuals perceive phone recycling. Many regard the recycling of a phone in exchange for a few dollars as impractical or not worthwhile. The importance of even very small donations to humanitarian causes, however, makes the acquisition of funds on their behalf both more acceptable and socially promoted. For this reason, Hope Phones rebrands cell phone recycling as cell phone donation. For many, the concept of long-term environmental damage may seem distant and amorphous, easily unhinging lax cell phone recycling from this process. Shortcomings in health care provisions, by contrast, are relatable to the American public en masse, and the idea that an old cell phone can fill these gaps for many less fortunate patients can be both moving and appealing. This is the aspirational component of the campaign, that a technological relic void of value for one person can be repurposed to bring hope (and health care) to others.

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continued on page 144

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INTERNATIONAL PERSPECTIVES

Prepublished online November 2013, National Environmental Health Association.

Geochemical Correlates to Type 1 Diabetes Incidence in Southeast Sweden: An Environmental Impact?

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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Abstract **The authors' aim was to explore whether geological factors contribute to geographical variation in the incidence of type 1 diabetes. All children diagnosed with type 1 diabetes in southeastern Sweden during 1977–2006 were defined geographically by their place of residence and were allocated *x* and *y* coordinates in the national grid. The population at risk, all children 0–16 years of age, was geocoded in a similar manner. Three of the analyzed minerals in moraine and one of the analyzed minerals in brook water plants were significantly associated with type 1 diabetes at the time of diagnosis. Additionally, the birthplace of the children who subsequently developed diabetes differed in relation to some of the minerals. In communities with high incidence and in communities with low incidence, children were diagnosed with type 1 diabetes in areas with the same high or low level of elements. The authors' findings in their pilot study indicate a possible geographical covariation of incidence and some geological factors.**

Introduction

Type 1 diabetes is a heterogeneous disease involving genetic, environmental, and autoimmune mechanisms (Akerblom, Vaarala, Hyoty, Ilonen, & Knip, 2002). That environmental factors play a central role is supported by the clear seasonal variation in the diagnosis of the disease (Padaiga et al., 1999) and its increasing incidence in many countries (Patterson et al., 2009). Several studies have also indicated different incidence rates and geographical variation between, and within, countries (Rytkonen et al., 2001; Samuelsson & Löfman, 2004).

Moreover, the offspring of an Asian transmigratory population showed a rising incidence of type 1 diabetes approaching that of the indigenous English population (Bodansky, Staines, Stephensen, Haigh, & Cartwright, 1992).

Several studies have found spatial clustering at the time of diagnosis (Cherubini et al., 1999; Staines et al., 1997). We and others have also reported time-space clustering at diagnosis (Law et al., 1997; Samuelsson, Carstensen, Johansson, & Ludvigsson, 1994). Furthermore, evidence has been produced of time-space clustering of date of birth in children who later develop diabetes (Dahlquist & Källen, 1996).

The studies of spatial clustering have shown that the incidence is higher in rural than in urban areas (Rytkonen et al., 2003; Staines et al., 1997) and higher in highlands than in lowlands (Patterson, Smith, & Webb, 1988). Socioeconomic factors may also influence the risk of developing diabetes, with a lower incidence in deprived areas and in areas with a high population density and overcrowding (Staines et al., 1997).

Several studies have examined the relationship between the risk of disease and compounds in drinking water (Zhao et al., 2001), as well as the acidity of the water (Stene, Hongve, Magnus, Ronningen, & Joner, 2002). It has been suggested that zinc, and to some extent copper and magnesium, have a protective effect against developing diabetes (Haglund, Ryckenberg, Selinus, & Dahlquist, 1996; Zhao et al., 2001), whereas high levels of nitrate/nitrite, mercury, and arsenic in drinking water may increase the risk (Lai et al., 1994; Parslow et al., 1997; Zhao et al., 2001). In contrast, however, some studies have found that no obvious associations exist between nitrate/nitrite, vanadium, and chromium and the risk of dia-

TABLE **1**

Minerals in Moraine (1a) and Minerals in Brook Water Plants (1b)

1a

Mineral	Unit
Group 1–2	
Magnesium oxide (MgO)	% dry weight
Sodium oxide (Na_2O)	% dry weight
# Calcium oxide (CaO) 4	% dry weight
Strontium (Sr)	% dry weight
Rubidium (Rb)	% dry weight
Barium oxide (BaO)	% dry weight
Group 3–12	
Manganese oxide (MnO)	% dry weight
Titanium oxide (TiO_2)	% dry weight
Iron oxide (Fe_2O_3)	% dry weight
Cobolt (Co)	ppm[a]
Copper (Cu)	ppm
Nickel (Ni)	ppm
Vanadium (V)	ppm
Zinc (Zn)	ppm
Chromium (Cr)	ppm
Molybdenum (Mo)	ppm
Group 13–18	
Silicon oxide (SiO_2)	% dry weight
Phosphorus oxide (P_2O_5)	% dry weight
Aluminium oxide (Al_2O_3)	% dry weight
Chlorine (Cl)	ppm
Arsenic (As)	ppm
Lead (Pb)	ppm

1b

Mineral	Unit
Group 3–12	
Cobalt (Co)	ppm
Chromium (Cr)	ppm
Copper (Cu)	ppm
Nickel (Ni)	ppm
Vanadium (V)	ppm
Zinc (Zn)	ppm
Cadmium (Cd)	ppm
Mercury (Hg)	ppm
Group 13–18	
Arsenic (As)	ppm
Lead (Pb)	ppm

Note. These minerals were analyzed by the Swedish Geological Survey. The different compounds are arranged according to groups in the periodical system.
[a]ppm = parts per million.

betes (Casu, Carlini, Contu, Bottazzo, & Songini, 2000; Haglund et al., 1996).

A previous study from southeast Sweden showed a clear geographical variation in incidence of type 1 diabetes (Samuelsson & Löfman, 2004). During a period of 25 years (1977–2001) the community with the highest incidence in this region had an incidence of 52.4/100,000 children 0–15 years with type 1 diabetes compared with an incidence of 14.3/100,000 in the community with the lowest incidence. A recent follow-up study from the same region found that relative socioeconomic deprivation influences the incidence of type 1 diabetes, with the lowest incidence in the most deprived areas (Holmqvist, Löfman, & Samuelsson, 2008). The aim of our study was to explore whether geological factors may contribute to the geographical variation of incidence of type 1 diabetes in the southeast region of Sweden.

As we had no specific hypothesis regarding physiological mechanisms or the geological factor(s) that might determine the risk of diabetes, we simply began a mapping exercise using analyzed data from a survey of the study area conducted by the Geological Survey of Sweden (SGU). The density of the number of analyzed sites varied depending on which compound was analyzed. Using this data-mining approach, our study would be able to clarify any spatial associations and generate further hypotheses on exposure mechanisms and causality.

Methods

Setting and Population at Risk

In the Swedish health care system, all children up to 16–18 years of age with diabetes are diagnosed and treated at pediatric clinics. The catchment areas of these clinics form a continuous geographical region of 42,356 km^2, which, during 1977–2006, had a mean population of risk comprising 276,930 children ≤16 years of age. The highest figures were seen in 1977 (300,139) and 1978 (298,601); the lowest were in the final two years of the study period, 2005 and 2006 (258,453 and 256,682, respectively). This variation is due mainly to variations in birth rate. Different parts of the study region had a similar variation, as seen in the entire country during those years. Southeast Sweden has five counties (Blekinge, Jönköping, Kalmar, Kronoberg, and Östergötland), 49 municipalities, and 525 parishes. Large parts of the region are predominantly rural. Each county has one or two pediatric clinics and, by Swedish standards, one or two major cities (above 50,000–60,000 inhabitants, all ages included).

Sampling of Cases and Geocoding in a GIS

From the seven pediatric clinics in southeastern Sweden, we included all 2,412 (1,282 boys and 1,130 girls) children ≤16 years of age who had been diagnosed with type 1 diabetes between 1977 and 2006.

Information regarding date of birth, sex, residential address (at the time of diagnosis), and date of diagnosis (= first insulin injection), was extracted from medical records by nurses or doctors in the diabetes teams at each clinic. As all cases are registered locally, we can be sure that the study includes all diabetic cases in the region. To strengthen this further we compared our figures with the reports to the Swedish Diabetes Registry, which is estimated to have completeness close to 100%. We found complete concordance.

The Swedish population registry is updated every year. All patients were linked to the registry as recorded on December 31 for the year of diabetes onset by use of the personal identification number that is unique to all Swedish citizens. The patients were then defined geographically according to their place of residence by matching the population registry to a national property register, as reported in a previous study (Kohli, Noorlind Brage, & Löfman, 2000). Using this procedure, the *x* and *y* coordinates in the national grid for the centroids of the residences were added to the data file and geocoded in GIS (GIS software ArcInfo 9.2). We were unable to find the coordinates for the place of residence and birthplace of 26 of the 2,412 children. The remaining 2,386 children diagnosed with type 1 diabetes in southeast Sweden were included and allocated *x* and *y* coordinates in the national grid. Thirty-three of the 2,386 children were diagnosed within the area but lived outside and were therefore excluded from the analysis. Furthermore, 1,932 of the 2,353 children included were born within the study area

TABLE **2**

Calculated Incidence Rate Ratios (IRRs) with 95% Confidence Interval (*CI*) for Type 1 Diabetes in Relation to Certain Minerals in Moraine and Brook Water Plants in the Area of Residence at the Time of Diagnosis

Variable	Range	# of Squares	Obs[a]	Exp[a]	Age 0–16 Population	IRR (95% *CI*) Univariate	*p*-Value	Adjusted IRR Multivariate	*p*-Value
Level of barium oxide in moraine in the area	>90th perc[a]	4656	96	223	26521	1	(.000*)	1	–
	10th–90th perc	32949	1719	1807	215122	2.21 (1.80, 2.71)	.000	2.21 (1.80, 2.74)	.000
	<10th perc	6665	538	330	39298	3.78 (3.04, 4.70)	.000	3.93 (3.13, 4.94)	.000
Level of manganese oxide in moraine in the area	>90th perc	4896	234	255	30400	1	(.000*)	1	–
	10th–90th perc	32226	1618	1703	202781	1.04 (0.90, 1.19)	.61	1.09 (0.94, 1.27)	.27
	<10th perc	7148	501	401	47760	1.36 (1.17, 1.59)	.000	1.20 (1.01, 1.42)	.045
Level of titanium oxide in moraine in the area	>90th perc	3941	182	189	22482	1	(.024*)	1	–
	10th–90th perc	36279	1953	1991	236994	1.02 (0.87, 1.19)	.82	0.90 (0.77, 1.06)	.73
	<10th perc	4050	218	180	21465	1.26 (1.03, 1.53)	.025	0.96 (0.78, 1.19)	.19
Level of zinc in moraine in the area	>90th perc	5649	391	368	43767	1	–	1	–
	10th–90th perc	32993	1679	1723	205181	0.92 (0.82, 1.02)	.92	0.89 (0.79, 1.01)	.68
	<10th perc	5628	283	342	31993	0.99 (0.85, 1.16)	.90	0.90 (0.76, 1.08)	.22
Level of zinc in brook water plants in the area	>90th perc	7102	600	587	69996	1	–	1	–
	10th–90th perc	33581	1600	1617	192544	0.97 (0.88, 1.07)	.52	1.06 (0.97, 1.17)	.21
	<10th perc	3559	153	154	18305	0.98 (0.82, 1.17)	.78	0.78 (0.64, 0.93)	.007
Level of mercury in brook water plants in the area	>90th perc	770	34	37	4706	1	(.000*)	–	–
	10th–90th perc	8219	440	488	62573	0.97 (0.69, 1.38)	.88	–	–
	<10th perc	960	80	28	3597	3.08 (2.06, 4.61)	.000	–	–

Note. Logistic regression model.
[a]Obs = observed number of children 0–16 years of age at diagnosis; Exp = expected number of children; Perc = percentile.
* = the Chi-squared test for trend (*df* = 1).

and their birthplace was geocoded in the same way as their place of residence at onset.

The population at risk, comprising all children 0–16 years of age, were aggregated in 82,000 200-m squares covering the study area and geocoded by the centroid of each square.

Sampling of Environmental Data

We received information about the mineral and the pH-value sampling locations from SGU. The sampling locations were matched to cases and population at risk by a GIS procedure using a nearest distance method. The samples of the different elements (Table 1a) were taken in moraine 1 m (3.28 feet) underground in 6,029 different places in the study region. All samples were freeze dried or vacuum dried and sifted to <0.06 mm before analysis with X-ray fluorescence (radiofluorescence) on the C horizon of the moraine. Some elements were also analyzed in living brook water plants in 6,929 different places (Figure 1) with the exception of mercury, which was analyzed in 1,179 places. The analyses were based mainly on roots from *Carex* species and *Filipendula antipyretica*. The content of the oxides are stated in percentage dry weight and the trace elements in parts per million.

The pH values were measured on dried samples (0.06 mm) in 977 places within the study area (Figure 1). The mean value of pH in this area was 5.06 ± 0.5, range 4–8.3. We grouped the different areas, regardless of the units of measurement, in three quantile intervals: areas with a concentration of a mineral <10th percentile; areas with a concentration >90th percentile; and areas with a concentration in between.

To evaluate the spatial relationship between diabetes incidence and barium oxide concentration we first explored data by interpolated maps using ordinary kriging. The >6,000 observations of barium oxide were averaged at community level. To conform to a common scale, data were converted to their *z* scores and *z* incidence, and *z* barium oxide was compared in a raster analysis.

Statistical Analysis

The expected number of cases was estimated by using the number of children 0–16 years of age in southeast Sweden during 1995, which was 284,190 children. This figure is also rather near the average number of children in the area during 1977–2006: 276,930 children.

TABLE 3

Calculated Incidence Rate Ratios (IRRs) with 95% Confidence Interval (*CI*) for Type 1 Diabetes in Relation to Certain Minerals in Moraine and in Brook Water Plants in the Area at the Time of Birth

Variable	Range	# of Squares	Obs[a]	Exp[a]	Age 0–16 Population	IRR (95% *CI*) Univariate	*p*-Value	Adjusted IRR Multivariate	*p*-Value
Level of barium oxide in moraine in the area	>90th perc[a]	4662	82	183	26521	1	(.000*)	1	–
	10th–90th perc	32640	1411	1484	215122	2.12 (1.70, 2.65)	.000	2.30 (1.83, 2.89)	.000
	<10th perc	6566	439	271	39298	3.60 (2.85, 4.58)	.000	3.95 (3.07, 5.09)	.000
Level of manganese oxide in moraine in the area	>90th perc	4885	193	210	30400	1	(.000*)	1	–
	10th–90th perc	31945	1337	1399	202781	1.04 (0.89, 1.21)	.62	1.05 (0.89, 1.24)	.58
	<10th perc	7049	402	329	47760	1.33 (1.12, 1.58)	.001	1.48 (1.21, 1.81)	.000
Level of titanium dioxide in the area	>90th perc	3903	144	155	22482	1	(.017*)	1	–
	10th–90th perc	35928	1602	1635	236994	1.06 (0.89, 1.25)	.54	0.92 (0.77, 1.09)	.33
	<10th perc	4018	186	148	21465	1.35 (1.09, 1.68)	.007	1.25 (0.98, 1.59)	.07
Level of copper in moraine in the area	>90th perc	4826	215	247	35779	1	(.000*)	1	–
	10th–90th perc	32552	1547	1406	203767	1.26 (1.10, 1.46)	.001	1.03 (0.86, 1.24)	.72
	<10th perc	6471	170	286	41395	0.68 (0.56, 0.84)	.000	0.51 (0.40, 0.66)	.000
Level of lead in moraine in the area	>90th perc	4618	171	181	26266	1	(.000*)	1	–
	10th–90th perc	33202	1546	1432	207483	1.15 (0.98, 1.34)	.09	1.29 (1.11, 1.52)	.002
	<10th perc	6029	1215	326	47192	0.70 (0.57, 0.86)	.001	0.64 (0.52, 0.79)	.000
Level of nickel in moraine in the area	>90th perc	5561	269	298	43134	1	(.000*)	1	–
	10th–90th perc	34797	1595	1525	220994	1.16 (1.02, 1.32)	.027	1.23 (1.03, 1.47)	.02
	<10th perc	3491	68	116	16813	0.65 (0.50, 0.85)	.001	0.79 (0.57, 1.10)	.15
Level of zinc in moraine in the area	>90th perc	5570	312	302	43767	1	(.03*)	1	–
	10th–90th perc	32750	1436	1416	205181	0.98 (0.87, 1.11)	.77	0.82 (0.71, 0.94)	.006
	<10th perc	5529	184	221	31993	0.81 (0.67, 0.97)	.021	0.86 (0.69, 1.08)	.19
Level of zinc in brook water plants in the area	>90th perc	6993	491	483	69996	1	–	1	–
	10th–90th perc	33295	1314	1329	192544	0.97 (0.88, 1.08)	.60	1.00 (0.90, 1.11)	.98
	<10th perc	3533	127	126	18305	0.99 (0.81, 1.20)	.91	0.67 (0.54, 0.82)	.000
Level of mercury in brook water plants in the area	>90th perc	765	29	31	4706	1	(.000*)	–	–
	10th–90th perc	8148	369	413	62573	0.96 (0.66, 1.40)	.82	–	–
	<10th perc	952	72	24	3597	3.25 (2.11, 5.01)	.000	–	–

Note. Logistic regression model.
[a]Obs = observed number of children 0–16 years of age at diagnosis; Exp = expected number of children; Perc = percentile.
* = the Chi-squared test for trend (*df* = 1).

A logistic regression model was performed to assess the significance of the effect for each variable (univariate). A regression model also performed included several variables (multivariate). The variables were analyzed in quantile intervals, as described, by the regression model. The number of cases in each quantile interval was used as the dependent variable and the corresponding number of children at risk as offset. SPSS version 15.0 was used for the data analysis.

Ethics

Our study was approved by the research ethics committee at the Faculty of Health Sciences, Linköping University, Linköping, Sweden.

Results

The overall annual incidence during the study period 1977–2006 was 28.84 per 100,000 children 0–16 years of age (10.4 in the community with the lowest incidence and 52.4 in the community with the highest). During the first 25 years of the study period, a constant increase in incidence occurred,

FIGURE **1**

Maps of Sampling Locations in the Study Area

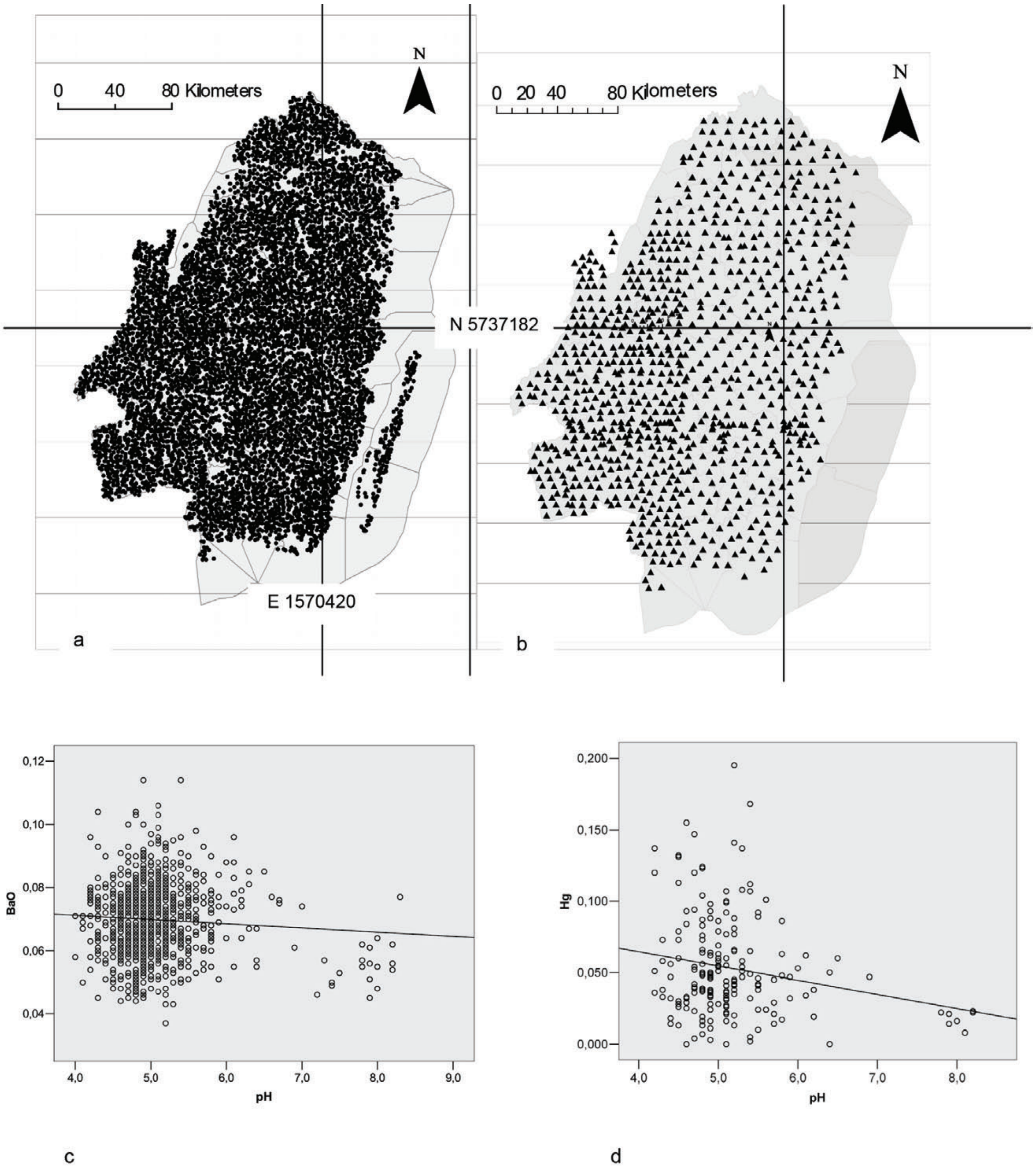


Figure 1a shows sampling locations for geochemical variables measured in brook water plant roots, mostly from *Carex* species and *Filipendula antipyretica* and comprises 6,929 sites. Figure 1b shows the 977 locations of pH measurements. The correlation between pH and different geochemical variables was made after geographical matching. For mercury, 82 analyzed points showed values over detection limit. Figure 1c shows regression of barium oxide over pH, $n = 951$, $R = .07$/Adj, $R^2 = .004$, $p < .05$. Figure 1d shows regression of mercury over pH, $n = 82$, $R = 0.305$/Adj, $R^2 = .082$, $p < .01$.

most pronounced during 1995–2001. From 2002 to 2006 the incidence decreased from 42.54 to 35.07.

Poisson regression modeling, as mentioned earlier in the method section, was performed using different quantile intervals all variables. For models using one variable at a time, three of the elements in Table 1a and one element in Table 1b seemed to be significantly associated with type 1 diabetes at the time of diagnosis. This was most obvious for barium oxide in moraine and mercury in brook water plants. The incidence rate ratio (IRR) was 3.78 (95% confidence interval [*CI*]: 3.04–4.70) in areas with levels of barium oxide in the moraine <10th percentile. Corresponding figures for mercury were 3.08 (95% *CI*: 2.06–4.61) in areas with levels <10th percentile. The other elements showing a significant association were manganese oxide and titanium oxide (Table 2). When analyzed separately (i.e., only male, only females, or a certain age group), males and females showed about the same pattern as the different age groups (0–5 years, 6–10, and 11–16 years of age at diagnosis).

Some of the elements in moraine and in brook water plants at the birthplace of children who later developed type 1 diabetes also differed significantly. The pattern for copper, lead, and nickel was the opposite of that for the oxides, with lower risk among children residing in the areas with levels <10th percentile (Table 3).

When analyzed separately, we found no relationship between the levels of zinc in brook water plants or zinc in moraine and the incidence of diabetes at the time of diagnosis. When zinc was included in the multivariate model with the other elements, the relationship became significant both at birth and at diagnosis. This was most obvious for zinc in brook water plants (IRR 0.67 and 0.78, respectively, in areas with zinc <10th percentile) (Tables 2 and 3). Mercury in brook water plants was not analyzed in this multivariate model as there were too few cases exposed to the sparse sites analyzed for mercury. We also noted in the multivariate analysis that the IRR for diabetes increased in areas with levels of barium oxide <10th percentile, both at diagnosis and at the time of birth. When including mercury in the model (only 554 patients), the IRR for diabetes was 10.5 (95% *CI*: 5.5–19.9.5) in areas with barium oxide <10th percentile at the time of diagnosis and 9.1 (95% *CI*: 4.7–17.6) at the time of birth.

No association occurred between the incidence of diabetes and pH at diagnosis (IRR for pH values <10th percentile 0.98 and 1.02 for pH values between 10th and 90th percentile) or at the time of birth (IRR for pH values <10th percentile 0.92 and 1.03 for pH values between 10th and 90th percentile). A pH value <10th percentile is below 4.5 and a pH value >90th percentile is above 5.7. Low pH values were related both to a higher concentration of barium oxide and to a higher concentration of mercury (Figure 1). A weak positive relationship occurred between barium oxide in moraine and mercury in brook water plants ($R = 0.16$, adjusted $R^2 = 0.024$; $p = .001$; $n = 446$).

When looking at smaller parts of the community areas, a pattern emerged whereby children were diagnosed in areas with the same level of the elements in communities with high incidence of diabetes as well as in communities with low incidence. For instance, both in the community with the highest incidence of diabetes in the region and in the community with the lowest, the children with diabetes were diagnosed in areas with low levels of barium oxide in moraine. Moreover, the areas with the same pattern had a tendency to be adjacent to each other. For instance, barium oxide levels <10th percentile and copper levels >90th percentile shared the same area many times, mostly in areas with high incidence of diabetes.

The trend analysis for the interpolated incidence map showed an increasing incidence to the northeast (Figure 2a). In contrast, the barium oxide trend showed a decreasing trend towards the northeast (Figure 2c). Although the correlation between the variables was not significant, a tendency towards an inverse relationship was seen (Figures 2a and 2c). A certain geographical autocorrelation occurred for barium oxide, which showed a more clustered pattern than the incidence (Figures 2b and 2d) (Moran's Index for barium oxide was 0.57; $p < .01$).

Discussion

To the best of our knowledge, the present study is the first ecological analysis of the risk of type 1 diabetes in relation to minerals in moraine and brook water plants in the neighborhood of residential areas. It must, however, be considered a pilot study and therefore requires confirmation from similar studies of different populations in the future.

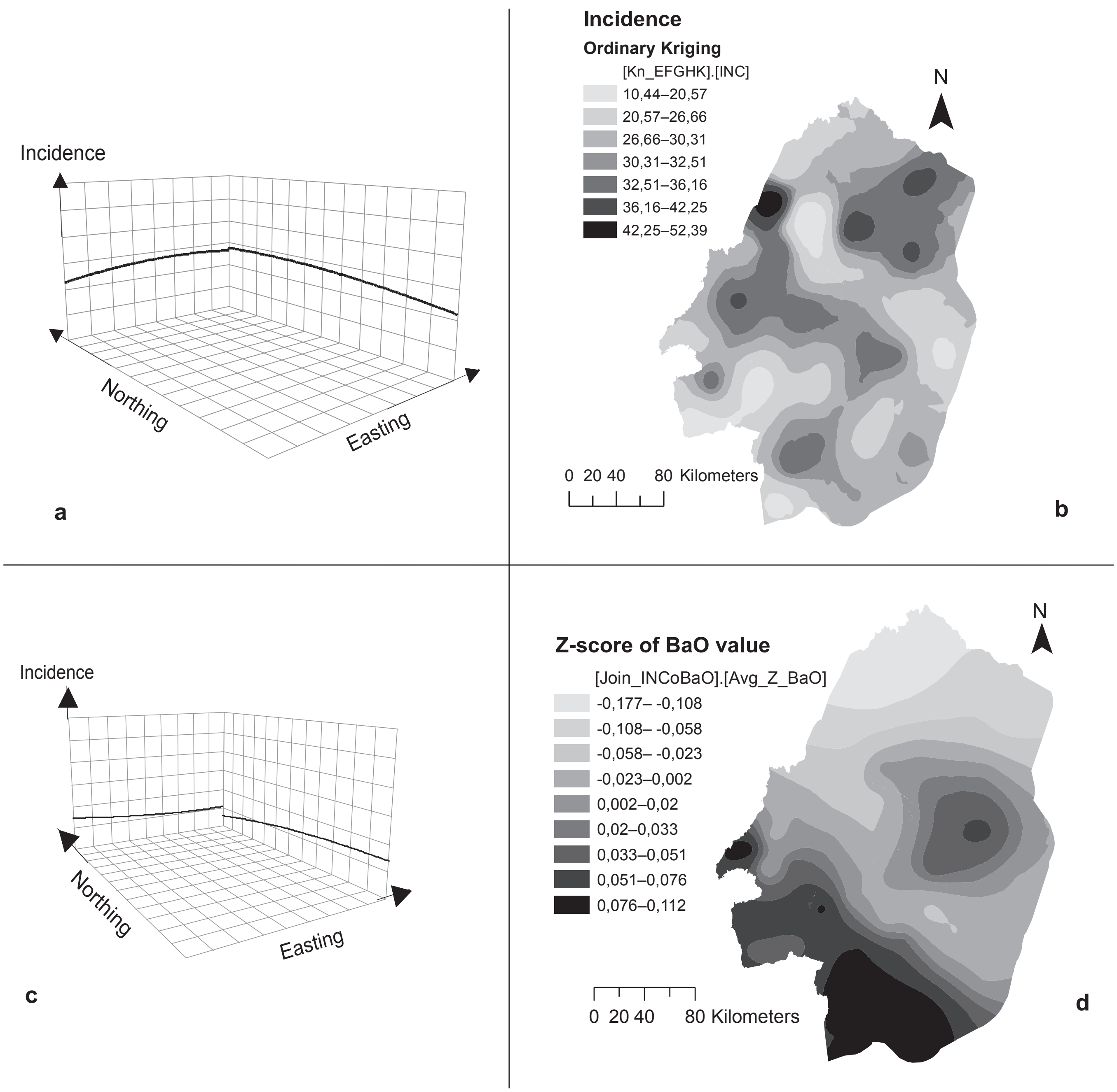
The Swedish medical system provides treatment at a pediatric department for all children with diabetes up to 16–18 years of age. We are, therefore, confident that all children with newly diagnosed diabetes in the area are included. As reported in an earlier study from this region (Samuelsson & Löfman, 2004), no substantial differences in population shifts have occurred between areas in the region. No obvious migration to or from the region has occurred during the study period, nor do there seem to be large differential changes in the number of children in the region over time. As the study area is also a part of a confluent continental area, no support exists for stating the effects of genetic pocket areas.

As an ecological study, one weakness is that exposure to any of the elements/minerals is not estimated at individual level, but measured indirectly by estimates of the minerals in moraine and brook water plants in the area at time of birth and time of diagnosis. Although drinking water is a probable direct or indirect exposure route, these estimates are crude and leave the complex mechanisms of exposure routes to further speculation. Another problem is that we do not know at an individual level the most relevant time window for environmental exposure. To some extent we circumvented this by estimating the relative exposure levels in the residential areas at the time of birth. We also know that about 50% of the children lived at the same place from birth to the time of diagnosis of the children who subsequently developed type 1 diabetes.

It is not easy to understand or to interpret why areas with a very low concentration of barium oxide are associated with a significantly higher incidence of type 1 diabetes than are areas with a high concentration. Barium is a metallic alkaline earth metal, chemically similar to calcium. It is never found in nature in its pure form due to its reactivity with air. Barium carbonate has been used as a rat poison. On the other hand, barium sulfate is used as a radio contrast for X-ray imaging. It is also well known that all water- or acid-soluble barium compounds are extremely poisonous. High doses affect the nervous system with symptoms such as weakness, anxiety, dys-

FIGURE **2**

Relationship Between Diabetes Incidence and Barium Oxide (BaO)



Figures 2a and 2c depict the inverse trend between diabetes incidence and barium oxide (BaO) and Figure 2b shows the corresponding map for the incidence of diabetes. Figure 2d shows the *z*-score of BaO, both variables interpolated by ordinary kriging.

pnea, and paralysis. This may be due to its ability to block potassium ion channels. A literature research targeting the relationship between diabetes and barium was negative, and information on the toxic effects of low dose exposure is generally sparse.

The finding that low levels of mercury in brook water plants in the area, both at birth and at the time of diagnosis, are related to high incidence of diabetes has, as with barium, no support in the literature, although a confounder may explain this covariation. They can be related to other environmental factors that are more connected either to the risk of diabetes or to mercury having a protective effect. The opposite pattern might have been more expected as mercury is a well-known toxic metal that induces oxidative stress. When including both compounds in the same regression model, the risk of diabetes increased three-fold in areas with combined low levels of barium oxide in moraine and mercury in brook water plants.

Some studies have found that low levels of zinc in groundwater (Haglund et al., 1996) or in domestic drinking water (Zhao et al., 2001) may increase the risk of developing type 1 diabetes. Low levels of zinc may not only affect the immune system with increased susceptibility to certain infections such as Coxsackie B viruses (Chandra & Chandra, 1986) but also result in decreased protection against free radicals and cytokines. Free radicals and cytokines have been reported to be important effector molecules in the destruction of beta-cells (Kubish et al., 1994; Mandrup-Poulsen et al., 1990). We were unable to confirm this because we found a lower IRR of diabetes in areas with a low concentration of zinc than in areas with a high concentration. This finding was equal for the time of birth and the time of diagnosis and remained so even after adjustment for the other elements. Zinc may, of course, be a confounder or just a random finding that lacks relevance. Furthermore, possible interactions in the kinetics of various metals may complicate the picture considerably. In our study, zinc had a positive correlation with the minerals studied in moraine, with the exception of barium oxide. Explaining the role of zinc in relation to type 1 diabetes in that way is contradictory as some of the metals are related to high incidence of diabetes in low concentrations (manganese oxide and titanium dioxide) whereas others displayed the opposite pattern (copper, lead, and nickel).

In their study from Norway, Stene and coauthors (2002) reported that acidity in the tap water of individual households may be involved in the etiology of type 1 diabetes. A recent study from Germany (Winkler, Mollenhauer, Hummel, Bonifacio, & Ziegler, 2008) found no such relationship. Stene and co-authors speculated that the pH value in itself was a confounder with no direct relationship to diabetes but was related to certain minerals or microorganisms, which in turn could be involved in the etiology. As our study found no relationship between the risk of developing diabetes and the pH value in the environment around the area of residence at birth or at the time of diagnosis, we can agree with both studies (Stene et al., 2002: Winkler et al., 2008) and conclude that the pH value seems not to be directly involved in the process that leads to type 1 diabetes. The lack of association with pH, however, may result from the mobilization of preventive and toxic compounds at a specific pH level, which might even out the effects of the individual elements. It may also indicate that a confounder is at play that impacts the covariation of mercury and barium oxide, one of which may act as a proxy for the other.

Conclusion

The statistical modeling used in our study found a strong association between type 1 diabetes and low levels of barium oxide in moraine and low levels of mercury in brook water plants in the area of residence, both at the time of birth and at the time of diagnosis. These findings should be interpreted with caution due to the limitations of our study. For instance, no direct mechanism exists indicating the influence of the minerals/elements on type 1 diabetes. The results can, however, be of some importance in generating hypotheses as children tended to be diagnosed in areas with similar concentrations of the minerals/elements in communities with high incidence and those with low incidence of the disease.

Acknowledgements: This study was supported by the research council in southeast Sweden (FORSS). We are grateful to the nurses and doctors in charge of diabetes care at the pediatric departments in Jönköping, Kalmar, Karlskrona, Linköping, Norrköping, Västervik, and Växjö for providing patient data.

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continued on page 154

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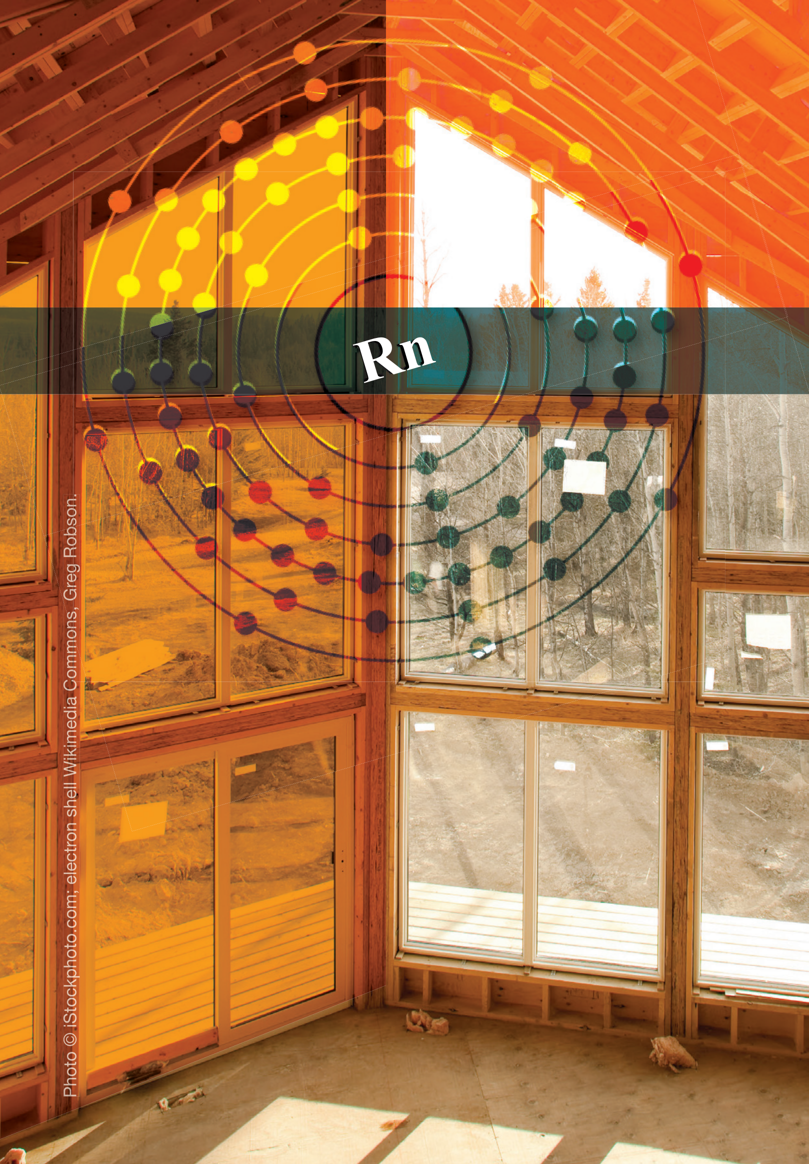

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NEHA Radon Resistant New Construction (RRNC) Training

March 25–27, 2014 ▪ Washington, DC

Are you interested in expanding your knowledge and commitment in radon resistant techniques? If so, then this training opportunity is for you!

The National Environmental Health Association (NEHA), in cooperation with U.S. EPA Indoor Environments Division, is sponsoring a 2½ day **all-expenses-paid** training for environmental health (EH) professionals to implement radon resistant new construction (RRNC). **Attendees are expected to serve as NEHA field partners who will be resources for residential construction activities in their community for a minimum of one year.**

The training includes

- technical information on components of RRNC,
- state and local building code processes, and
- risk assessment and risk communication information about the health effects of long-term exposure to elevated levels of radon gas.

Attendees will

- work with U.S. EPA staff, local code officials and builders, other affiliate partners, nationally recognized instructors, and NEHA field partners—past attendees of this training—who have successfully implemented RRNC in their communities;
- learn new skills to increase consumer awareness of radon hazards, build local coalitions, and collaborate with other stakeholders and nonprofit organizations such as Habitat for Humanity and homebuilder associations; and
- assist in developing an action plan with specific and measurable goals for a RRNC program appropriate for their community.

How to Apply

Please e-mail an application to Marissa Mills at mmills@neha.org by February 28, 2014. Participants will be notified by March 5, 2014, if selected.

Applications must be on agency letterhead and include

- *each* attendee name, position title, complete mailing address, phone, fax, and e-mail address;
- community and/or industry partners that will be attending;
- description of current or planned radon activities including partner organizations;
- description of the area to be served, approximate number of new residential construction building permits in the past year, and the radon zone classification, if known;
- information on previous radon or RRNC training; and
- a statement indicating the support of management to undertake this program.

NEHA strongly encourages joint applications from the same community—teaming public/EH professionals with building code, zoning, or planning department officials, and/or interested builders or homebuilder association representatives.

NATIONAL ENVIRONMENTAL HEALTH ASSOCIATION

For more information, please contact Marissa Mills, Project Assistant, at mmills@neha.org or 303.756.9090, ext. 304.

Dual Home Screening and Tailored Environmental Feedback to Reduce Radon and Secondhand Smoke: An Exploratory Study

Prepublished online December 2013,
National Environmental Health Association.

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Abstract **Combined exposure to secondhand smoke (SHS) and radon increases lung cancer risk 10-fold. The authors assessed the feasibility and impact of a brief home screening and environmental feedback intervention to reduce radon and SHS (Freedom from Radon and Smoking in the Home [FRESH]) and measured perceived risk of lung cancer and synergistic risk perception (SHS x radon). Participants (*N* = 50) received home radon and SHS kits and completed baseline surveys. Test results were shared using an intervention guided by the Teachable Moment Model. Half of the participants completed online surveys two months later. Most (76%) returned the radon test kits; 48% returned SHS kits. Of the returned radon test kits, 26% were >4.0 pCi/L. Of the returned SHS kits, 38% had nicotine >.1 µg/m^3. Of those with high radon, more than half had contacted a mitigation specialist or planned contact. Of those with positive air nicotine, 75% had adopted smoke-free homes. A significant increase occurred in perceived risk for lung cancer and synergistic risk perception after FRESH.**

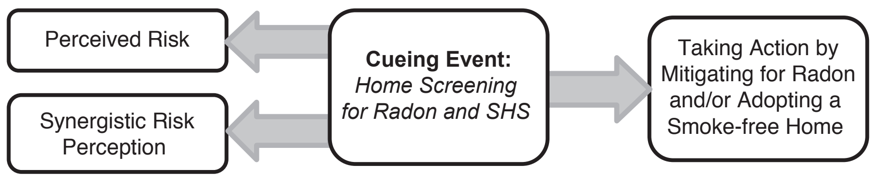
Introduction

Lung cancer is the second most commonly diagnosed cancer and has the highest mortality rate of all cancers (National Cancer Institute, 2007). Radon is a leading preventable cause of lung cancer, second only to smoking (U.S. Department of Health and Human Services [HHS], 2005), and accounts for about 10% of all lung cancer cases each year in the U.S. (Committee on Health Risks of Exposure to Radon [BEIR VI], National Research Council, 1999). The U.S. Environmental Protection Agency (U.S. EPA) estimates that 1 in 15 homes in the U.S. tests positive for radon at or above the U.S. EPA action level of 4 picocuries per liter (pCi/L) (HHS, 2005). Secondhand smoke (SHS) is the third leading preventable cause of lung cancer, responsible for an estimated 3,000 lung cancer deaths per year among nonsmokers (HHS, 2006). The combination of firsthand smoke and SHS and radon exposure increases the risk of lung cancer nearly 10-fold (U.S. Environmental Protection Agency [U.S. EPA], 2004). Interestingly, households with smokers are less likely to test for radon than those without smokers (Centers for Disease Control and Prevention, 1999), despite the fact that smokers are at greater risk for developing lung cancer from radon exposure than never-smokers (Bonner et al., 2006; Lagarde et al., 2001). Although the perceived risk from smoking and SHS is high (Rayens et al., 2007), less is known about the combined perceived risks of these two environmental exposures (Hampson, Andrews, Lee, Lichtenstein, & Barckley, 2000).

Our exploratory study aims were to 1) describe the feasibility of a brief, tailored dual home screening and environmental feedback intervention (FRESH; Freedom from Radon Exposure and Smoking in the Home) to assess and reduce radon and SHS exposure; 2) examine changes in risk perception and health actions following FRESH; and 3) assess differences in parents'

FIGURE 1

Study Model Adapted From Teachable Moments Heuristic



SHS = secondhand smoke.

perceived risk of lung cancer and synergistic risk perception (SHS x radon) between those with and without home smoking at baseline. First, we hypothesized that parents would have greater perceived risk of lung cancer and synergistic risk (SHS x radon) postintervention. Second, those with home smoking would be less likely to take action to reduce exposure to radon and SHS than those without smoker(s)/smoking in the home. Third, we hypothesized that home smoking would be associated with perceived risk of lung cancer and synergistic risk at baseline.

Exposure to radon, a colorless, odorless, radioactive gas that forms from the decomposition of uranium in the ground, is associated with an estimated 15,400 to 21,800 lung cancer cases in the U.S. each year (Committee on Health Risks of Exposure to Radon [BEIR VI], National Research Council, 1999). Radon can be detected with a simple, commercially available test; however, the proportion of people who have tested is low (Wang, Ju, Stark, & Teresi, 2000). If high levels of radon are found, properly installed mitigation systems reduce the risk of exposure.

SHS, a major source of indoor air pollution, causes coronary heart disease, lung cancer, asthma, and sudden infant death syndrome (HHS, 2010). The home is the primary source of SHS for children and a major source of SHS exposure for adults (HHS, 2006).

If exposed to 4 pCi/L of radon over a lifetime, 7 per 1,000 of those who never smoked would develop lung cancer compared to 62 of 1,000 smokers who would develop lung cancer (U.S. EPA, 2004). Despite the synergistic risk of radon and tobacco smoke, smokers are less likely than nonsmokers to engage in protective radon behaviors (Hampson, Andrews, Barckley, Lichtenstein, & Lee, 2006).

Our study was guided by the Teachable Moment Model (McBride, Emmons, & Lipkus, 2003), positing that a health event (i.e., feedback on home radon and SHS levels) serves as a cue to perceive a health threat that can motivate an individual to reduce the threat (McBride et al., 2003; Figure 1). Using tailored environmental feedback may be one way to create teachable moments by generating heightened receptivity to behavior change (Lawson & Flocke, 2009).

Methods

Design and Sample

The exploratory study was a quasi-experimental, pre-post design with quota sampling to ensure that 60% of participants would be in the home smoking group. The participants were parents or primary caregivers of children recruited from a pediatric practice during an office visit. To be eligible for the study, participants had to own their homes, have access to a telephone, and never have tested for radon. Once the quota of those without smokers in the home was reached, participants were eligible only if they had a smoker in the home. Participants completed the baseline survey on a laptop computer while in the physician's office at the time of recruitment. After completing the baseline survey, participants were asked to colocate a radon and an airborne nicotine sampler in their homes for 72 hours. The two-month postintervention surveys were completed online. The study was approved by the University of Kentucky Medical institutional review board.

Intervention

The brief, tailored environmental feedback intervention, FRESH, was designed to create a teachable moment for lung cancer risk reduction by motivating participants to simultaneously test their homes for radon and SHS and delivering a tailored environmental feedback intervention. After receiving the test results (4–8 weeks posttesting), trained research staff with educational preparation in nursing and health communications delivered FRESH via telephone and mail. The intervention was tailored based on one of four conditions: high radon/high SHS, high radon/low SHS, low radon/high SHS, low radon/low SHS. Using the FRESH action checklist as a guide, the research team discussed the participant's risk based on the screening results and suggested actions to lower their risks such as adopting a smoke-free home, radon mitigation, and quitting smoking. Following the phone call, the participant received a mailed packet with a copy of the checklist and general educational materials on radon, SHS, and quitting tobacco. If participants did not respond to requests for telephone intervention delivery, test results, tailored feedback, and materials were mailed to the home.

Measures

Demographic characteristics and smoking indicators were assessed in the baseline questionnaire; perceived risk of lung cancer and synergistic risk were measured in both surveys. Self-reported health actions following the intervention were assessed at two months. Demographic characteristics assessed were age, sex, race, marital status, education, employment status, and household income. Study measures are summarized in Table 1.

Data Analysis

Descriptive statistics, including means and standard deviations or frequency distributions, were used to summarize study variables. Chi-square tests of association determined differences in categorical variables between two groups, including smokers versus nonsmokers, home smoking versus home nonsmoking, and those who completed the study versus those who dropped out prior to the follow-up survey. Differences in perception of risks from pre- to postintervention were determined using paired *t*-tests. Two-sample *t*-tests were used to compare perceived risk of lung cancer and synergistic risk perception at baseline between home smoking and home nonsmoking groups. Data analysis was conducted using SAS version 9.3 with an alpha level of .05 throughout.

Results

The average age of the participants was 37.8 (*SD* = 7.9). The majority were female, non-Hispanic white, and married (Table 2). Most had at least some postsecondary education, were employed, and had an average annual household income of $50,000 or more. One-fifth were current smokers and three-fifths had home smoking. Nearly half (48%) of the participants had one to three smokers living in the home, and 50% reported their family was exposed to tobacco smoke while at home. Of those with home exposure to tobacco smoke, 56% were exposed all seven days of the past week. Of the 38 participants who returned readable radon test kits, 10 (26%) tested at or above the U.S. EPA action level of 4 pCi/L. The mean radon level for all 38 returned kits was 2.9 (*SD* = 2.4). Of the 24 families who returned a readable airborne nicotine test kit, 9 (38%) tested at or above .1 μg/m^3. The mean airborne nicotine level for all 24 readable test kits was 0.13 (*SD* = 0.14).

TABLE **1**

Study Measures

Variable	Measures
Smoking indicators	Current smoking: Do you currently smoke cigarettes, even just once in a while? *Yes/No* Home smoking: Of those people living in your home including yourself, how many people smoke (e.g., cigarettes, pipes, cigars, etc.)? and Thinking about the past seven days altogether, how many days was your family exposed to tobacco smoke at home? *>0 for either indicates positive for home smoking*
Perceived risk	Perceived risk of lung cancer: How would you rate your risk of developing lung cancer in your lifetime, on a scale of 1 to 10 where 1 is the lowest and 10 is the highest risk? Synergistic risk: Please rate the risk from being exposed to radon and smoking a pack of cigarettes per day, compared to the risk of only smoking a pack of cigarettes a day with no radon exposure. *1–10 scale with higher score equivalent to more risk*
Radon and air nicotine	Home radon level: Short-term radon test kit by AirChek. Scores at or above 4.0 pCi/L considered high for radon (U.S. Environmental Protection Agency [U.S. EPA], 2004). Secondhand smoke (SHS) level: Passive airborne nicotine monitor developed by Lee and co-authors (1992, 1995). Values >.1 μg/m^3 positive for SHS.
Health actions	Home mitigation: Have you mitigated your home? *Yes/I plan to mitigate my home, but haven't done so yet* Steps toward mitigation: In the last 60 days, I contacted a mitigation professional to assess my house for mitigation. *Have already completed the action/planned to take action, but have not done so yet* categorized as taking steps toward mitigation; *Didn't plan this to take this action* is the inactive category. Rules about smoking in the home: What best describes the rules about smoking in your home? *Smoking is not allowed at all* categorized as smoke-free; *Smoking allowed in all areas of home/smoking is limited to certain areas/ smoking is not allowed when children are inside the home* all considered not having adopted a smoke-free home.

Feasibility of FRESH

Of the 58 parents/caregivers approached, 50 agreed to participate and they completed the baseline survey (86% response rate). Of those, 38 returned readable radon test kits (76%) while only 24 sent back readable air nicotine kits (48%). Twenty-five of those who completed the baseline survey participated in the two-month follow-up survey (50% retention rate). No significant differences existed in demographic characteristics or smoking indicators, including smoking status and home smoking, between those who participated in the follow-up and those who did not complete the two-month survey. Participation in the follow-up survey was just as likely for those with elevated radon (≥4 pCi/L) and positive airborne nicotine (≥.1 μg/m^3) as those without. Similarly, no differences existed between smokers and nonsmokers or those with and without home smoking on whether they returned a readable radon or airborne nicotine test kit. Of those who participated in the environmental feedback intervention, most were in the low radon/low SHS condition (*n* = 21); nine were in the high radon/low SHS; six were in the low radon/high SHS; and one was in high radon/high SHS.

Impact of FRESH on Perceived Risk of Lung Cancer, Synergistic Risk, and Health Actions

Participants reported a significant increase in perceived risk of lung cancer and synergistic risk perception from baseline to postintervention (mean = 0.64; $t = 2.1$, $p = .05$; mean = 1.48; $t = 2.5$, $p = .02$, respectively). No differences in change in perceived risk of lung cancer or synergistic risk perception occurred between those with and without home smoking. Of the 10 households with radon at or above the U.S. EPA action level of 4 pCi/L, seven participated in the follow-up, including three smoking and four nonsmoking; of these, none had mitigated, but four (57%) had taken steps toward mitigation by either contacting a mitigation professional or planning to do so. Of these, half were in the home smoking and half in the home nonsmoking groups, suggesting that 67% of smoking homes participating in the follow-up had taken steps to mitigate and 50% of nonsmoking homes had done so.

Of the nine families with elevated airborne nicotine, six were in the home smoking group and three identified their households as nonsmoking. Four of the nine participated in the follow-up; of these, three were smoking homes and one nonsmoking. Three of the four (75%) had made their home smoke free by follow-up, including the nonsmoking home and two of the three smoking ones; this suggests that 67% of those in smoking homes and 100% in nonsmoking homes that participated in the follow-up and had elevated air nicotine reported having made their homes smoke free.

TABLE 2

Frequency Distributions of Demographic Characteristics and Smoking Indicators (N = 50)

Variable	*n* (%)
Sex	
Female	44 (88)
Male	6 (12)
Race/ethnicity	
White, non-Hispanic	48 (96)
Other race/ethnicity	2 (4)
Married	
Yes	46 (92)
No	4 (8)
Some postsecondary education	
Yes	43 (86)
No	7 (14)
Employed	
Yes	37 (74)
No	13 (26)
Annual income $50,000 or more	
Yes	43 (86)
No	7 (14)
Current smoker	
Yes	10 (20)
No	40 (80)
Smoking in the home	
Yes	30 (60)
No	20 (40)

TABLE 3

Comparison of Baseline Perception of Lung Cancer Risk and Synergistic Risk Perception Between Those With and Without Smoking in the Home (N = 50)

Perception of Risk	Household Smoking		$\lvert t \rvert$ (p-Value)
	Home Smoking (n = 30) Mean (SD)	No Home Smoking (n = 20) Mean (SD)	
Lifetime risk of lung cancer (1 = lowest risk; 10 = highest risk)	4.67 (2.14)	2.71 (2.20)	3.0 (.005)
Synergistic risk of radon exposure *and* smoking compared to smoking alone (1 = much less risky; 10 = much more risky)	5.90 (2.44)	7.17 (2.31)	1.8 (.08)

Differences in Risk Perceptions by Smoking Status at Baseline

Those with home smoking reported a higher mean perceived lifetime risk of lung cancer than those without home smoking at baseline (Table 3). No difference occurred, however, in perceived synergistic risk of tobacco smoke and radon between those with and without home smoking.

Discussion

FRESH is a feasible intervention that holds promise in promoting perceived risk of lung cancer and synergistic risk perception and prompting action to reduce risk. Recruiting in a pediatrician's office with a supportive health care provider champion facilitated high study participation. Participants were receptive to messages emphasizing children's health and safety, and the health care recruitment setting highlighted the seriousness of the problem. Interestingly, testing for radon was more prevalent than testing for SHS exposure. The provider champion had tested his home for radon and that could have influenced participants to focus more on radon testing. This testing discrepancy may also have been due to societal stigma related to smoking (Graham, 2012) or the denial of the need for the test (i.e., smoking outside or not having a smoker living in the home). Several participants with positive air nicotine were surprised given that their homes were "smoke free." They didn't realize the importance of smoking away from entryways, windows, and vents. It is encouraging that parents who tested their homes were just as likely to return the radon and airborne nicotine test kits regardless of whether smokers were in the home.

Parents reported greater perceived risk of lung cancer and synergistic risk (SHS x radon) postintervention. It is promising that parents in homes with smokers reported a significant change in perceived risk and synergistic risk, especially since they reported lower synergistic risk at baseline compared to those without smoking in the home. Given that few view radon as an immediate health risk in their home or understand their risk status (Duckworth, Frank-Stromborg, Oleckno, Duffy, & Burns, 2002; Hill, Butterfield, & Larsson, 2006), these exploratory findings show promise in creating a teachable moment on perceived risk and synergistic risk (SHS x radon) through tailored environmental feedback, especially with those who have smokers in the home and are at most risk for lung cancer.

Parents who were provided evidence of elevated radon levels were likely to mitigate or plan to mitigate, consistent with the literature (Duckworth et al., 2002; Riesenfeld et al., 2007). Those with at least one smoker in the home were as likely to take action to reduce exposure to radon and SHS compared to those without smoker(s) in the home postintervention, albeit based on a small sample size. Given that smokers are least likely to demonstrate protective radon behaviors (Hampson et al., 2006), the results of our exploratory study are encouraging.

Consistent with the literature (Hampson et al., 2006), those with household smoking were less likely to perceive synergistic risk (SHS x radon) but were more likely to perceive lung cancer risk than those without smokers in the home. Participants with household smoking may have already believed that they were susceptible to lung cancer, so the addition of radon risk may not have significantly changed that perception.

Our exploratory study had several limitations. First, the small sample of relatively high-income participants reduces generalizability of the findings. Requiring home ownership to participate minimized the chance of recruiting lower socioeconomic status parents. The low retention rate is another limitation. Only half of the participants enrolled participated in the follow-up survey. Future studies need to include escalating payments and other ways of reducing attrition (Seed, Juarez, & Alnatour, 2009) (e.g., use of electronic and social media reminders). Another limitation was that participants were less likely to return the SHS kits. For some families, this may have been due to societal stigma associated with smoking, while others may have felt it unnecessary since no one smoked in their home. While no differences existed in demographics or smoking indicators between those who completed the follow-up and those who did not and those who returned readable test kits and those who did not, the sample sizes in the subgroups were small in our exploratory study.

Future studies need to include a larger number of households for increased statistical power. No test kit is commercially available for both radon and airborne nicotine, and participants had separate instructions for testing and returning each test kit, which may have caused confusion. Further, participants had no incentive to return either test kit. The social implications of reporting home SHS exposure may have made some participants underreport smoking in the home, as evidenced by three positive air nicotine tests from among the nonsmoking group. This concern is lessened, however, by the fact that a group effect existed for perceived risk of lung cancer at baseline. Radon and SHS levels were not reassessed at follow-up, nor was long-term follow-up possible due to budgetary constraints. Further, we did not assess the fidelity of the actual location of the radon and SHS samplers in the homes. Environmental measurements at postintervention and long-term follow-up are recommended for future testing of FRESH.

Conclusion

FRESH deserves further study as it has the potential for preventing lung cancer especially among nonsmokers. Given that smokers also exposed to radon have higher rates of lung cancer (Committee on Health Risks of Exposure to Radon [BEIR VI], National Research Council, 1999; U.S. EPA, 2004), FRESH may also reduce lung cancer among smokers.

Acknowledgements: We appreciate the assistance and support of the Kentucky Department for Public Health Radon Program and the Kentucky Association of Radon Professionals. Funding for this work was from University of Kentucky Markey Cancer Center, University of Kentucky College of Nursing, and University of Kentucky Got Grants Program. The authors were also funded through a contract with the Kentucky Department for Public Health Radon Program (E. Hahn, PI) for radon awareness and community-based research.

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INTERNATIONAL PERSPECTIVES

Prepublished online December 2013, National Environmental Health Association.

Urinary Metabolites of DEET After Dermal Application on Child and Adult Subjects

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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Abstract **Urinary metabolites of DEET of 17 children (5–7 years of age) and 9 adults (23–25 years of age) were examined in the study described in this article. Urine samples were collected from each subject within eight hours after a single dermal application of 10 mL 12% DEET-containing insect repellent. Two metabolites, m-diethylaminocarbonyl benzoic acid (R3N0) and N-ethyl-m-toluamide (R0N1), with unchanged DEET, were identified in the urine. The major metabolite was R3N0, which was 78.2% and 46.1% of the total DEET metabolites from children and adults, respectively, indicating that the pathway of ring methyl oxidation predominated. The recovered DEET metabolites were observed significantly more from children (1,116 µg) than from adults (446.2 µg) (p < .001). The difference in dermal absorption, albeit primarily attributed to DEET loading, was found to be related to height by regression analysis. The inverse association between height and dermal absorption of DEET suggests that shorter individuals (i.e., children) are subjected to dermal uptake of DEET. To avoid unnecessary exposure, parents need to be cautious when applying DEET-containing insect repellent on children.**

Introduction

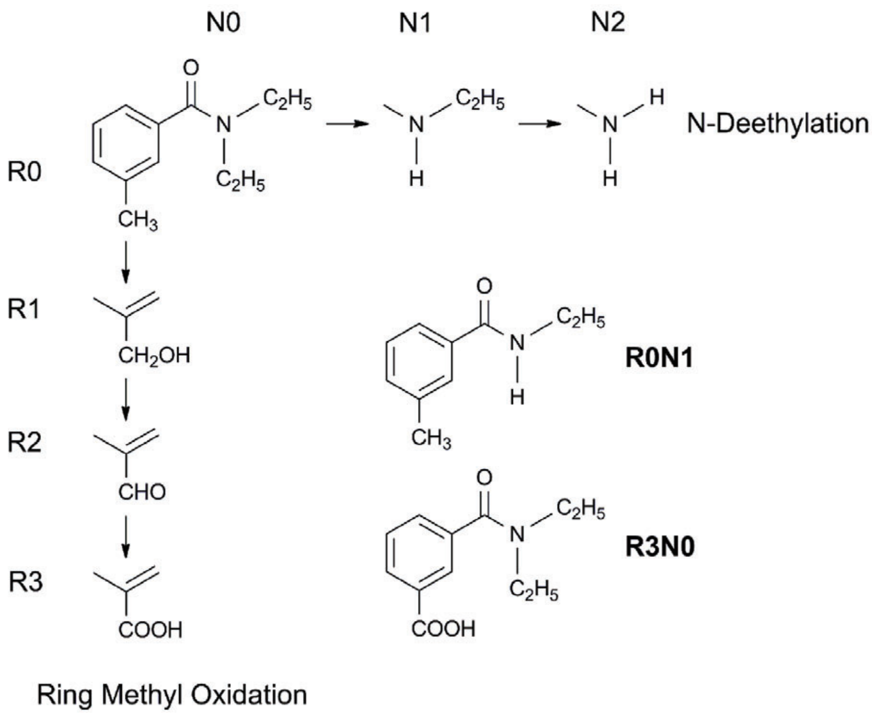
DEET, an effective ingredient of insect repellent, is widely used to prevent mosquito bites (Fradin & Day, 2002). High DEET concentrations in insect repellent increase the effectiveness (Qiu, Jun, & McCall, 1998), but also raise health concerns (New York Department of Environmental Conservation, 1994). Reported adverse effects of DEET include seizures (Koren, Matsui, & Bailey, 2003) and encephalopathy in children (Briassoulis, Narlioglou, & Hatzis, 2001) and hypotension, coma, and seizures in adults (Teinenbein, 1988).

To avoid these adverse effects, the U.S. Environmental Protection Agency (U.S. EPA) recommended less than 10% of DEET in insect repellent for use on children (U.S. EPA, 1998), but the recommendation did not apply to those under six months (Bell, Veltri, & Page, 2002). A newer report, however, indicated that applications of 10%–30% DEET-containing insect repellent rarely caused adverse effects (Flake, Hinojosa, Brown, & Crawford, 2005), and insect repellent of less than 30% DEET was considered safe in a clinical report (Miller, 2004).

To assess the adverse effects, an understanding of the metabolism of DEET in the human body is crucial; thus, a number of previous studies have identified DEET metabolites. A previous study had proposed a 3 × 4 scheme of metabolic pathways of DEET in rat liver microsomes (Constantino & Iley, 1999). The pathways are theoretically driven by reactions of N-deethylation (horizontal) and ring methyl oxidation (vertical) (Figure 1). In another study, m-(Diethylaminocarbonyl) benzoic acid (R3N0), m-(ethylaminocarbonyl) benzoic acid (R3N1), and m-(aminocarbonyl) benzoic acid (R3N2) were found in rat urine (Taylor & Spooner, 1990); and N-ethyl-m-toluamide (R0N1), toluamide (R0N2), N,N-diethyl-m-(hydroxymethyl)benzamide

FIGURE **1**

Pathways of DEET Metabolism



(R1N0), N-ethyl-m-(hydroxymethyl)benzamide (R1N1), and N,N-diethyl-m-formylbenzamide (R2N0) were identified in rat liver microsomes (Taylor, 1986). Therefore, DEET in animals is metabolized consistently following the two metabolic reactions.

In humans, DEET is generally absorbed by the skin and mainly excreted in the urine (Selim, Hartnagel, Osimitz, Gabriel, & Schoenig, 1995; Smallwood, DeBord, & Lowry, 1992); it has been confirmed that DEET in the human body is metabolized via liver oxidase from an in vitro study (Abu-Qare & Abou-Donia, 2008). It is reported that liver microsomes induced cytochromes P450 to metabolize DEET to R0N1 (CYP2A6, 2C19, 3A4, and 3A5) and R1N0 (CYP1A2, 2B6, 2D6*1, and 2E1) (Usmani et al., 2002), the initial DEET metabolites. A human study observed six DEET urinary metabolites, two of which were confirmed to be R3N0 and R3N1, but did not detect unchanged DEET from the urine (Selim et al., 1995).

The metabolism of DEET has been studied in animals or humans in many western countries, but is rarely addressed in Asia; yet it is not clear whether ethnic difference affects the metabolism as well as dermal absorption of DEET. To our knowledge, ours is the first study to identify and confirm DEET metabolites from human urine in Asia. We demonstrate profiles of DEET metabolites in the urine of children and adults after one-time dermal applications of DEET-containing insect repellent, and expect that the human subject testing information could be useful to biomonitoring in future related studies.

Materials and Methods

Human Subjects

Our study was approved by the Tzu-Chi Hospital/University institutional review board in Taiwan. All experiments were performed in compliance with the relevant laws and institutional guidelines. Researchers had to inform the participating subjects or parents/guardians of the child subjects of safety and risk associated with and processes of the study and the right of withdrawal from the study. An informed consent, signed by each subject or his/her parent/guardian, was obtained prior to the test. Seventeen children (5–7 years of age) from Tzu Chi University–affiliated kindergarten and nine adults (23–25 years of age) from Tzu Chi University were recruited. Each of them was confirmed not to use any DEET-containing insect repellent for the previous 10 days to avoid possible interference.

Materials

DEET (95.0%) and precursors of the synthesized DEET metabolites were purchased. The syntheses of DEET metabolites were conducted following a previous study (Constantino & Iley, 1999). Acetonitrile (high performance liquid chromatography [HPLC] grade) and hydrochloric acid (36.5%–38.0%) were used. Deionized water was generated by Millipore Direct-Q.

Sample Collection

Prior to the dermal application, a baseline urine sample was collected from each subject. Ten mL of insect repellent containing 12% DEET was applied onto bare skin of the arms and legs of each subject, and urine was collected within the following eight hours. For each urine collection, the urination time and the volume of urine were recorded in a pamphlet where the relevant information (e.g., birth date, age, sex, height, weight, surface area of application) was also kept. Records of urinary volume were important, because they were used for compositing and calculating to the quantities of metabolites. Child subjects were assisted by the kindergarten teachers, while adult subjects completed the tests by themselves following the testing protocol. All the urine samples were stored at -20°C before analysis.

Sample Analysis

After defrosting the urine, we composited samples collected within the first and second four hours by mixing 1% of the volume of each urine sample with one another. Composited samples were acidified to pH 2–3 with 1 N HCl, and 5 mL of each was drawn into a conditioned Sep-Pak Vac 6 cc (500 mg) C18 cartridge for solid phase extraction. We added 5 mL deionized water (pH 2–3) to wash the cartridge, and then 8 mL acetonitrile to elute the metabolites. The eluate was concentrated to 0.5 mL with nitrogen for analysis of HPLC. The recovery rates of the C18 cartridges were found to range from 88% to 97% for DEET and its metabolites.

We identified urinary metabolites of DEET with mass spectroscopy (MS) using Agilent 1100 HPLC-MS and analyzed them quantitatively using Agilent Series 1050 HPLC. The column was Eclipse XDB-C18, 4.6×250 mm, 5 µm and two-stage solvent gradients for mobile phase were used at a constant flow rate of 1 mL/min: 50% acetonitrile and 50% water (pH 2–3) for the first eight minutes, and 100% acetonitrile for the following seven minutes. For quantitative analysis, DEET and its metabolites were detected at the wavelength of 230 nm.

Calculation and Statistical Analysis

In order to calculate the permeability of DEET, the amounts of metabolites, according to the molar numbers, were converted back to those of the parental DEET, which were denoted as equivalent DEET. The conversion formula is listed below:

$$\textit{Equivalent DEET} = \sum_{i=0}^{3}\sum_{j=0}^{2}\frac{W_{RiNj}}{M.W._{RiNj}} \times M.W._{DEET}$$

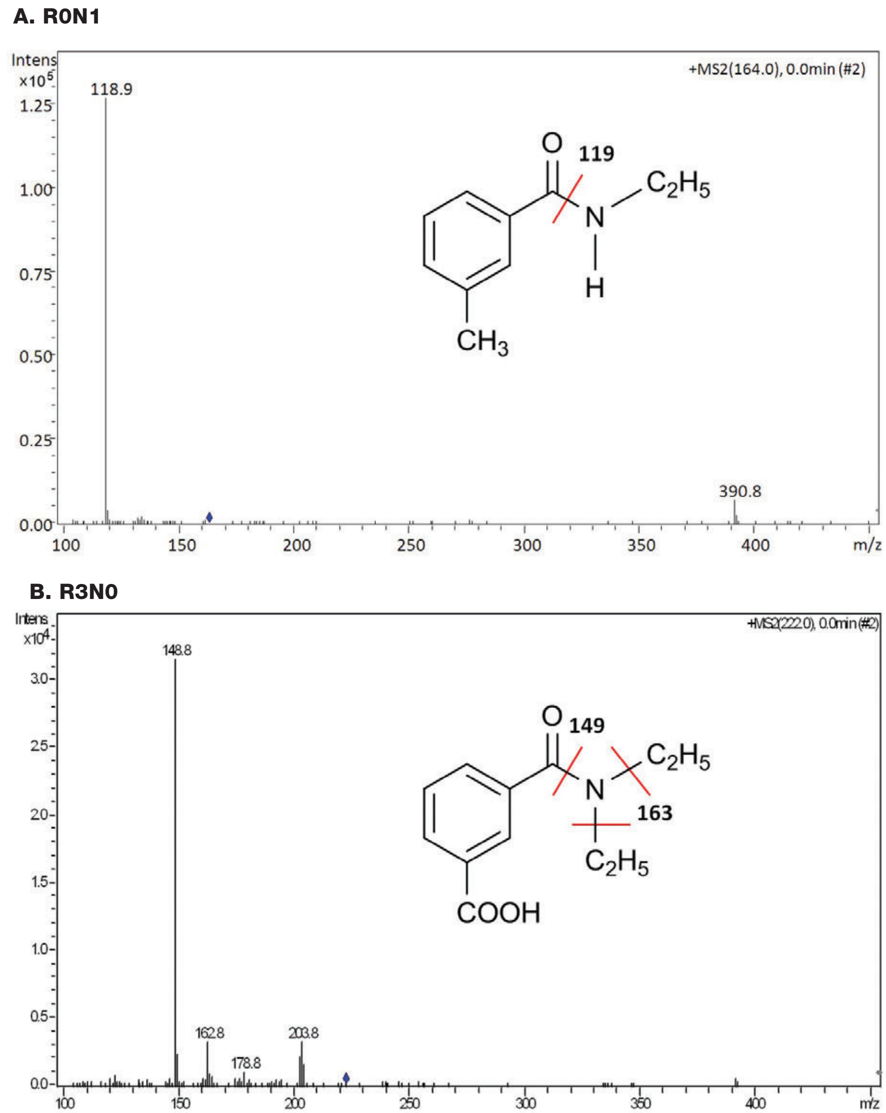
W_{RiNj} = amount (µg) of metabolite $RiNj$, $i = 0, 1, 2, 3$; $j = 0, 1, 2$

$M.W._{RiNj}$ = molecular weight of metabolite $RiNj$, $i = 0, 1, 2, 3$; $j = 0, 1, 2$

$M.W._{DEET}$ = molecular weight of DEET, 191.27 g/mol

FIGURE **2**

Mass Spectra of R0N1 and R3N0



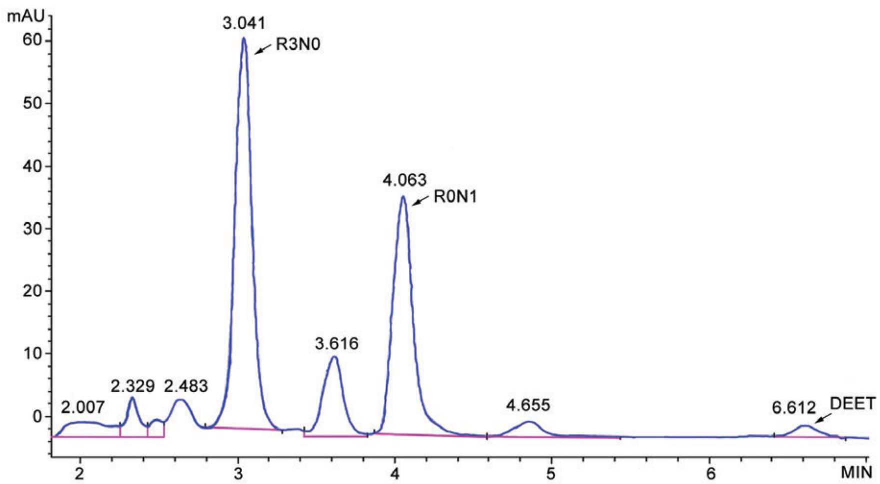
In accordance with Fick's first law of diffusion, a steady-state flux (J) of percutaneous absorption (µg/cm²/s) equals the product of permeability (P) (cm/s) and drug concentration (C) (µg/cm³), giving the formula as

$$J = PC.$$

The DEET concentration was 12% (weight/volume), but, as the solvent quickly evaporated after the dermal application, the true concentration (C) remaining on the skin should be proportional to DEET loading (mg/cm²), which was calculated by dividing the applied amount of DEET (mg) by the application area of skin (cm²). The steady-state flux (J) was calculated by dividing the quantity of urinary metabolite (e.g., R0N1, R3N0) by the application area (cm²) and time (s). In theory, permeability (P) remains the same for the applied DEET onto the presumably identical skins, and thus J is proportional to C or DEET loading. In our study we conducted independent *t*-tests to compare the two age groups; Pearson correlation analysis to examine any possible relevant factors to J of the total metabolites (J_E_DEET), including sex, weight, height, application area, age group (child/adult), and DEET loading; and stepwise linear regression analysis to determine the most affecting factors on the dermal absorption of DEET. All statistical analyses were performed using SPSS 16.0.

FIGURE 3

Chromatogram of DEET Metabolites in Children's Urine



Results

We detected R3N0, R0N1, and unchanged DEET from most of the urine samples after the dermal application of insect repellent using HPLC-MS, with the fragment profiles of the detected compounds matching that of the known (synthesized R0N1, R3N0 and commercially manufactured DEET). The M+1 values were 164 and 222 m/z for R0N1 and R3N0, respectively, with accountable fragments for both compounds (Figure 2). None of the metabolites or DEET was detected from the baseline samples, indicating that the detected urinary compounds were most likely to originate from DEET. Having confirmed the metabolites using HPLC-MS, we used regular HPLC for further qualification and quantification. The retention time for R3N0, R0N1, and DEET was 3.0, 4.1, and 6.6 min, respectively (Figure 3).

Because of different bare skin areas exposed to the insect repellent application, the DEET loading were approximately above and below 1 mg/cm^2 for child and adult subjects, respectively (Table 1). Male subjects, with more height and weight, had more exposed surface areas than females did for either age group. Among the all urinary metabolites, R3N0 was the most for both age groups. For child subjects, R3N0 was approximately 80% of the total equivalent DEET; for adults, R3N0 accounted for nearly half of the urinary metabolites (Table 1). Apparently, the recovered DEET metabolites, as converted to the equivalent DEET, were observed more from children (1,116 µg) than from adults (446.2 µg), and the difference was statistically significant ($p < .001$). No significant difference occurred in total metabolites between the male and female subjects of either group, indicating that gender was not an affecting factor of the DEET metabolism.

Pearson analysis results show that sex was not significantly related to any other variables, but weight, height, application area, and age group were highly correlated with one another (Table 2). DEET loading was correlated with most variables, especially with J of R3N0 (J_R3N0) ($r = .86$) and J of the total equivalent DEET (J_E_DEET) ($r = .85$), but not with J of unchanged DEET (J_DEET), indicating that Fick's first law of diffusion was followed and the majority of DEET was metabolized to R3N0 with the amount of unchanged DEET varying by chance. The correlation coefficient between J_E_DEET and J_R3N0 was as high as 0.95, in support of the finding that R3N0 was the major metabolite.

We conducted stepwise multivariate regression analysis to determine significantly affecting factors of recovered DEET metabolites, with J_E_DEET used as the dependent variable and others as the independent variables. Following Fick's first law of diffusion, DEET loading was the predominant factor, and the other significant factor added to the equation was height, which was inversely associated with the permeated DEET (Table 3). The inclusion of height suggests that transdermal permeation of DEET is governed by body stature or possibly age, besides the known principle of diffusion.

Discussion

As compared to previous studies, our work used high doses of DEET on participating subjects because of the consideration of insect bites occurring commonly in the subtropical climate of Taiwan. In a previous child study without evaluating the application dose of DEET, a median concentration of 7×10^{-4} µg/mL was reported from the urine of children of ages 1–6 (Arcury et al., 2007); in our study, the equivalent DEET was recovered to be 1.3 µg/mL (median) in children's urine. As for adults, eight employees of an American national park were applied with one gram of pure DEET in a 71% insect repellent on the skins and clothes per day, and urinary concentrations of DEET ranging from nondetection to 5.69 µg/mL were recovered for 24 hours; other metabolites, however, were not determined (Smallwood et al., 1992).

Our study, with a similar application amount, resulted in the median equivalent DEET concentration of 1.68 µg/mL, which fell in the low end of the range derived by Smallwood and co-authors (1992). Considering that our collection duration was eight hours, one-third of theirs, we think that our urinary data were fairly consistent with that of the previous report.

The major DEET metabolite, R3N0, was also found in human urine of the study conducted by Selim and co-authors (1995), which did not detect any unchanged DEET. The difference in detecting unchanged DEET between our study and theirs may have been due to the applying doses, which were approximately 80 times higher in our study. Other studies, referred to in the

TABLE **1**

Basic Information and DEET Urinary Metabolite Result of Child and Adult Subjects

Demographic		Basic Information					Metabolite Result			
		Weight (kg)	Height (cm)	SAA[a] (cm^2)	DEET loading (mg/cm^2)		R3N0 (µg)	R0N1 (µg)	Unchanged DEET (µg)	Total Equivalent DEET (µg)
Child										
Female ($n = 7$)	Mean	20.7	115.3	958.4	2.36	*GM*[a]	872.4	187.2	50.3	1081
	SD[a]	2.2	3.0	650	2.0	*GSD*[a]	1.6	1.8	3.1	1.5
Male ($n = 10$)	Mean	23.1	118.1	1116	1.17	*GM*	873.2	175.9	91.1	1141
	SD	2.8	4.5	424	1.0	*GSD*	1.6	3.6	2.7	1.8
All ($n = 17$)	Mean	22.1	116.7	1002	1.83	*GM*	872.9	180.5	71.3	1116
	SD	2.8	4.0	511	1.5	*GSD*	1.6	2.8	2.9	1.7
Adult										
Female ($n = 4$)	Mean	48.5	156.3	2061	0.64	*GM*	267.4	117.0	100.8	472.9
	SD	6.0	2.6	626	0.3	*GSD*	1.5	1.6	1.6	1.5
Male ($n = 5$)	Mean	67.6	168.6	3401	0.37	*GM*	166.5	99.2	155.4	426.0
	SD	10.5	7.6	768	0.1	*GSD*	1.9	2.2	2.4	2.0
All ($n = 9$)	Mean	59.1	163.1	2806	0.51	*GM*	205.5	106.8	128.2	446.2
	SD	13.0	8.6	970	0.2	*GSD*	1.8	1.9	2.1	1.8

[a]SAA = surface area of application; *SD* = standard deviation; *GM* = geometric mean; *GSD* = geometric standard deviation.

article of Selim and co-authors (1995), reported that unchanged DEET was identified with applying doses similar to or higher than ours (Moody, Benoit, Riedel, & Ritter, 1989; Wu, Pearson, Shekoski, Soto, & Stewart, 1979).

It can be summarized that DEET was excreted to urine quickly, and doses exceeding the capacity of metabolism would leave portions not metabolized. The unchanged DEET in urine was roughly 6.4% for children, but up to 28.7% for adults; even with the fewer DEET urinary metabolites in total, the adult subjects had more unchanged DEET in quantity than did children (Table 1). This result that children metabolized DEET far more than adults was supported by a number of studies (Evans et al., 1989; Ginsberg et al., 2002; Yanni et al., 2010), which indicated that hepatic clearances of a variety of drugs were higher from children than from adults.

Selim and co-authors (1995) also reported R3N1, the successor of R3N0 and R0N1, but it was not found in our study. This may have been owing to the high dose that made the enzymes work mostly to yield the initial metabolite, R0N1, or possibly to a certain extent due to the ethnic difference (western vs. Asian). As previous studies indicated, DEET metabolism underwent two pathways, ring methyl oxidation to give R1N0 and N-deethylation to yield R0N1 initially. Several succeeding metabolites, as shown in Figure 1, were observed from a number of human or animal studies (Schoenig, Hartnagel, Osimitz, & Llanso, 1996; Taylor, 1986; Taylor & Spooner, 1990; Wu et al., 1979), and even from a fungus study (Seo et al., 2005). Ring methyl oxidation was favored when the concentration was low, and N-deethylation caught up when DEET became rich (Constantino & Iley, 1999); the clearance with human liver microsomes, reported by Usmani and co-authors (2002), was approximately eight times higher for ring methyl oxidation than for N-deethylation. It is obvious that transdermal permeation of DEET would have the occurrence of ring methyl oxidation first as DEET was in low quantity during the early process of absorption. Our results showing the high percentage of R3N0 (78.2% for children, 46.1% for adults), the product of ring methyl oxidation, fair amounts of R0N1 (16.2% for children, 23.9% for adults), and the initial metabolite of N-deethylation were consistent with that of previous studies.

The regression analysis indicated that the difference in J_E_DEET between the two age groups was attributed to DEET loading and height. DEET loading resulted from Fick's first law of diffusion, whereas height, as an alternative of age, was inversely related to the dermal absorp-

TABLE **2**

Pearson Correlations Between Absorption and Demographic Variables

Variables	Sex	Weight	Height	Application Area	Age Group	DEET Loading	J_R3N0[a]	J_R0N1[a]	J_E_DEET[a]
Weight	-0.19	1							
Height	-0.10	0.95**	1						
Application area	-0.21	0.80**	0.88**	1					
Age group	0.03	0.92**	0.97**	0.79**	1				
DEET loading	0.24	-0.50**	-0.55**	-0.71**	-0.48*	1			
J_R3N0	0.19	-0.59**	-0.64**	-0.70**	-0.58**	0.86**	1		
J_R0N1	0.08	-0.43*	-0.50**	-0.61**	-0.45*	0.71**	0.63**	1	
J_DEET[a]	-0.17	-0.22	-0.27	-0.25	-0.22	0.09	0.21	0.43*	1
J_E_DEET	0.14	-0.59**	-0.65**	-0.73**	-0.59**	0.85**	0.95**	0.84**	0.42*

[a]J_X = the calculated steady-state flux of metabolite X (μg/cm^2/s).
*p < .05 (two-tailed).
**p < .001 (two-tailed).

TABLE **3**

Linear Regression Analysis of Absorption and Affecting Factors

Model		Unstandardized Coefficients		Standardized Coefficients	Significance	R^2
		B	*SE*	Beta		
1	(Constant)	0.018	0.023		.454	.729
	DEET loading	0.098	0.012	0.854	<.001	
2	(Constant)	0.274	0.117		.029	.777
	DEET loading	0.081	0.014	0.708	<.001	
	Height	-0.002	0.001	-0.263	.036	

Note. Dependent variable was J_E_DEET.

tion of DEET, implying that the lower-height subjects (i.e., children) had more dermal absorption of DEET than did the taller ones. Aging on percutaneous absorption may have been a factor, as a previous study reported that aged people had less dermal absorption of hydrophilic compounds than young ones (Roskos, Maibach, & Guy, 1989); that fact, however, was not likely to fit in with our finding, because the age gap in this study was not as large and DEET is more lipophilic than hydrophilic. Another factor that can be of concern is temperature, which has been confirmed to enhance percutaneous absorption of drugs by a number of studies (Boman & Maibach, 2000; Chang & Riviere, 1991; Craig, Cummings, & Sim, 1977; Tominaga & Tojo, 2010; Wiechers, 1989). The child subjects of our study had more physical activities and stayed less frequently in air-conditioned environments; thus, the dermal temperature had to be usually higher than that of the adult subjects and consequently resulted in more DEET absorption. No matter what really caused children to absorb more DEET than adults, our result did demonstrate such a fact, to which parents or guardians have to pay attention while using these products on children.

Conclusion

Our study identified DEET urinary metabolites R3N0 and R0N1, and unchanged DEET, from child and adult subjects. Ring methyl oxidation was the predominant reaction to give rise to R3N0 more than R0N1 for the first eight hours. Dermal absorption of DEET was not only driven by the loading on the skin, but also inversely affected by body stature.

Acknowledgements: The authors thank all participating subjects and the parents and teachers of the young subjects. This study was sponsored by the National Science Council (NSC 94-2314-B-320-012 and NSC 95-2314-B-320-011) in Taiwan.

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References on page 168

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2. A description of the nominee's educational background and professional experience.
3. A description of the nominee's employment history, including the scope of responsibilities.
4. A narrative statement of specific accomplishments and contributions on which the nomination is based, including professional association activities, publications, and community/civic activities.
5. Three endorsements (an immediate supervisor and two other members of the professional staff or other person as appropriate).

NOMINATIONS MUST BE RECEIVED BY APRIL 15, 2014. THREE COPIES OF THE NOMINATION DOCUMENT MUST BE SUBMITTED TO:

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INTERNATIONAL PERSPECTIVES

Prepublished online September 2013, National Environmental Health Association.

Framework for Handling Asbestos After a Tidal Surge

Although most of the information presented in the Journal refers to situations within the United States, environmental health and protection know no boundaries. The Journal periodically runs International Perspectives to ensure that issues relevant to our international membership, representing over 30 countries worldwide, are addressed. Our goal is to raise diverse issues of interest to all our readers, irrespective of origin.

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Abstract **The tidal surge associated with Tropical Cyclone Yasi—a Category 5 system—on February 3, 2011, culminated in asbestos-containing material (ACM) becoming comingled with soil, sand, vegetation, and other debris in the communities of Tully Heads and Hull Heads in Queensland, Australia. The situation was a major concern and the area was deemed by the Queensland Government a priority due to the potential public health risk. The immediate challenge was that no agreed-upon operational framework existed between key response organizations for handling ACM after a tidal surge. This resulted in the development of strategies for addressing this situation during the response. An expansion of "declared disaster officers" under Queensland's Disaster Management Act 2003 was required to allow licensed asbestos contractors to enter and clean up public and private land at Tully Heads and Hull Heads. This declaration was the first time a group of people other than enforcement officers had been given such powers in Queensland. The situation was handled effectively; however, lessons have been learned and improvements can be made to enhance efficiency, planning, and reporting.**

Introduction

The tidal surge associated with Tropical Cyclone Yasi culminated in asbestos-containing material (ACM) becoming comingled with soil, sand, vegetation, and other debris in the communities of Tully Heads and Hull Heads in Queensland, Australia. These are small communities on the northeast coast of Australia, approximately 160 km south of Cairns and 1,600 km north of Brisbane, the state capital of Queensland. The situation became a concern because improper handling and disposal of ACM could lead to future environmental and health issues (Luther, 2011). Further, no agreed-upon framework existed between key response agencies for handling ACM after a tidal surge, nor was there evidence of a similar situation occurring in recent Australian history.

To facilitate preparedness for disasters, this article discusses what occurred and provides a framework for handling asbestos, and possibly other public health risks, based on this experience. This article includes an overview of the cyclone, the definition of ACM, an outline of the public health risk of asbestos, discussion of the response, and a description of the processes used. The lessons learned are also discussed in order to provide an understanding of the challenges that may arise if a similar situation is experienced. Finally, an implementation strategy is provided along with a suggestion that this approach be applied as a generic framework for other public health risks.

At Tully Heads and Hull Heads, many of the houses alongside the road parallel and closest to the beach suffered major damage from the tidal surge, and some single-story houses washed away (see photo on page 2). The tidal surge penetration inland was approximately 500 m and reached around 1.5–2 m above the highest astronomical tide (James Cook University, 2011).

Prior to the cyclone, the largest hospital in the Cairns region, Cairns Base Hospital, was evacuated. An evacuation also occurred of low-lying areas between Cairns and Townsville (James Cook University,

Impact of tidal surge—damage to external walls on the seaward side of a house.

2011). The cyclone impacted a region with a population of approximately 450,000 (Australian Bureau of Statistics, 2012). No deaths or serious injuries occurred as a result of the impact of the cyclone. The power supply to more than 220,000 homes and businesses was interrupted, with supplies restored to 200,000 homes and business within the first week and the remainder connected within 23 days (Queensland Reconstruction Authority, 2011).

Tully Heads and Hull Heads Demographics

According to the Australian Bureau of Statistics (2011) census data, the population of Tully Heads and Hull Heads was 438 with 245 males and 193 females. The median age was 55 years. The most common ancestries were English, 33%; Australian, 27%; Irish, 9%; Italian, 8%; and German, 6%. The majority of people were born in Australia (86%) and the most common overseas countries were England, New Zealand, Hungary, Italy, and Netherlands. At the time of census the most common religions were Catholic, 31%; no religion, 24%; Anglican, 22%; Uniting Church, 8%; and Presbyterian and Reformed, 6%. The families were 25% couples with children, 59% were couples without children, and 16% were one-parent families.

ACM

Asbestos is a generic name that refers to a group of six naturally occurring fibrous silicate minerals (actinolite, amosite, anthophylite, chrysotile, crocidolite, and tremolite) (Mahini, 2005). These fibrous minerals have properties that make asbestos cost-effective and versatile. They have high tensile strength, chemical and thermal stability, flexibility, and low electrical conductivity (Masayuki & Seiirchiro, 2006). ACM is made up of products or materials that contain asbestos in an inert bound matrix such as cement or resin (Department of Health, 2009). This material is included in building products such as profiled sheets used on roofs and walls and flat sheets in flashings, imitation brick cladding, roof shingles, plaster patching compounds, textured paint, and vinyl floor tiles (Occupational Health and Safety Unit, 2011). Although asbestos is now banned from use in Australia, it was a component of thousands of different products used in the community and industry from the 1940s until the late 1980s with some uses of asbestos products, mainly friction materials and gaskets, continuing until December 31, 2003 (Department of Justice and Attorney-General, 2009).

What Is the Public Health Risk?

Asbestos exposure has been shown to cause mesothelioma as well as lung cancer and nonmalignant pulmonary and pleural disorders (Kukkonen et al., 2011). Mesothelioma is caused by the inhalation of needle-like asbestos fibers deep into the lungs where they can damage cells (National Occupation Health and Safety Commission, 2005). The latency period between initial exposure and the onset of disease is 15 or more years, but varies with intensity of exposure (Antao, Pinheiro, & Wassell, 2009). Mesothelioma was once rare, but its incidence is increasing throughout the industrial world as a result of past exposures to asbestos. Australia has the highest incidence in the world (National Occupation Health and Safety Commission, 2005) along with Belgium and Great Britain (Bianchi & Bianchi, 2006).

Asbestos poses a risk to health whenever asbestos fibers become airborne and people are exposed to these fibers (Occupational Health and Safety Unit, 2011). Asbestos fibers are common in the general environment at low levels. According to Queensland Health (2010), low-level exposure to asbestos fibers is considered unlikely to result in the development of an asbestos-related disease, as fiber concentrations are likely to be insufficient to increase cumulative lifetime exposure. ACM when dry is considered friable and may become crumbled, pulverized, or reduced to powder by hand pressure. In this form ACM poses the greatest health risk (Department of Education, 2007). Although asbestos can have dangerous health effects, it is safe if handled correctly (Department of Justice and Attorney-General, 2011). The lack of immediate health effects has often meant that victims are unaware of the risks and subsequently exposure to the hazard can continue over a long period, causing serious health effects (Department of Justice and Attorney-General, 2009).

If left undisturbed, ACM poses a negligible health risk because the asbestos fibers are bound (Queensland Health, 2010). Air testing near the ground has shown the concentration of fibers is very low for undisturbed asbestos, very similar to it not being present (The Western Australian Advisory Committee on Hazardous Substances, 1990). All types of asbestos, if disturbed (i.e., through drilling and cutting) and inhaled at sufficient doses, can be harmful (Valić, 2002). The potential health impacts posed by different asbestos minerals and fiber dimensions should be treated as equivalent (Department of Health, 2009). The World Health Organization control limit for asbestos fibers in any localized atmosphere, measured and averaged over four hours, is 0.1 asbestos fibers per cubic centimeter of air (0.1 f/cm^3). This is not deemed to be a safe level and any type of work activities involving asbestos should be designed to be as far below the control limit as possible (Health and Safety Executive, 2011).

ACM in a Disaster Response

One of the immediate problems after a disaster is the mixing of hazardous and nonhazardous wastes (Wisner & Adams, 2002). ACM is considered a hazardous waste and as such needs to be identified early during a disaster response and handled appropriately to prevent any risks to public health. This is particularly important to ensure subsequent disturbance and dissemination do not occur across the area and result in costly delays and extra investigative and remediation effort (Department of Health, 2009). Improper disposal of ACM may lead to future environmental health issues and highlights the need for an agreed-upon and coordinated approach when handling ACM after a disaster (Luther, 2011). Further, it is unlikely that the community, response volunteers, and workers will, in the first instance, be provided with appropriate personal protective equipment (PPE), thus increasing their risk of long-term health problems (Alexander & Liangmahe, 2008).

Response to Tropical Cyclone Yasi

The communities of Tully Heads and Hull Heads in the Cassowary Coast Regional Council (CCRC) area, Queensland, Australia, were heavily impacted by a tidal surge from Tropical Cyclone Yasi. Due to the age of some dwellings (pre-1990s), the tidal inundation resulted in ACM becoming comingled with other debris and impregnated in the soil across private and public land (see photo above). An initial environmental health rapid assessment of Tully Heads and Hull Heads was conducted on February 4, 2011, by teams of environmental health officers (EHOs) from Queensland Health in response to Tropical Cyclone Yasi.

Tully Heads with asbestos-containing material on public and private land.

The situation became a major concern and the area was deemed a priority due to the potential public health risk and growing media interest. The local communities used the media to get messages across to politicians and the general public about the issue and the need for rapid assistance. The issue was also escalated due to an upcoming state election.

The main public health concern was that if ACM was not cleaned up, or it was stored untreated, a further risk to public health may be presented through future daily activities such as lawn mowing and gardening. The immediate risk to public health associated with an appropriate cleanup was considered minimal; however, ACM is not biodegradable and if left unmanaged it presents an unknown public health risk in the future. Also, the full extent of the situation was unknown as the area was initially inaccessible for response agencies.

In response, key messages on the safe handling of ACM were communicated through newsletters, talk back radio, and information at community centers. A list of licensed contractors for asbestos removal and transport was made available, and residents, the state emergency service, and the army were also requested to notify CCRC of any suspected asbestos debris in the area and if possible to cordon off the area to restrict or prevent access.

The public was provided with access to asbestos kits containing gloves, masks, tape, plastic, and body suits at all community centers. Initially, it was business as usual for a disaster response:

- ACM on private property was the responsibility of the owner/occupier;
- owners/occupiers were requested to separate debris and place on the curbside and if possible double-wrap ACM in plastic; and
- CCRC collected ACM for disposal.

CCRC commenced remediating the public land at Tully Heads on February 10, 2011. This was being undertaken by a qualified asbestos removalist team. It quickly became evident that the contamination of the tidal surge areas of Tully Heads and Hull Heads was beyond the capacity of local resources. At the same time a significant amount of media and political interest was present about the need to remediate the area quickly to allow the community to go back to their homes. This included community meetings with Members of Parliament to rally support.

On February 11, 2011, the estimated cost for the ACM cleanup of residential properties was reported as far exceeding insurance payouts. Subsequently, property owners were going to find it difficult to cover the costs in ensuring the appropriate disposal of the hazardous waste and remediation of their property. Due to this issue and the growing concerns about the heavy amounts of ACM comingled with other debris, a decision was made by CCRC and Queensland Health that the cleanup of this area would need to be funded and undertaken by licensed asbestos contractors.

At the Local Disaster Management Group (LDMG) and District Disaster Management Group (DDMG) meeting on February 16, 2011, a decision was made that the Department of Public Works (QBuild) would lead the response in the Tully Heads and Hull Heads areas affected by the tidal surge. To guide this response an asbestos working group (AWG) was formed to coordinate and address all issues throughout Tully Heads and Hull Heads. The members of the AWG included Queensland Police (Chair); Queensland Health; QBuild; Department of Environmental and Resource Management; CCRC; Workplace Health and Safety; Department of Employment, Economic Development and Innovation; and the Department of Communities.

The AWG met for the first time on February 16, 2011, to formulate a strategy for commencing works. This included discussing the scope of work; possibilities for evacuating the area while heavy machinery was being used for the cleanup, transportation, and appropriate handling and disposal of ACM; provision of PPE to workers and residents; and appropriate communication to the residents. As a number of government agencies with a vested interest in asbestos-related issues were being represented, a course of action had to be determined and agreed upon (e.g., education rather than enforcement).

The need for the DDMG to extend the definition of "declared disaster officers" under Queensland's Disaster Management Act 2003 to clean up Tully Heads and Hull Heads was also discussed and later approved. These powers are generally provided for responders such as ambulance officers, fire officers, health officers, police officers, and other people who have the necessary expertise or experience to exercise the powers. Powers include the ability to control the movement of people, enter property, destroy or demolish property, etc. In this situation, expansion of the "declared disaster officers" was required to include licensed contractors so they could carry out the required tasks in the declared disaster area. This declaration was the first time a group of people had been given such powers in Queensland.

A key element of the AWG was to ensure effective communication with the residents and the general public. The AWG addressed a public meeting at Tully Heads on February 16, 2011. During the meeting residents were advised the group had been formed with a directive to ensure that their community was efficiently and safely cleaned as a matter of urgency. Residents were also advised that PPE and appropriate bagging materials in asbestos packs were available to all residents in the area to ensure exposure to asbestos fibers was minimized during cleanup operations (if they chose to stay in the area).

On February 18, 2011, the remediation of affected areas of public and private land commenced. As part of this an approved emergency disposal site at Hull Heads was opened. Teams of asbestos removal contractors were mobilized and they removed broken ACM from yards, streets, road verges, footpaths, and mangrove areas behind properties and the top layer (approximately 300 mm) of sand containing debris.

PPE was continually made available to all residents in the affected areas and QBuild was communicating with the community (via newsletters) about the works. Permission was granted for volunteers to assist on the condition that a licensed asbestos contractor was present. Also, Queensland Police Service and QBuild road blocks were put in place to ensure that only authorized residents and contractors were permitted access to the site during remediation.

One month after the cyclone crossed the coast, on March 7, 2011, the ACM cleanup in the areas of Tully Heads and Hull Heads was declared complete. Throughout the cleanup, newsletters were continually distributed to residents every few days about the status of the remediation and the precautions required ensuring the safe removal of ACM within their own yards.

It was estimated that within Tully Heads and Hull Heads, 50% of general waste piles contained ACM. Both communities were remediated with the top 300 mm of soil taken from affected areas with a total of 7,000 m^3 of contaminated soil disposed of at the old Hull Heads landfill. This soil was capped by 800 mm of a clay-based material and 300 mm of mulch, then fenced and signed to prevent any future environmental health risks.

Legislative Powers

In Queensland, Australia, the Public Health Regulation 2005 provides the primary legislative guidance for handling asbestos. Homeowners and owner-builders must hold a certificate to remove more than 10 m^2 of nonfriable (bonded) asbestos materials (Queensland Government, 2011). Removing more than 10 m^2 of nonfriable asbestos materials must be undertaken by a licensed asbestos removalist (Queensland Government, 2011). This presented a challenge as most private properties had more than 10 m^2 of nonfriable asbestos and occupiers did not hold an appropriate certificate.

The ambiguity of the legislation and its administration added complexity to the response. Administration is shared between local government, Queensland Health, and Workplace Health and Safety. Generally, enforcement on private property is undertaken by EHOs from Queensland Health, and Workplace Health and Safety regulate licensed asbestos contractors or removalists (e.g., workplaces and businesses). On Queensland government-owned land, the Department of Public Works overseas asbestos management activities for the state.

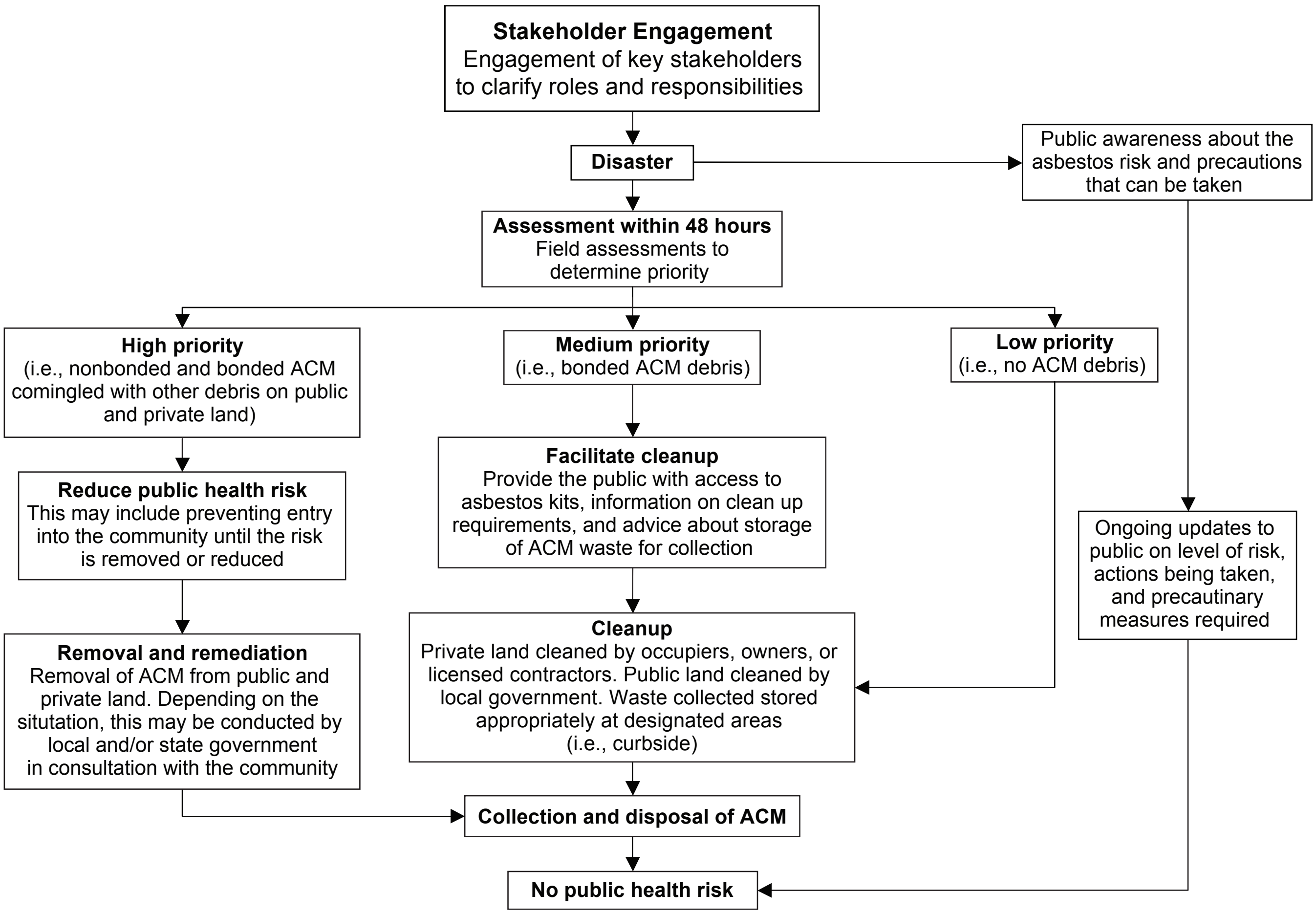
Access to property was not a major issue during the response. Business-as-usual processes were followed during the response, which included seeking permission to enter. As it was a declared disaster area under the Disaster Management Act 2003, however, any place in the declared area could be accessed by authorized people.

Funding

The cleanup of Tully Heads and Hull Heads was funded through the Australian Government Natural Disaster Relief Recovery Arrangements (NDRRA). The arrangements are formed around three levels of government: local, state, and fed-

FIGURE **1**

Asbestos Handling Framework



ACM = asbestos-containing materials.

eral (Department of Community Safety, 2012). The Queensland disaster management arrangements, however, include an additional state government tier between local and state governments known as disaster districts (Department of Community Safety, 2012). This model allows for an effective operational service delivery to support local communities in a large geographical and decentralized state.

Once a disaster is declared by the Premier of Queensland or relevant minister, grant schemes and other assistance programs are activated. For example, when local resources are overwhelmed in a disaster situation, assistance (including funding) is sought from the district. This process is replicated up through the system until support is received. The support and delivery of services are always focused on the local government.

In this situation, support was provided by the district and state to clean up Tully Heads and Hull Heads. The overall cost for the cleanup is unknown but the damage from Tropical Cyclone Yasi was estimated to cost U.S. $950 million (The Australian, 2011).

Framework for Handling Asbestos After a Disaster

During the response it became evident that no agreed-upon framework existed for the handling of ACM after a disaster. Using the steps undertaken during the response and through consultation with the Far North Queensland Public Health Disaster Management Working Group (working group), a framework for handling asbestos after a disaster has been developed (Figure 1). The working group is made up of environmental health and disaster management experts from across the region.

The purpose of the framework is to provide guidance on the steps to be undertaken. For this framework to be effective and applied locally it is vital for LDMGs to undertake stakeholder engagement and determine who performs the necessary steps (e.g., conducting field assessments, clarifying the priority definition, and who completes the remediation). It is recommended that an assessment of a community be completed within 48 hours to determine the priority of the required actions. Due to limitations in the rapid identification of ACM this would only include a visual inspection. Although this is not 100% accurate this method allows an investigator to establish a reasonable belief that ACM is or is not present. This approach is supported in Queensland by the Public Health Act 2005, which allows reasonable belief to be used as justification for a public health order (a direction for certain action to be undertaken).

The assessment of ACM involves answering three questions about the house/structure:

- Is it built before 1990?
- Does the material look like a material known to contain asbestos?
- Is the material installed in a location where ACMs were known to be used?

The underlying theme of any visual inspection would be to categorize any suspected material as ACM (Queensland Health, 2008). The subsequent actions required would be determined by priority:

- high-priority areas would be defined as areas where nonbonded and bonded ACM are comingled with other debris (e.g., tidal surge);
- medium-priority areas have bonded ACM debris that has not been comingled; and
- low-priority areas are where no ACM debris is present.

The immediate actions required for a high-priority area may include preventing entry into a community until the risk is removed or reduced. Meanwhile, for a medium-priority area this may result in a local government providing access to asbestos kits, information on cleanup requirements, and advice about storage of ACM waste for collection. For a low-priority area, no action is required.

A need exists for ongoing public awareness to be undertaken as a parallel activity. This would include the provision of advice around the handling and storage of ACM (e.g., keep wet and covered), and where residents could collect asbestos-handling kits. Advice would also be provided about the separation of ACM from other debris, labeling requirements, and the need to restrict access, particularly for children.

Implementation Strategy

To enhance preparedness for disasters, the following suggested actions have been provided for governments and key response organizations to effectively manage ACM. It is recommended these actions be complemented by the asbestos-handling framework in Figure 1.

1. Establish an agreed-upon framework for handling ACM after a disaster.
2. Develop an understanding of the communities that have buildings with suspected ACM. This information can then be used to determine the priority areas for assessment after a disaster. It is recommended that an asbestos map (similar to a flood map) of communities be created.
3. Conduct monitoring of airborne ACM (in areas where ACM is believed to be present) to measure the level of public health risk and allow appropriate actions to be taken (e.g., preventing people from entering a community unless they are wearing PPE).
4. Identify disposal sites for ACM and the processes that will be undertaken to transport the debris.
5. Develop fact sheets to ensure consistent and accurate messages are communicated to the public.
6. Determine who will conduct rapid assessments (within 48 hours) of communities after a disaster. It is also important to ensure these people have the appropriate skills and training. Also, formalize how this will be conducted (e.g., the tools that will be used, transportation to assessment sites, and reporting structure).
7. Maintain a store of asbestos cleanup packs that can be efficiently accessed and distributed throughout the community after a disaster.
8. Document the process in the appropriate disaster management plan.

Application of Framework to Other Public Health Risks

The asbestos-handling framework (Figure 1) and implementation strategy can be applied to any public health risk after a disaster. It provides a generic template for ensuring preplanning is conducted with all stakeholders, provides a mechanism for systematically identifying public health risks prior to a disaster, and allows interventions to be based on evidence. For example, point two from the implementation strategy could be expanded beyond identifying buildings with suspected ACM to gathering baseline public health data (e.g., drinking water system, location of chemical plants, type of sewage systems, etc). All of these are vital for identifying and measuring the impact of a disaster on public health. Such an approach would ensure a comprehensive framework is available to efficiently prepare for and respond to the public health risks associated with disasters.

Lessons Learned

- Most communities in Australia have some structures containing ACM and during cyclones, floods, fires and tidal surges, this material will be scattered and may pose a public health risk.
- The main objective during a disaster response must be to minimize the risk to all of the community while removing ACM and other hazardous material.
- During a disaster situation, simple plans produce less confusion.
- Community awareness of what is happening during a disaster response is vital.
- An agreed-upon framework needs to be in place between responding organizations for handling ACM prior to a disaster.
- A framework for handling ACM needs to be included in disaster management plans, which also requires a disposal site that will receive the material during the response.
- A comprehensive environmental health response framework is required to address all public health risks before and after a disaster.

Conclusion

The tidal surge associated with Tropical Cyclone Yasi facilitated the comingling of ACM with soil, sand, vegetation, and other debris in Tully Heads and Hull Heads thus providing a unique challenge to both the community and the response organizations managing the disaster cleanup. The main challenge was no agreed-upon framework existed between key response organizations for handling ACM after a tidal surge. This required the strategies for addressing this situation to be developed during the response. The response included the establishment of an asbestos working group, which developed and implemented a strategy for remediating these communities. This approach allowed the situation to be handled effectively. A number of improvements can be undertaken, however, to enhance efficiency. The most vital improvement is the development of an agreed-upon framework for handling ACM among response organizations, which will open communication lines, enhance preparedness, and more broadly improve responses to public health risks associated with future disasters.

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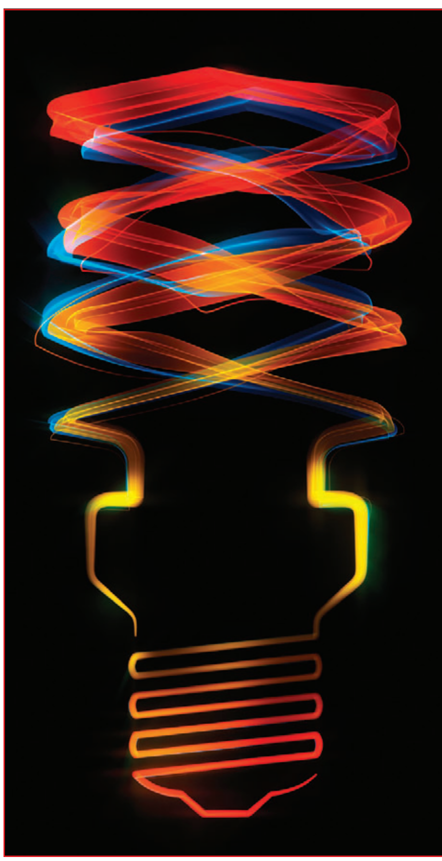
2014 Educational Contribution Award

This award was established by NEHA's board of directors to recognize NEHA members, teams, or organizations for an outstanding educational contribution within the field of environmental health. This award provides a pathway for NEHA members and environmental health agencies to share creative methods and tools to educate one another and the public about environmental health principles and practices. Don't miss this opportunity to submit a nomination to highlight the great works of your colleagues!

Nominations are due in the NEHA office by March 17, 2014.

For more information, please visit **www.neha.org/about/awardinfo.html**.
Nomination materials can be obtained by e-mailing Terry Osner at **tosner@neha.org**.

NATIONAL ENVIRONMENTAL HEALTH ASSOCIATION



2014 Environmental Health Innovation Award

This award was established by NEHA's board of directors to recognize a NEHA member or organization for creating a new idea, practice, or product that has had a positive impact on environmental health and the quality of life. Innovative change that promotes or improves environmental health protection is the foundation of this award.

Environmental health professionals face the dilemma of finding and implementing new ways of doing business without sacrificing the quality of their environmental health programs. This annual award recognizes those who have made an innovative contribution to the field, as well as encourages others to search for creative solutions. Take this opportunity to submit a nomination to highlight the innovations being put into practice in the field of environmental health!

Nominations are due in the NEHA office by March 17, 2014.

For more information, please visit
www.neha.org/about/awardinfo.html.
Nomination materials can be obtained
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NATIONAL ENVIRONMENTAL HEALTH ASSOCIATION

CAREER OPPORTUNITIES

Food Safety Inspector

UL Everclean Services is the leader in the restaurant inspections market. We offer opportunities throughout the country. We currently have openings for professionals to conduct Q.A. audits of restaurants.

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Butte, MT	Pocatello, ID
Cleveland, OH	Puerto Rico
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NEHA offers wide-ranging opportunities for professional growth and the exchange of valuable information on the international level through its longtime Sabbatical Exchange Program. The sabbatical may be taken in England, in cooperation with the Chartered Institute of Environmental Health (CIEH), or in Canada, in cooperation with the Canadian Institute of Public Health Inspectors (CIPHI). The sabbatical lasts from two to four weeks, as determined by the recipient. The exchange ambassador will receive up to $4,000 as a stipend, depending on the length of the sabbatical, and up to $1,000 for roundtrip transportation.

The application deadline is **March 3, 2014**. Winners will be announced at the NEHA 2014 Annual Educational Conference & Exhibition in Las Vegas, Nevada, in July 2014. The sabbatical must be completed between August 1, 2014, and June 1, 2015.

For more information, contact Terry Osner at tosner@neha.org.

To access the online application, visit www.neha.org/about/awardinfo.html.

College of Health Professions and Social Work
TEMPLE UNIVERSITY®

Assistant Professor of Environmental Health
Non-Tenure Track Appointment

Position Summary: The Department of Public Health at Temple University seeks an Assistant Professor, non-tenure track appointment to teach in its undergraduate and graduate programs. Specific responsibilities will include teaching undergraduate or graduate courses (3-4 per semester) in the content area of Environmental Health, curriculum evaluation, updates of existing courses, and serving on various departmental and university committees.

The Department of Public Health has several masters' degrees, including the MPH, and a PhD in Public Health. With over 300 undergraduate majors, a CEPH accredited BS degree in public health is also offered. Public health education at Temple University has been accredited by the Council on Education in Public Health since 1985; current accreditation runs through 2016. The department has 26 full time faculty and numerous adjunct faculty from the community. Faculty have a diverse range of public health interests and research activities, including social and behavioral sciences, global health, epidemiology, biostatistics, environmental health, and health policy.

Appointment: The appointment is an Assistant Professor, 9-month, non-tenure track, faculty position within the Department of Public Health, which is housed in the College of Health Professions and Social Work. Salary is competitively set and full benefits are included with the position. This appointment is for one academic year, August 18, 2014 to June 30, 2015 with the potential for renewal.

Qualifications: Applicants must have a doctoral degree in a field relevant to the environment and human health, as well as a prior record of outstanding teaching, excellent oral and written communication skills, and an ability to interact with a variety of public health professionals. Teaching experience and expertise should include core environmental health competencies at the graduate and undergraduate level, and at least one of the following areas: environmental regulation and policy, GIS, workplace health and safety, or instrumentation and exposure assessment.

Please send curriculum vitae, contact information for three references, and an introductory letter to:

Public Health Faculty Search Committee
Department of Public Health
Temple University
1301 Cecil B Moore Avenue
Ritter Annex, 9th floor
Philadelphia, PA 19122-6091

Electronic submissions may be sent to: hausman@temple.edu

Temple University is an equal opportunity/affirmative action institution. People of color, women, veterans, and persons with disabilities are encouraged to apply.

EH CALENDAR

UPCOMING NEHA CONFERENCES

July 7–10, 2014: NEHA's 78th Annual Educational Conference & Exhibition in Partnership with the International Federation of Environmental Health, The Cosmopolitan of Las Vegas, NV. For more information, visit www.neha2014aec.org.

NEHA AFFILIATE AND REGIONAL LISTINGS

Alabama
April 9–11, 2014: 2014 Annual Education Conference, sponsored by the Alabama Environmental Health Association, The University of Alabama at Birmingham, AL. For more information, visit www.aeha-online.com.

Arizona
March 12–13 , 2014: AZEHA Spring Conference, sponsored by the Arizona Environmental Health Association, Arizona State University, Tempe, AZ. For more information, visit www.azeha.org.

California
March 31–April 4, 2014: 63rd Annual Educational Symposium, "Harvest the Knowledge," hosted by the Redwood Chapter of the California Environmental Health Association, Napa Valley Marriott Hotel, Napa, CA. For more information, visit www.ceha.org/events.

Idaho
March 19–20, 2014: 2014 Annual Educational Conference, sponsored by the Idaho Environmental Health Association, Boise, ID. For more information, visit www.ieha.wildapricot.org.

Iowa
April 1–2, 2014: Iowa Governor's Conference on Public Health, partnered by the Iowa Environmental Health Association, Scheman Conference Center, Ames, IA. For more information, visit www.ieha.net.

Kentucky
February 18–20, 2014: KAMFES Conference, sponsored by the Kentucky Association of Milk, Food, and Environmental Sanitarians, Marriott Griffin Gate Resort, Lexington, KY. For more information, visit www.kamfes.com.

Michigan
March 18–21, 2014: 2014 Annual Education Conference, sponsored by the Michigan Environmental Health Association, Big Rapids, MI. For more information, visit www.meha.net.

Minnesota
January 30, 2014: Winter Education Conference, sponsored by the Minnesota Environmental Health Association, University of Minnesota Continuing Education and Conference Center, St. Paul, MN. For more information, visit www.mehaonline.org/events.

New Jersey
March 2–4, 2014: Annual Conference & Exhibition, sponsored by the New Jersey Environmental Health Association, Tropicana Resort and Casino, Atlantic City, NJ. For more information, visit www.njeha.org.

Ohio
April 15–16, 2014: 2014 Spring Annual Education Conference, sponsored by the Ohio Environmental Health Association, Worthington Double Tree Hotel, Columbus, OH. For more information, visit www.ohioeha.org/annual-education-conference.aspx.

Utah
April 2–4, 2014: Spring Conference, sponsored by the Utah Environmental Health Association, Moab, UT. For more information, visit www.ueha.org/events.html.

Washington
May 12–13, 2014: 2014 Annual Educational Conference, "Environmental Public Health—Improving Quality of Life in Our Communities," sponsored by the Washington State Environmental Health Association, Great Wolf Lodge, Grand Mound, WA. For more information, visit www.wseha.org/2014-aec/.

Wisconsin
April 15–16, 2014: Spring Education Conference, sponsored by the Wisconsin Environmental Health Association, Kalahari Resort, Wisconsin Dells, WI. For more information, visit www.weha.net.

TOPICAL LISTINGS

Healthy Homes
May 28–30, 2014: National Healthy Homes Conference, sponsored byt the U.S. Department of Housing and Urban Development, Rebuilding Together, HGTV, and DIY Network, Nashville, TN. For more information, visit www.healthyhomesconference.org.

Smart Growth
February 13–15, 2014: 13th Annual New Partners for Smart Growth Conference, organized by the Local Government Commission, Hyatt Regency Hotel, Denver, CO. For more information, visit www.newpartners.org/program.

JEH QUIZ

FEATURED ARTICLE QUIZ #4

Effects of Centralized and Onsite Wastewater Treatment on the Occurrence of Traditional and Emerging Contaminants in Streams

Available to those holding an Individual NEHA membership only, the *JEH* Quiz, offered six times per calendar year through the *Journal of Environmental Health,* is a convenient tool for self-assessment and an easily accessible means to accumulate continuing-education (CE) credits toward maintaining your NEHA credentials.

1. Read the featured article carefully.
2. Select the correct answer to each *JEH* Quiz question.
3. a) Complete the online quiz at www.neha.org (click on "Continuing Education"),
   b) Fax the quiz to (303) 691-9490, or
   c) Mail the completed quiz to
   JEH Quiz, NEHA
   720 S. Colorado Blvd., Suite 1000-N
   Denver, CO 80246.

   Be sure to include your name and membership number!
4. One CE credit will be applied to your account with an effective date of January 1, 2014 (first day of issue).
5. Check your continuing education account online at www.neha.org.
6. You're on your way to earning CE hours!

Quiz Registration

Name

NEHA Member No.

Home phone

Work phone

E-mail

JEH Quiz #2 Answers

October 2013

1. e	4. d	7. b	10. d
2. a	5. c	8. c	11. a
3. d	6. a	9. d	12. d

➔ Quiz deadline: April 1, 2014

1. Nationwide, an estimated __ gallons of untreated sewage are discharged annually through leaking or overflowing sewer lines.
   a. 1 million
   b. 3–10 million
   c. 1 billion
   d. 3–10 billion

2. From the study data, streams draining residential areas showed relatively __ differences with respect to the type of wastewater treatment utilized in that area.
   a. no
   b. small
   c. large

3. All of the following are considered emerging contaminants in streams from wastewater with the exception of
   a. hormonally active compounds.
   b. personal care products.
   c. nutrients.
   d. pharmaceuticals.

4. In a 1999–2000 survey, pharmaceutical compounds, hormones, and other organic compounds were detected in __ of the streams sampled.
   a. 80%
   b. 70%
   c. 60%
   d. 50%

5. Onsite wastewater treatment systems are used by about __ of the U.S. population.
   a. 20%
   b. 30%
   c. 40%
   d. 50%

6. Which of the two counties sampled in this study had the highest number of residents relying on onsite wastewater treatment?
   a. Durham County.
   b. Orange County.

7. It is estimated that __ of U.S. septic systems are older than the estimated effective lifespan of an onsite wastewater treatment system.
   a. 11%
   b. 30%
   c. 50%
   d. 72%

8. Overall, NO_2+NO_3-N concentrations were statistically higher in samples from sites in areas of centralized wastewater treatment than from sites in areas of onsite wastewater treatment.
   a. True.
   b. False.

9. Concentrations of __ were lower in samples from sites in areas of onsite wastewater treatment than from sites in areas of centralized wastewater treatment.
   a. chloride
   b. chloride and nitrogen
   c. calcium
   d. potassium and sulfate

10. The greatest number of pharmaceuticals (six) was detected in samples from a site which is located in an area of __ wastewater treatment.
    a. centralized
    b. onsite

11. The highest coliform densities for the June 1 and June 16 sampling periods were detected in samples from a site that is located in an area of __ wastewater treatment.
    a. centralized
    b. onsite

12. The results of the study indicate that properly functioning onsite wastewater treatment systems had little effect on stream quality.
    a. True.
    b. False.

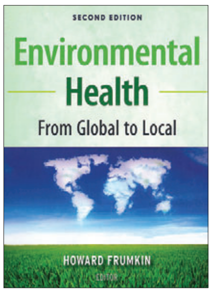
RESOURCE CORNER

Resource Corner highlights different resources that NEHA has available to meet your education and training needs. These timely resources provide you with information and knowledge to advance your professional development. Visit NEHA's online Bookstore for additional information about these, and many other, pertinent resources!

NEHA BOOKSTORE
www.neha.org/store

Environmental Health: From Global to Local (Second Edition)

Edited by Howard Frumkin (2010)

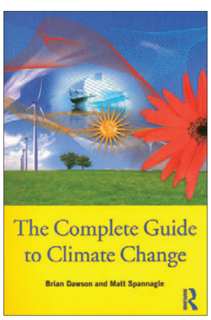


This comprehensive introductory text offers an overview of the methodology and paradigms of this burgeoning field, ranging from ecology to epidemiology, from toxicology to environmental psychology, and from genetics to ethics. Expert contributors discuss the major issues in contemporary environmental health: air, water, food safety, occupational health, radiation, chemical and physical hazards, vector control, and injuries. Also emphasizing a wide variety of issues of global interest, the thoroughly revised second edition contains updated information on such timely topics as toxicology, exposure assessment, climate change, population pressure, developing nations and urbanization, energy production, building and community design, solid and hazardous waste, and disaster preparedness.

1,221 pages / Hardback / Catalog #409
Member: $89 / Nonmember: $94

The Complete Guide to Climate Change

Brian Dawson and Matt Spannagle (2009)

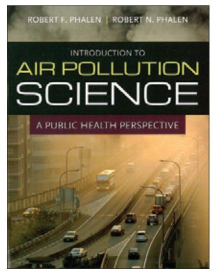


For anyone trying to separate the fact from the fiction, *The Complete Guide to Climate Change* is an indispensable resource. Taking you through the A to Z of the key scientific, geographical, and sociopolitical issues involved in the study of the environment and the implications of mankind's effect upon it, topics covered include 1) environmental science—the carbon cycle and the "greenhouse gases"; 2) the impacts of climate change on life, land, and sea; 3) mitigation strategies from carbon capture to carbon taxes; 4) the Kyoto Protocol and UNFCCC; and 5) renewable fuel sources, from wind to solar power. Including guides to the latest scientific and governmental thinking on climate change, this book will tell you all you need to know about perhaps the biggest issue facing mankind today.

436 pages / Paperback / Catalog #1077
Member: $32 / Nonmember: $35

Introduction to Air Pollution Science: A Public Health Perspective

Robert F. Phalen and Robert N. Phalen (2012)



This book offers a broad foundation for understanding the environmental issues associated with air pollution and its impact on human health. Echoing the approach to air pollution currently used by the U.S. Environmental Protection Agency, this groundbreaking book gives readers a solid grasp of this evolving field. It contains in-depth coverage of diverse subjects including sampling and analysis; visibility, climate, and the ozone layer; human exposures to air pollutants; toxicology and epidemiology studies; as well as risk assessment and ethics. This timely resource also addresses more specific issues like acid deposition, ozone depletion, environmental justice, clean technologies, and global climate change, providing readers with the analytical skills they need to comprehend today's air pollution challenges.

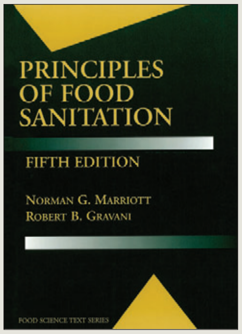
331 pages / Paperback / Catalog #1123
Member: $79 / Nonmember: $85

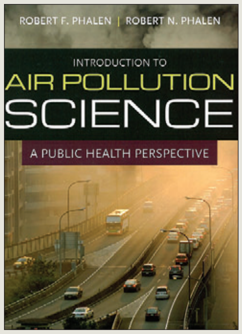
Public Health for the 21st Century: The Prepared Leader

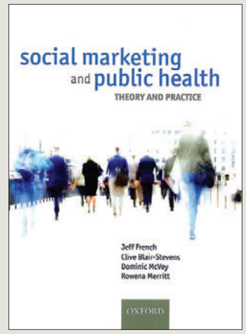
Louis Rowitz (2006)

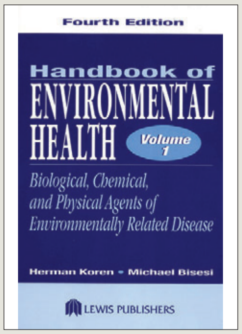
Public health has moved to the forefront of national interest and scrutiny in the light of present day events. Public health professionals are now regulars in all forms of media, something unheard of just a few years ago. The issues are well known—bioterrorism, SARS, West Nile virus—and they are enough to panic a population without skillful leadership. *Public Health for the 21st Century: The Prepared Leader* examines public health leadership in terms of emergency preparedness and specific skills and tools. As modern-day threats force leaders to look at how they address disasters and drive communities to prepare themselves, this book provides tools and real-life cases to hone management skills to prepare agencies to deal with large-scale events.

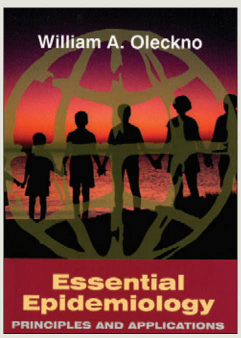
521 pages / Paperback / Catalog #932
Member: $104 / Nonmember: $109

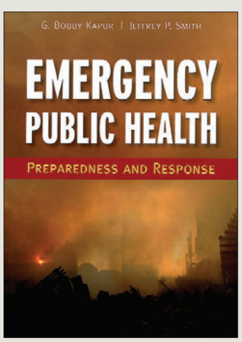












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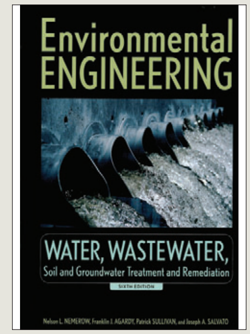

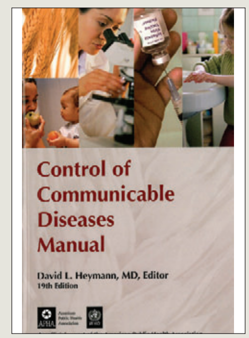

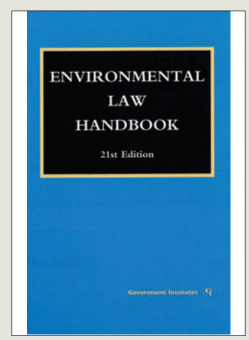

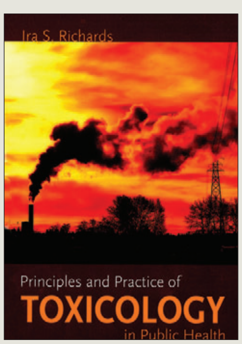

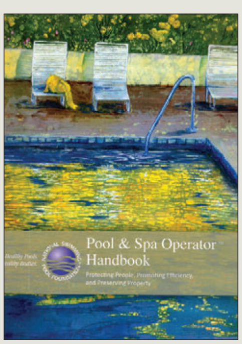

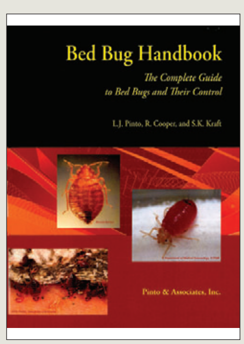

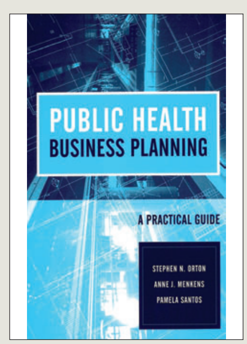

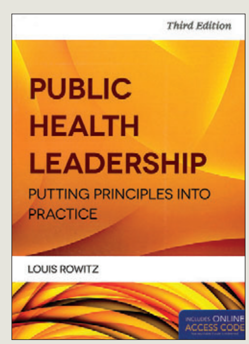

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Risk Assessment—TBD

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Vector Control—Zia Siddiqi, PhD, BCE, Director of Quality Systems, Orkin, Inc., Atlanta, GA. zsiddiqi@rollins.com

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Managing Editor's Desk
continued from page 194

porary issues such as health effects of global climate change, sustainability, healthy communities, the built environment, and smart growth. And we earned a substantial amount of nondues income from the Center, which has helped us to keep the financial costs of NEHA lower for our members.

Future Annual Educational Conference (AEC) & Exhibition Sites

As you know, our upcoming annual conference will be in Las Vegas next July 7–10. What you might not know is that the 2015 AEC will be in Orlando at the Marriott Renaissance at Sea World. (Also noteworthy is the fact that we have sensational room rates at the premier hotel properties where we'll be meeting—The Cosmopolitan of Las Vegas at $139 and the Marriott in Orlando at $129!)

As I write, we are about to make a final decision on the site of our 2016 AEC. It will be held either in the general Los Angeles area or in San Antonio.

Food Safety and the Private Sector

As many NEHA members know, we devote a considerable percentage of our work program to the topic of food safety—especially for the regulatory community of which so many NEHA members are a part. Food safety spans across all sectors of employment, however, including of course the private sector.

Realizing that we can do more not only to teach the private sector but also to build important bridges, relationships, and understandings between the private and public sectors, NEHA has constructed a series of workshops designed especially for private sector food safety professionals. We refer to this program as I-FIIT-RR (Industry-Foodborne Illness Investigation Training & Recall Response). With Food and Drug Administration (FDA) funding support, we are sponsoring three of these workshops this year. They are all modeled after our very successful Epi-Ready program. As a result of this specialized training, we anticipate more effective community-wide responses to foodborne illness outbreaks and a lightening of the load on environmental health specialists and local health departments.

We are also expanding our credentialing program to be more relevant to private-sector food safety professionals, with the launch of two new credentials this year (2013–14 fiscal year). The new Certified in Comprehensive Food Safety (CCFS) credential incorporates many of the considerations featured in our standard Certified Professional-Food Safety (CP-FS) credential but adds additional considerations from the realm of food manufacturing. Our new food safety auditing credential enables us to verify important knowledge, skills, and abilities for this line of important food safety work. We have finished the job task analysis (JTA) and exam for the food safety auditor credential, which should be launched in spring of 2014.

A Food Safety Education Novel Product

From our AEC to our Web site to our *Journal* and on and on, NEHA expends considerable effort to develop and deliver education on a host of environmental health issues. One of our more creative endeavors is currently underway. We are working with one of the world's largest publishers, Wiley, to develop short and focused videos across a number of food safety–related topics such as how to wash your hands, how to calibrate a thermometer, etc. Our plan is to rent or make these videos available to health departments for inspection interventions. We envision that environmental health professionals in the middle of an inspection would now be able to pull these videos up on their iPads and then show the appropriate person what they need to do or how to do it.

Developments With Our Highly Regarded CP-FS Credential

We now have a new and high-quality textbook available for this credential. In addition, we are moving to map the CP-FS JTA to the FDA's Voluntary Retail Food Standards. Assuming that this all works out, this will make this credential a much more valuable offering.

Healthy Homes Specialist (HHS) Credential

We just completed an update and overhaul of this important credential. We've now gone through an extensive exercise to reexamine the base of knowledge, skills, and abilities that holders of this credential need to have. It is planned in February 2014 to refine and update the item bank to test candidates for this knowledge.

Registered Environmental Health Specialist/Registered Sanitarian (REHS/RS) Credential

With all the tremendous work going into updating NEHA's CP-FS and HHS credentials, we've also focused attention on updating our premier REHS/RS credential. We have completed the JTA and are currently working on the item review for the exam. The new exam is anticipated to be available spring of 2014.

Work With World Bank

Several of our staff have been in conversation with the World Bank (and more specifically, the World Bank's Global Food Safety Partnership), which has expressed a strong interest in some of our training materials and instructor-led courses. They see these products as having applicability in other countries around the world. We are hopeful that we can work something out that would enable NEHA to expand the reach of some of our educational and training material.

Training for Small Water System Operators

We have completed our part of a multiparty grant submission that would have NEHA play a central role in the development of training and smartphone apps that will assist small water system operators in their job functions.

InFORM Conference

We developed an environmental health educational track for the 2013 InFORM (Integrated Foodborne Outbreak Response and Management) conference that was held on November 18–21 in San Antonio. Our participation in this conference ensured that environmental health education was represented in conjunction with related education for epidemiology and laboratory science.

Integrated Pest Management

We received funding this year from the Centers for Disease Control and Prevention to put on three integrated pest management workshops.

Radon Resistant New Construction

We have also secured funding from the U.S. Environmental Protection Agency to continue to offer workshops on radon resistant new construction standards, practices, and concerns.

Epi-Ready Workshops for FDA Rapid Response Teams

On yet another successful grant front, we have funding to present seven Epi-Ready workshops this year to FDA Rapid Response teams.

Sustainability and Climate Change

As we work to expand opportunities for environmental health professionals in contemporary topics such as health effects of global climate change, sustainability, healthy communities, smart growth, etc., we are looking forward to participating in an upcoming conference in Denver (in February) on this very topic. Our ultimate goal is to land some grant funding that will enable us to establish a viable program in this area that will open up pathways for environmental health to join programs of this nature in communities across the nation.

NEHA Foodborne Illness Outbreak and Investigation Capacity Survey

Two years ago and still very much within the postrecession legacy of difficult times within environmental health programs, the Council to Improve Foodborne Outbreak Response asked NEHA to design and conduct a study concerning the impact of the recession on food safety programs. Our study is now complete. (Access the report at www.neha.org/pdf/NEHA_FBIOutbreakCapacityAssessment_ResultsReport.pdf.) Though the results weren't surprising, they were eye opening. For example, we learned that 31% of local food safety programs reduced their staffing, some by more than 30% over the last year, and 53% of programs have kept their salaries the same for two years.

Staff Changes

We've had a number of staffing changes in the Denver office. Jill Cruickshank was promoted to the important chief operating officer position, which is a first for NEHA. This is a brand new position. It allows Jill C., our former marketing and communications manager, to concentrate on the operations of our growing organization to ensure that we are functioning efficiently and successfully. We've strengthened our credentialing program with the hire of Patricia Churpakovich, who comes to us from the highly acclaimed Web MD program out of New York. Patricia is now running our credentialing program.

We've also hired TJay Gerber to assist in credentialing. TJay brings exceptional customer service skills and management experience to his position. We recently hired Erik Kosnar to help with our education development work and Clare Sinacori to replace Jill C. as our marketing and communications manager. Clare comes to us from the Colorado State Department of Wildlife and brings an extensive base of marketing experience and talent with her to the position. NEHA also brought Matt Lieber on full time to assist the marketing and communications department.

We've also said goodbye to Shelly Wallingford, Misty Duran, and Alyssa Crum, who worked as part of the credentialing team.

NEHA Budget

Finally, I thought that the membership would be interested to know that we just adopted a budget for next year of approximately $6 million, which is actually down by about $1 million from last year. This decrease primarily reflects a loss of some of our grants and the income we were earning from our Center program.

NEHA now employs 32 people together with another six people who do contract work for us. This is quite a growth from the $280,000 budget and staff of four that I saw when I first came to NEHA!

In closing, I hope that this report was enlightening and informative. We have tremendously talented and industrious staff at NEHA who have taken to heart our mission statement that involves advancing you—the environmental health professional—and the profession itself. I hope this listing speaks directly to your hearts and enables you to feel good about your NEHA membership and our dedication to you and our wonderful cause.

nfabian@neha.org

Experience the *Journal of Environmental Health (JEH)* as you've never experienced it before!

The JEH is now electronic and is being delivered to you via e-mail. This interactive version of the JEH allows you to 1) engage with the content through video; 2) quickly search for information you desire with the use of the search feature; 3) highlight passages you want to make a note of; 4) access web links and e-mail addresses found in articles, advertising, and listings; and 5) read the JEH on any computer or mobile device such as a tablet or mobile phone.

As a NEHA member, you will receive the E-Journal in addition to the hard copy—absolutely free—for all issues of the JEH from November 2013 through September 2014!

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KEYNOTE SPEAKER

NEHA Is Pleased to Announce the 2014 AEC Keynote Speaker: ***Dr. Mark Keim***

The National Environmental Health Association is pleased to announce that Mark Keim, MD, Associate Director for Science at Centers for Disease Control and Prevention (CDC), will address attendees of the 78th Annual Educational Conference (AEC) & Exhibition as the keynote speaker.

With the expanded international audience at this year's AEC, you'll want hear Dr. Keim's perspective on emerging and contemporary issues, including the far-reaching health effects of global climate change.

In addition to his current role, Dr. Keim has spent many years working for CDC in many capacities including acting associate director in the Office of Terrorism Preparedness and Emergency Response, medical officer and team leader at the International Emergency and Refugee Health Branch, and acting associate director for science in the Division of Emergency and Environmental Health Services. He is also an adjunct faculty member at the Rollins School of Public Health at Emory University.

Dr. Keim has provided consultation for the management of dozens of disasters involving the health of literally millions of people throughout the world. Dr. Keim is the author of several hundred scientific presentations, 40 journal publications, and 13 book chapters.

Dr. Keim received numerous awards for his work in CDC's emergency operations during the World Trade Center, anthrax letter, and Hurricane Katrina emergencies, as well as for leading the U.S. health sector response after the Indian Ocean tsunami.

He has been a member of the White House Subcommittee for Disaster Reduction since 2006. He served as a review editor for the United Nations Intergovernmental Panel on Climate Change from 2009 to 2011.

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There's a reason they call Las Vegas the entertainment capital of the world. Whatever your idea of a good time is, you're almost sure to find it in Las Vegas.

For more information, visit neha2014aec.org and click the "Destination" tab.

NEHA AEC Venue & Hotel

The Cosmopolitan of Las Vegas

3708 Las Vegas Boulevard South, Las Vegas, NV 89109

Discounted room rates will be available starting at $139 USD per night plus taxes and fees.

PRELIMINARY SCHEDULE

Schedule is subject to change.

Saturday / July 5	Sunday / July 6	Monday / July 7	Tuesday / July 8	Wednesday / July 9
EHAC Meeting	EHAC Meeting	Pre-Conference Workshops	Educational Sessions	Exhibition Open
IFEH Council Meeting	IFEH Faculty Forum Business Meeting		"Thank You Luncheon" for guests staying at the AEC designated hotel for two or more nights	Poster Session
	IFEH Faculty Forum & EHAC Joint Meeting	IFEH Regional Meetings	Awards Ceremony & Keynote Address	Silent Auction
	IFEH AGM Meeting	Community Event	Exhibition Grand Opening & Party	Student Research Presentations
		1st Time Attendee Workshop		Networking Luncheon
		Annual UL Event		Educational Sessions

Thursday / July 10	Friday / July 11	Saturday / July 12	Sunday / July 13
Town Hall Assembly	Credential Review & Certification Courses	Credential Review & Certification Courses	Credential Exams
Educational Sessions			
Lunch On Your Own			
President's Banquet			

IFEH = International Federation of Environmental Health

EHAC = National Environmental Health Science & Protection Accreditation Council

Registration

Registration information is available at neha2014aec.org. For personal assistance, contact Customer Service toll free at 866.956.2258 (303.756.9090 local), extension 0.

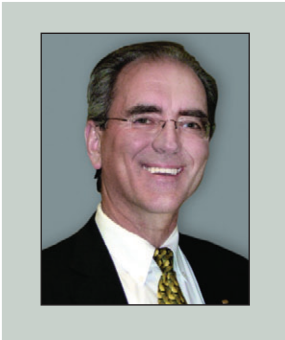
	Member	Non-Member
Full Conference Registration	$575	$735
One Day Registration	$310	$365
Student/Retired Registration	$155	$230

Stay at the designated AEC hotel—The Cosmopolitan of Las Vegas—for two or more nights and attend a free "Thank You Luncheon" on Tuesday, July 8.

Certain terms and conditions apply.

neha2014aec.org

MANAGING EDITOR'S DESK



Nelson Fabian, MS

A Sampling of NEHA Activities and Developments

Through the opportunity of this column, I've long enjoyed the chance to do two things. I've explored ideas that pertain to environmental health. I've also speculated on the meaning of all the little stories, added up, within our profession.

On occasion, as with this column, I report to the membership on some of the developments taking place within NEHA. As the person responsible for the overall operation of NEHA, I appreciate the importance of periodically reporting to the membership about what is going on within your association. So here goes!

The *Journal of Environmental Health*

The big new news with this program is, of course, that we're now providing every member with a free subscription to our E-*Journal* (in addition to the print *Journal*) from November 2013 to September 2014 while your membership is active. We hope you are enjoying it and especially the new capabilities you now have with the material we publish in it.

I also want to share with you some interesting factors that are now weighing on this special program of ours and gaining our attention.

Though the *Journal* is not a consumer publication, it is a "magazine" of sorts. As such, it too has been touched by the digital insurgency and the impact that this revolution has had on the world of publishing … and the way we learn. As publishing companies continue to come to terms with what the migration to digital means, several trends are now becoming apparent. One major trend in particular has caught my eye.

> ***I hope this listing speaks directly to your hearts and enables you to feel good about your NEHA membership and our dedication to you and our wonderful cause.***

"News" is more and more being reported on Web sites (to ensure currency) while "ideas" (and their exploration) are increasingly becoming the feature of print. How do our *Journal* and our other information delivery channels fit into this trend?

Another trend that is more specific to peer-reviewed professional journals (such as ours) is the trend toward "open access." Much has been written about how people in general believe that anything that is digital should be free. What does this mean for our *Journal* (and even the value proposition of NEHA membership) in the future?

And finally, social media has clearly become a primary source for both information and knowledge. As such, content producers like associations are looking for better ways to integrate user-generated content with more traditionally generated content. How should that play out in NEHA?

It's a fascinating world out there! Our professional staff is thinking through how best NEHA should respond to these trends. As we continue the NEHA journey and seek to remain a valuable, reliable, and relevant source of information and knowledge for you, however, we welcome, and in fact, urge you to join this journey with us. On all these fronts—and on any other you care to express an opinion on—please let us know what you think. We are posting this column and all responses to it on our blog at http://neha-org.blogspot.com.

On an entirely different front and on a topic that seldom gets discussed, I thought the membership would appreciate knowing that the *Journal* program currently operates with some 140 peer reviewers (from 14 countries) and seven technical editors who oversee the entire peer-review process for the manuscripts we publish.

The Center for Priority Based Budgeting

By action of our board, we have now decoupled the Center program from NEHA. NEHA successfully incubated this project and the Center has now launched as its own self-standing and independent entity.

Thanks to the Center program, we've learned a lot about how communities do their budgeting and how we can accentuate the importance of environmental health to policy makers. We also learned a lot about how we can work to open up new job opportunities for environmental health in contem-

continued on page 188

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