Linking Exposure Assessment Science With Policy Objectives for Environmental Justice and Breast Cancer Advocacy: The Northern California Household Exposure Study

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Objectives. We compared an urban fence-line community (neighboring an oil refinery) and a nonindustrial community in an exposure study focusing on pollutants of interest with respect to breast cancer and environmental justice.

Methods. We analyzed indoor and outdoor air from 40 homes in industrial Richmond, California, and 10 in rural Bolinas, California, for 153 compounds, including particulates and endocrine disruptors.

Results. Eighty compounds were detected outdoors in Richmond and 60 in Bolinas; Richmond concentrations were generally higher. Richmond's vanadium and nickel levels indicated effects of heavy oil combustion from oil refining and shipping; these levels were among the state's highest. In nearly half of Richmond homes, $PM_{2.5}$ exceeded California's annual ambient air quality standard. Paired outdoor–indoor measurements were significantly correlated for industry- and traffic-related $PM_{2.5}$, polycyclic aromatic hydrocarbons, elemental carbon, metals, and sulfates (r=0.54–0.92, P<.001).

Conclusions. Indoor air quality is an important indicator of the cumulative impact of outdoor emissions in fence-line communities. Policies based on outdoor monitoring alone add to environmental injustice concerns in communities that host polluters. Community-based participatory exposure research can contribute to science and stimulate and inform action on the part of community residents and policymakers. (*Am J Public Health.* 2009;99:S600–S609. doi: 10.2105/AJPH.2008.149088)

With a sprawling oil refinery in the background, Marleen Quint, Wanna Wright, and Etta Lundy stood on a hill overlooking Richmond, California, holding up a photograph of Quint's mastectomy scars.[1] The women were propelled by their breast cancer diagnoses to ask whether their own cancers as well as neighborhood problems with asthma, sore throats, rashes, other cancers, and children's development were related to chemical exposures from nearby industry and rail, truck, and marine shipping corridors. Their question is part of an emerging crossover of interests between environmental justice and breast cancer advocacy[2–5] that is driven not only by personal experiences but also by breast cancer statistics for ethnic minority women; environmental hypotheses that link the same pollutants to breast cancer and to health issues of concern in low-income, minority communities; and new partnerships between communities and scientists.[6–9]

In the United States, the breast cancer incidence rate is higher among African American women younger than 40 years than among White women in the same age group,[10,11] and mortality rates among African American women are higher in all age groups, even when access to mammography and treatment are equivalent.[12] Among older women, the incidence rate is lower in the African American population than in the White population,[11] but the gap may be closing.[13,14] Meanwhile, incidence rates are rising rapidly among US immigrants[15–17] and in industrializing nations.[18]

Environmental chemical pollutants hypothesized to cause breast cancer include some that have been associated with higher breast cancer risk in several human studies, for example polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs); animal mammary gland carcinogens, including PAHs, pesticides such as chlorothalonil, and flame retardants such as tris(2,3-dibromo-1-propyl) phosphate; and endocrine-disrupting compounds (EDCs), including bisphenol A, alkylphenols, phthalates, and pesticides such as dichlorodiphenyltrichloroethane (DDT) and pyrethroids.[19–21] Thus, the chemicals of interest in breast cancer research include urban air pollutants, industrial chemicals, and pesticides to which low-income, minority populations are disproportionately exposed.[6,22] Such exposures are also hypothesized to affect health outcomes such as premature puberty, asthma, obesity, and cognitive development that disproportionately affect low-income, minority populations.[23–26]

NORTHERN CALIFORNIA HOUSEHOLD EXPOSURE STUDY COLLABORATIVE

Through the National Institute of Environmental Health Sciences environmental justice grants program, we established a community-based participatory research (CBPR) collaboration involving the Silent Spring Institute, which focuses on the environment and women's health, especially breast cancer[27]; Communities for a Better Environment (CBE), the environmental justice organization in which Quint, Wright, and Lundy are active[28]; and faculty at Brown University and the University of California, Berkeley. We addressed breast cancer and environmental justice concerns in an exposure study that expanded the Silent Spring Institute Household Exposure Study (part of the Cape Cod Breast Cancer and Environment Study)[29] to neighborhoods bordering an oil refinery in Richmond, where CBE has an active environmental justice campaign, and rural Bolinas, California, which provided a regional comparison.

We decided to conduct an exposure study because an epidemiological breast cancer study

within Richmond probably would not have been informative, given the community's size and lack of relevant historical exposure measurements. An exposure study of compounds of toxicological concern can assess the extent of a problem and inform exposure reduction efforts.[30] We focused on household exposures because people spend 90% of their time indoors, often at home, and household environments have not been well characterized.[31,32] We tested for compounds hypothesized to affect breast cancer and additional products of oil combustion and refining that serve as indicators for the large number of uncharacterized emissions from oil refineries.

Our study included several goals related to policy, exposure science, and community education. Our policy goals were to provide data that would inform local decisions about the Richmond oil refinery, California state biomonitoring and chemicals policies,[33,34] and national debates regarding the use of EDCs in consumer products.

Our exposure science goals were to test for chemical markers of oil refinery emissions in homes, characterize the cumulative effects of emissions in an environmental justice community by measuring an exceptionally large and diverse set of pollutants from outdoor and indoor sources, assess geographic and sociodemographic differences in EDC exposures by comparing Cape Cod, Massachusetts, with an industrial neighborhood in California, and describe outdoor EDC levels. (An environmental justice community is composed of low-income or ethnic minority residents disproportionately affected by environmental pollution.) To our knowledge, no previous reports on these issues have been published.

Finally, one of our educational goals was to inform community members about important determinants of their indoor air quality. The other goal was to inform them about current scientific knowledge on potential relationships between indoor exposures and health, including breast cancer.

RESEARCH SETTING

To inform CBE's organizing and advocacy, we focused on the Liberty and Atchison Village neighborhoods of Richmond, in Contra Costa County: 66 acres that border a Chevron oil refinery and truck, rail, and marine shipping corridors[35] (see Figure S1, available as a supplement to the online version of this article at http://www.ajph.org). The area's uniform 1- and 2-story housing was constructed for shipyard workers during World War II. According to the 2000 census, the community was 61% Latino (many residents were monolingual Spanish speakers), 18% African American, and 3% Asian American; 26% of residents had incomes below the federal poverty level ($17 603 for a family of 4), and half had incomes below 200% of the poverty level.[36] Richmond has high cancer and respiratory risks associated with toxic industrial releases.[37] Contra Costa's 15% asthma prevalence rate is among the state's highest,[38] and its breast cancer incidence rate is higher than the statewide rate.[39] These statistics highlight Richmond's enhanced vulnerability to multiple pollutant exposures.

The Richmond Chevron refinery is one of the nation's largest, covering 2900 acres and processing more than 240 000 barrels of crude oil a day[40] into gasoline, jet fuel, diesel fuel, and lubricants. It employs approximately 1000 workers.[41] CBE was concerned about air pollution from flaring (venting and uncontrolled burning of gaseous emissions in routine operations and emergencies)[42] and about requested permit changes to replace and add equipment[43] that reportedly would increase overall throughput[44] and increase emission of sulfur dioxide, sulfates, and metals[45] through refining of lower grade crude oil with higher sulfur content.

FOCUS ON INDOOR POLLUTANTS WITH OUTDOOR SOURCES

Here, in our first report on the CBPR process and study design, we focus on results that pertain most directly to environmental justice. We describe the compounds detected (as an indication of cumulative impact) and pollutants with significant outdoor sources, as evidenced by higher outdoor concentrations in industrial Richmond than in rural Bolinas. We include for comparison an indoor-source chemical (dibutyl phthalate) to demonstrate the contrast between outdoor- and indoor-source compounds.

Results of additional analyses focusing on indoor-source chemicals, many of which are EDCs, will be published in a subsequent article, and analyses of questionnaire responses and refinery emergency releases as predictors of pollutant levels are under way. We previously reported dramatic geographic differences in brominated flame retardants (polybrominated diphenyl ethers), with the higher levels observed in California than other areas probably due to the state's strict furniture flammability standard.[46] We have written elsewhere about our methods for reporting personal exposure results to participants.[47,48]

METHODS

After a CBPR collaborative process[49,50] designed to consider what data would address mutual goals of the project partners, we sampled air and dust from 40 homes in Richmond and 10 in Bolinas, as well as outdoor air near each home. Samples were analyzed for industrial and traffic pollutants, such as particulates, metals, PAHs, ammonia, and sulfates, and for many EDCs, including pesticides, flame retardants, phthalates, and phenols.

CBPR Strategy

We gathered information on community health concerns, drew on CBE's relationships with public officials, held annual community meetings, and convened an advisory council that included neighborhood activists, breast cancer and biomonitoring activists, a state health official, and an academic researcher. On the basis of this input, we designed research to assess the cumulative effects and specific sources of indoor pollution originating from outdoor emissions.

The advisory council requested a comparison with rural northern California that would supplement the comparison with Cape Cod and maximize the contrast for assessing the cumulative impact of Richmond outdoor emissions. This request led to an additional community partnership with the Commonweal Biomonitoring Resource Center in Bolinas,[51] a nonindustrial coastal community. We deliberated how our results would affect refinery expansion plans. We expected to detect compounds associated with oil refining in Richmond homes; however, we were mindful that if we did not, our results might undermine CBE's refinery campaign, even though any negative findings might be due to inadequacies in our methods.

The research protocol was approved by Brown University's institutional review board

in a novel agreement that covered both academic and community-based researchers, representing a significant accomplishment for our collaborative. Traditionally, university institutional review boards do not cover outside organizations, leaving community groups with the expense of contracting with an independent review board and diminishing the academic–community partnership. All individuals with access to personally identifiable data were formally trained with respect to ethics in human subjects research.

Selection of Households for Sampling

Balancing the goals of collecting representative neighborhood data and creating opportunities to involve CBE members, we recruited 40 nonsmoking households in the Atchison Village and Liberty neighborhoods of Richmond through door knocks at randomly selected addresses (22 participants) and announcements at community meetings (18 participants). We obtained a list of all 550 eligible residential addresses from the county tax assessor's office and the management of the Liberty Village Apartments. We mailed a letter describing the study in English and Spanish to each home.

Using a randomized address list, a CBE researcher approached 132 nonvacant residences, contacting a resident at 74 (56%); 31% of contacted eligible homes participated. Six residents agreed but could not be scheduled, 3 were ineligible, and 43 declined; we were unable to contact 58 residents. We used snowball sampling to recruit 10 participants in Bolinas; the sample size was constrained by costs (for a chart of the sampling procedure, see Figure S2, available as a supplement to the online version of this article at http://www.ajph.org).

Data Collection and Chemical Analysis

CBE staff were equipped and trained to collect samples and conduct interviews. Samples were collected between June and October 2006. EDC indoor sampling and analytical methods have been described elsewhere.[29] Paired indoor and outdoor air samplers collected parallel 24-hour integrated samples. University Research Glassware (Chapel Hill, NC) personal pesticide samplers (polyurethane foam plus sorbent XAD2) were used to collect semivolatile compounds at a flow rate of approximately 8 L per minute (as described by Rudel et al.[29]). For the 42 homes sampled between August and October 2006, respirable particulate ($PM_{2.5}$, particulate matter less than 2.5 microns in diameter) samples were also collected at a flow rate of approximately 5 L per minute on Teflon filters alongside the semivolatile sampler attached to the same high-volume pump. A parallel sample was collected on a quartz filter for carbon fractions and water-soluble ions.[52,53] Field blanks and duplicate samples were collected for quality assurance and quality control purposes.

A researcher observed characteristics of the home, including room size, open and closed windows, and rugs and carpets, and interviewed participants about demographic characteristics, consumer product use, and expectations about the study. The Southwest Research Institute (San Antonio, TX) and the Desert Research Institute (Las Vegas, NV) analyzed the samples for 153 analytes, including phthalates, alkylphenols, other phenols, parabens, banned and contemporary-use pesticides, PAHs, polybrominated diphenyl ethers, PCBs, particulates, metals, water-soluble ions, carbon fractions, and ammonia.

We obtained 24-hour integrated measurements from all 15 California Environmental Protection Agency monitors where $PM_{2.5}$ speciation data were gathered during the time of our data collection[54] (Figure S3 shows monitor locations, available as a supplement to the online version of the article at http://www.ajph.org). Comparison data from state monitors were available for $PM_{2.5}$, elemental and organic carbon, sulfates, nitrates, and metals. We selected $PM_{2.5}$, elemental and organic carbon, sulfates, vanadium, nickel, and sodium for comparison because they are indicators of specific emission source categories. We calculated summary measures for August through October 2006.

Data Analysis

In addition to comparing Richmond data and Bolinas data, we compared outdoor measurements with indoor measurements and with state monitors. For each analyte, the method reporting limit was defined as the maximum of the analytical detection limit and the 90th percentile of the field blank concentrations. Values below the method reporting limit were not included in the percentage detected but were treated as estimated values to allow visualization of distributions (e.g., in box plots) and comparison of medians. We used the Fisher exact test to evaluate differences in the numbers of compounds detected between Richmond and Bolinas and the Wilcoxon rank-sum test to assess differences in pollutant levels.

To address environmental justice concerns about outdoor emissions sources in Richmond, we evaluated the contribution of outdoor sources to indoor pollution by comparing outdoor with indoor concentrations and calculating Spearman rank correlations between outdoor and indoor levels for compounds that had higher concentrations or were more frequently detected in Richmond outdoor air than in Bolinas outdoor air. For these compounds, measured or estimated values were available for at least 70% of indoor–outdoor pairs. The level of statistical significance was set at $P<.05$.

RESULTS

Participants were predominantly middle-aged women. With respect to race/ethnicity (participants were allowed to select more than one option), 41% of the participants in Richmond self-identified as Hispanic, 54% self-identified as White, and 11% selected another race/ethnicity; 38% were interviewed in Spanish. In Bolinas, none of the participants were Hispanic, 80% were White, and 40% selected another race/ethnicity. In Richmond, 37% had a college education, as compared with 100% in Bolinas (see Table S1, available as a supplement to the online version of the article at http://www.ajph.org).

Cumulative Effects

Chemical exposures in Richmond were greater than those in Bolinas. We detected 80 compounds in Richmond outdoor air and 60 in Bolinas outdoor air. Differences in indoor air were more pronounced, with 104 compounds detected in Richmond and 69 in Bolinas (detection frequencies are shown in Table S2, available as a supplement to the online version of the article at http://www.ajph.org). In the case of the 56 compounds detected in both communities, outdoor levels were significantly higher for 33 in Richmond and 1 (diethyl phthalate) in Bolinas (Wilcoxon $P\leq.05$). Median and maximum concentrations of these 33 compounds are shown in Table 1.[52,55–62] Richmond outdoor levels were significantly

TABLE 1—Outdoor and Indoor Air Concentrations ($\mu g/m^3$) and Correlations for Compounds Detected at Higher Levels in Richmond Than in Bolinas, CA: 2006

	Outdoor Air				Indoor Air				
	Richmond		Bolinas		Richmond		Bolinas		Outdoor–Indoor Correlation[a]:
Chemical	Median	Maximum	Median	Maximum	Median	Maximum	Median	Maximum	Richmond
Particulate matter and related carbon fractions[d]									
$PM_{2.5}$	10	17	5.5	9.5	11	28	7.1	18	0.54*
Elemental carbon fraction 1	0.32	0.85	0.11	0.47	0.56	2.1	0.35	1.8	0.52*
Elemental carbon fraction 2	0.15	0.57	<MRL	0.45	0.23	0.70	0.088	0.46	0.78*
Total elemental carbon	0.35	0.94	0.067	0.63	0.54	1.6	0.20	0.95	0.58*
Organic carbon fraction 1	<MRL	0.79	<MRL	1.1	2.1	6.3	1.8	3.5	0.03
Organic carbon fraction 2	0.88	1.9	<MRL	1.4	3.4	5.2	2.6	6.5	0.20
Organic carbon fraction 3	0.87	1.9	0.55	1.2	3.2	8.0	2.8	9.0	<0.01
Organic carbon fraction 4	0.32	0.77	0.11	0.61	1.1	2.9	0.87	2.3	0.01
Total organic carbon	2.1	4.8	<MRL	4.7	10	17	8.3	22	0.08
Total carbon	2.7	5.7	<MRL	5.3	11	19	8.5	23	0.14
Metals and ions[e]									
Aluminum	0.021	0.090	0.0092	0.086	0.023	0.11	0.030	0.25	0.50*
Calcium	0.090	0.37	<MRL	0.053	0.093	0.28	0.045	0.11	0.81*
Copper	<MRL	0.055	<MRL	0.0094	<MRL	0.054	<MRL	0.023	0.05
Iron	0.063	0.24	<MRL	0.034	0.055	0.32	0.028	0.15	0.79*
Lead	0.0010	0.0040	<MRL	0.0017	0.0012	0.0041	<MRL	0.0015	0.62*
Manganese	0.0020	0.0080	<MRL	0.0012	0.0015	0.0062	0.0013	0.0030	0.78*
Nitrates	1.0	3.2	0.34	0.84	0.95	3.3	0.25	1.1	0.62*
Potassium	0.051	0.11	0.018	0.062	0.050	0.21	0.033	0.11	0.46*
Sulfates	2.2	3.9	1.3	2.5	1.6	3.6	1.4	3.4	0.91*
Vanadium	0.0050	0.023	0.0018	0.0028	0.0035	0.020	0.0016	0.0023	0.93*
Polycyclic aromatic hydrocarbons[f]									
1-methylphenanthrene	0.00042	0.0010	<MRL	0.00068	0.0013	0.0041	0.0010	0.0030	0.17
2-methylphenanthrene	0.00076	0.0020	<MRL	0.0012	0.0021	0.0060	0.0015	0.0047	0.30
3-methylphenanthrene	0.00069	0.0020	<MRL	0.0012	0.0019	0.0066	0.0017	0.0048	0.27
9-ethylphenanthrene	0.00034	0.00091	<MRL	0.00046	0.0013	0.0043	0.0012	0.0032	0.27
Acenaphthene	0.0048	0.011	0.00081	0.0046	0.0061	0.029	0.0034	0.0089	0.57*
Fluoranthene	0.0010	0.0027	<MRL	0.0038	0.00098	0.012	0.00068	0.0015	0.49*
Fluorene	0.0055	0.011	0.0011	0.0056	0.0081	0.028	0.0052	0.012	0.54*
Phenanthrene	0.0086	0.017	0.0022	0.015	0.012	0.044	0.0097	0.018	0.48*
Pyrene	0.00063	0.0019	<MRL	0.0019	0.00090	0.028	0.00071	0.00097	0.22
Phthalates[g]									
Bis(2-ethylhexyl) adipate	0.0023	0.0087	0.0015	0.0021	0.032	0.075	0.023	0.069	0.13
Bis(2-ethylhexyl) phthalate	0.017	0.24	<MRL	0.024	0.079	0.21	0.056	0.11	–0.02
Other									
Ammonia[b]	3.0	32	0.67	2.0	24	180	7.9	32	0.03
O-phenylphenol[c]	0.0012	0.0048	0.00052	0.0010	0.0083	0.061	0.013	0.019	–0.04

Note. MRL = method reporting limit (defined as the maximum of the analytical detection limit and the 90th percentile of the field blanks. Estimated values (i.e., quantified by the laboratory but below the MRL) were used in the calculation of summary statistics. Sources listed for each chemical class are based on cited literature and not specifically characterized in this study. Included are compounds measured at significantly ($P \leq .05$; Wilcoxon rank-sum test) higher concentrations in Richmond outdoor air than in Bolinas outdoor air.

[a]Spearman rank correlation coefficients.

[b]Sources: petroleum refining, agricultural activity, human and pet metabolic processes, and household cleaning products.[57,58]

[c]Sources: pesticides, disinfectants, preservatives, and other uncharacterized sources.[60,62]

[d]Sources: combustion sources including traffic, home heating, cigarette smoke, cooking, and candle burning.[52,55,56]

[e]Sources: petroleum refining, shipping, power generation, and other industrial activity; traffic; and crustal/soil.[52]

[f]Sources: combustion sources including traffic, power generation, home heating, cigarette and incense smoke, and cooking.[59]

[g]Sources: plastics, consumer products including cosmetics and pesticides, and other uncharacterized sources.[60,61]

*$P \leq .05$.

higher for refinery-related sulfates, vanadium, and ammonia and other industry- and transportation-related pollutants, including $PM_{2.5}$, PAHs, carbon fractions, and metals. Outdoor levels of 2 phthalates and *o*-phenylphenol were also significantly higher in Richmond (detailed results for EDCs will be published in an upcoming article).

Indoor air in nearly half of Richmond homes exceeded California's annual ambient air quality standard for $PM_{2.5}$, often considered an aggregate measure of air pollution; indoor levels were higher than outdoor levels in both communities (Figure 1). EDCs were detected more frequently indoors than outdoors in both communities.

Indoor Penetration of Outdoor Pollutants

To examine the impact of outdoor pollutant emissions on indoor air, we evaluated relationships between paired outdoor and indoor measurements for the 33 chemicals measured at higher levels outdoors in Richmond. Figure 2 illustrates outdoor and indoor concentrations for an example outdoor-source and an example indoor-source pollutant. In the case of sulfates, a frequent by-product of industrial pollution with few indoor sources (Figure 2ab), there was a strong correlation ($r=0.92$; $P<.001$) between paired outdoor and indoor measurements, and outdoor concentrations were consistently higher than indoor concentrations, indicating that outdoor sulfates were penetrating indoors. Strong correlations between outdoor and indoor concentrations were observed for vanadium, selenium, calcium, iron, and manganese (Spearman ρ range: 0.7–0.9; $P<.001$), and outdoor concentrations were higher than indoor concentrations.

Outdoor–indoor levels and correlations for $PM_{2.5}$, many of the PAHs (e.g., acenaphthene, fluorene, and fluoranthene), lanthanum, and elemental carbon (Spearman ρ range: 0.4–0.6; $P<.05$) suggested both outdoor and indoor sources and indicated that outdoor air is an important source of these pollutants indoors. By contrast, there were high indoor levels of di-*n*-butyl phthalate, commonly found in personal care products, and a lack of correlation between paired outdoor and indoor measurements, indicating that indoor sources dominate (Figure 2cd). A similar pattern was observed for other EDCs, including bis(2-ethylhexyl) phthalate, bis(2-ethylhexyl) adipate, and *o*-phenylphenol, and for organic carbon fractions, ammonia, and some PAHs (e.g., pyrene and methylphenanthrenes).

Comparison With State Monitors

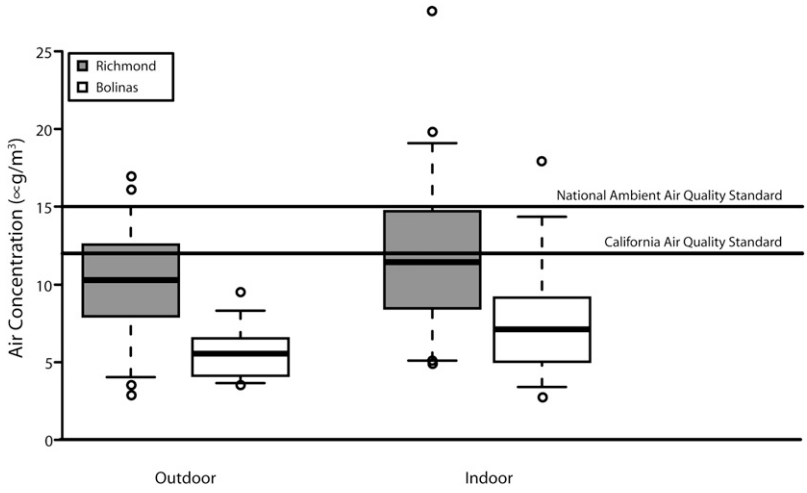
In another approach to analyzing the influence of local and regional outdoor sources, we compared outdoor measurements in Richmond and Bolinas with California Environmental Protection Agency monitoring data for the same time period as our study. For vanadium and nickel (which are markers of heavy oil combustion, especially from oil refineries and marine ports[52,63,64]), Richmond was near the top of the distribution, with the second-highest 95th percentile concentration. For sulfates, which tend to be influenced by both regional and local sources, including power plants, automobiles, and oil refineries, Richmond levels were in the top third.

In the case of pollutants such as $PM_{2.5}$, elemental and organic carbon, and nitrates deriving primarily from mobile sources, Richmond was in the lower half of the distribution. For sodium, a marker of ocean air, Richmond levels were among the highest of all monitoring sites. In Bolinas, levels for all pollutants were low, whereas sodium levels were comparable to those in Richmond. Results for vanadium, nickel, $PM_{2.5}$, and sulfates are shown in Figure 3, ordered according to 95th percentile concentrations.

DISCUSSION

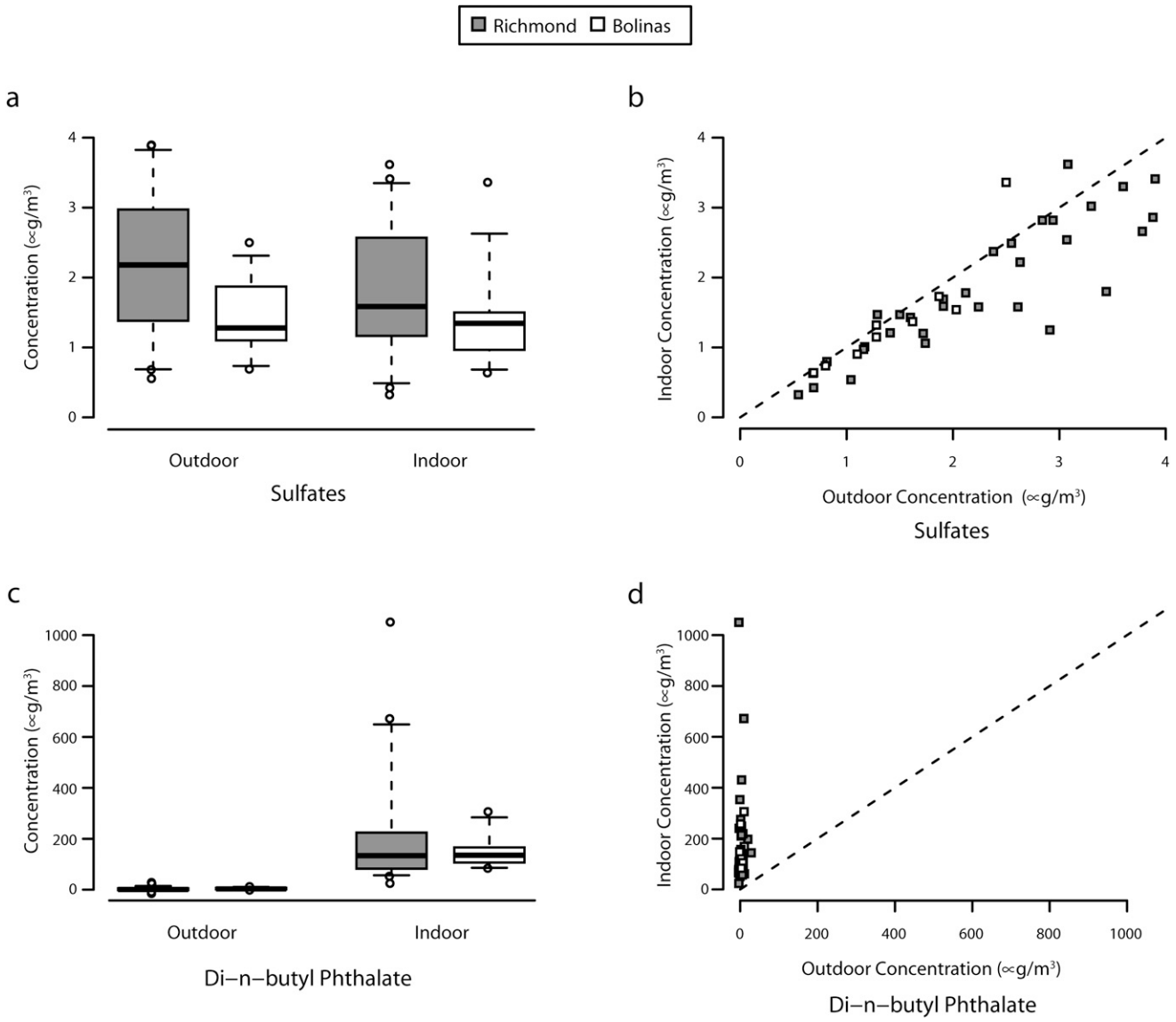
Our results provide evidence regarding 3 important environmental justice concerns: the character and magnitude of cumulative exposures in urban fence-line communities (communities that neighbor polluting facilities), the limitations of outdoor ambient monitoring as an indicator of personal exposure, and the impact of specific local sources on air quality in proximate neighborhoods. As expected, more pollutants and higher outdoor concentrations were detected in Richmond than in Bolinas. Heavy oil combustion was a more prominent factor than traffic in differences between the 2 communities. Despite high traffic in Richmond, outdoor concentrations of $PM_{2.5}$ and traffic-related pollutants were in the low half of the range reported by state monitors, perhaps as a result of meteorological effects of the study neighborhood's proximity to the coast.

By contrast, Richmond levels of nickel and vanadium (known to come from heavy oil combustion, especially in refinery operations



Note. Solid lines are medians; boxes are interquartile ranges; vertical lines are 5th and 95th percentiles; circles are extreme data points below the 5th percentile and above the 95th percentile; and horizontal dotted lines represent annual federal and state ambient air quality standards for $PM_{2.5}$.

FIGURE 1—Levels of fine particulate matter ($PM_{2.5}$) in homes in Richmond and Bolinas, CA: 2006.



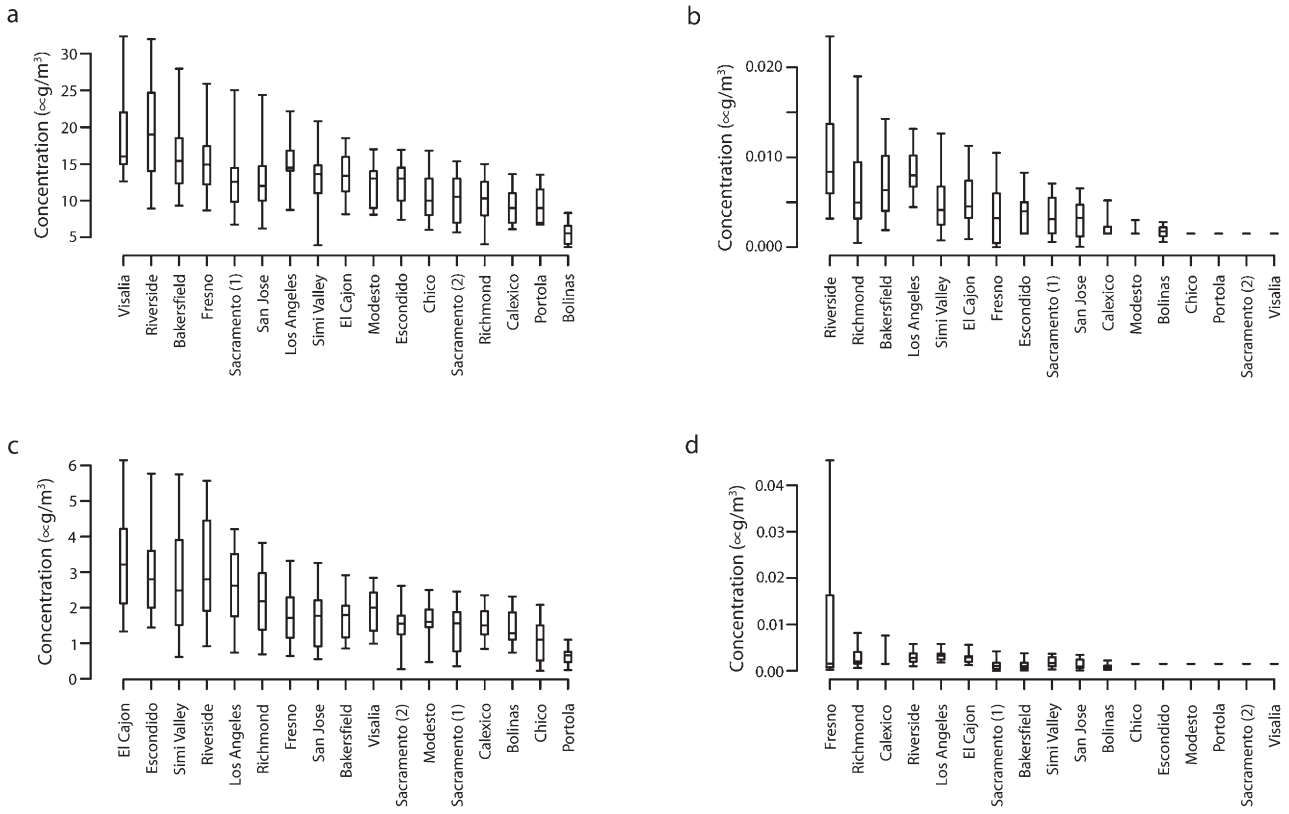
Note. Panels a and c are box plots comparing distributions for sulfates and di-*n*-butyl phthalate, respectively. Panels b and d show the correlations between indoor and outdoor concentrations across both communities. The dotted line represents 1:1.

FIGURE 2—Relationships between (a) sulfates, (b) indoor and outdoor concentrations of sulfates, (c) di-*n*-butyl phthalate, and (d) indoor and outdoor concentrations in di-*n*-butyl phthalate: Richmond and Bolinas, CA, 2006.

and marine shipping) were among the highest in the state. These compounds, along with sulfates (also associated with refineries), penetrated into Richmond homes, as demonstrated by correlations between outdoor and indoor concentrations. Health studies have shown that sulfates, nickel, and vanadium are some of the most harmful $PM_{2.5}$ components.[65,66] Furthermore, these compounds are indicators of probable exposures to hundreds of unmeasured compounds given that refinery emissions are complex and poorly characterized.[67,68]

The significant correlations we observed between outdoor and indoor levels of $PM_{2.5}$, sulfates, and other pollutants are consistent with the results of other studies showing that outdoor air pollution is an important determinant of indoor exposures.[31,59,69,70] The finding that local outdoor emissions penetrate indoors bears directly on Richmond refinery permits for activities that increase or decrease outdoor emissions, and our observations have implications for facility reviews elsewhere as well.

The much higher levels of pollutants indoors than outdoors indicate that traditional environmental impact assessments based on outdoor air quality are inadequate to represent personal exposures. California's ambient air quality standard is not intended to be applied to indoor air; however, it is of concern that



Note. Monitor locations are ranked according to the 95th percentile concentration in order of highest to lowest from left to right. Solid lines are medians; boxes are interquartile ranges; and vertical lines are 5th and 95th percentiles.

FIGURE 3—Comparison of study site outdoor air pollution levels in Richmond and Bolinas, CA, with California state monitors, 2006.

nearly half of Richmond homes exceeded this standard for $PM_{2.5}$ during the summer, when outdoor $PM_{2.5}$ levels are markedly lower than in winter. Epidemiological studies have consistently linked this pollutant to respiratory and cardiovascular problems, including premature death.[71] In addition, the PAHs and other compounds we detected may be associated with breast cancer.[20,21] Socioeconomic stressors in Richmond may amplify the detrimental health effects of chemicals we observed.[8,72]

Our results also have implications for policies concerning EDCs in consumer products. Outdoor levels were lower than and not correlated with indoor levels for chemicals coming primarily from consumer products, such as di-*n*-butyl phthalate. We observed few differences in EDC levels between our 2 markedly different communities. These results suggest that consumer products contribute substantially to indoor air quality and indicate the need for state or national remedies, such as the efforts of breast cancer organizations to secure proactive chemical policies and launch consumer campaigns to reduce the use of EDCs.[73,74]

Limitations of our study include the small number of homes sampled in Bolinas. Also, as a result of financial constraints, we sampled each home only once. Multiseason sampling would better characterize long-term, typical exposures and capture higher wintertime $PM_{2.5}$ levels. We were unable to collect samples directly representing emissions from refineries or other sources to compare with household contaminant profiles. In addition, although our study focused on a poor, largely Latino community, members of racial/ethnic minority groups and less educated residents were underrepresented in our sample. Finally, given the large number of comparisons of individual chemicals, some of our findings may be attributable to chance.

The strengths of the study include the use of a standard protocol in Richmond and Bolinas, the inclusion of paired indoor and outdoor samples and a broad range of analytes, comparability with state monitoring data, collaboration between diverse academic and community partners, and attention to individual and community communications regarding the study. Unlike environmental justice investigations of industrial and transportation pollution that typically rely on ambient air monitoring or facility emissions data, we collected unique local data on personal exposures in the home.

Public Health Applications

During our study, refinery permit changes were proposed that could increase harmful pollutant emissions[45] in Richmond via the refining of higher sulfur crude oil. CBE mobilized testimony against the plan before the Richmond Planning Commission and the city council, urging

them to consider the existing cumulative pollutant burdens documented in our study. News stories highlighted our results,[75,76] and city council members asked us to testify.[77] Study participants used their individual data and aggregate results in their own testimony, vividly demonstrating how our study helped activate and expand community engagement in environmental justice issues. At community meetings, discussions spontaneously turned to ways to use results to negotiate health protections from the oil company.

Testimony and media coverage of our findings led the Richmond Planning Commission to attempt to restrict high-sulfur crude oil refining. However, as gasoline prices climbed and the company offered Richmond $60 million in mitigation benefits, the city council reversed the planning commission's recommendation and approved the Chevron proposal in July 2008.[44] Thus, although our study influenced deliberations, the company's socioeconomic and political muscle in this cash-strapped city wielded a stronger influence. Later, the November 2008 election of new council members changed the balance again, and Richmond residents also passed a ballot measure that would require Chevron to pay the city an annual business license fee estimated at $26.5 million.

Although we cannot yet assess the significance of this study for CBE, we now have empirical results to support concerns about the effects of refinery emissions, and we know that CBE valued the study's process. As CBE staff conducted interviews and set up sampling equipment, the study helped demystify science by moving the data-gathering process into people's homes. That experience encouraged community members to think in new ways about sources of chemicals around them. These discussions enabled CBE to connect its organizing work with technical analysis—each central to environmental justice—and may strengthen CBE's long-term organizing and advocacy capacity.

This experience illustrates the CBPR view that both scientific outcomes and the research process are important. The future of the refinery expansion continues to unfold as CBE pursues a long-term mobilization effort that includes disseminating results from our study, engaging in litigation, and conducting a health symptoms survey with other neighborhood and environmental justice groups.

Conclusions

Environmental justice assessments should consider indoor exposures from local polluters. In this study, we found that cumulative air pollution burdens were more pronounced indoors than outdoors in an urban industrial environmental justice community in comparison with a rural community. Indoor air in nearly half of the environmental justice community homes in our study exceeded the California ambient air quality standard for respirable particulates, even though the residents were nonsmokers. High levels of contaminants associated with oil refining and marine shipping were detected both outdoors and indoors. Participation in this CBPR study mobilized and supported community efforts to block permits for the neighboring oil refinery. Our results also can inform a variety of individual- and policy-level exposure reduction efforts and the design of future studies focusing on air pollutants and breast cancer and other health outcomes. ■

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Contributors

J. Green Brody, R. Morello-Frosch, P. Brown, C. Pérez, and R.A. Rudel planned and supervised the study. J. Green Brody led the writing, with substantial contributions from the other authors. A. Zota conducted the statistical analysis and reported results to participants. All of the authors helped to conceptualize ideas and review drafts of the article.

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Human Participant Protection

The research protocol was approved by Brown University's institutional review board. Participants provided informed consent prior to initiation of sample collection in their homes.

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ERRATUM

In: Brody JG, Morello-Frosch R, Zota A, Brown P, Pérez C, Rudel RA. Linking exposure assessment science with policy objectives for environmental justice and breast cancer advocacy: the Northern California Household Exposure study. *Am J Public Health.* 2009;99(S3):S600–S609. doi: 10.2105/AJPH.2008.149088.

Figures and tables were improperly edited. On page S603, Table 1 should read:

TABLE 1—Outdoor and Indoor Air Concentrations (μg/m^3) and Correlations for Compounds Detected at Higher Levels in Richmond Than in Bolinas, CA: 2006

	Outdoor Air				Indoor Air				Outdoor–Indoor Correlation[a]:
	Richmond		Bolinas		Richmond		Bolinas		
Chemical	Median	Maximum	Median	Maximum	Median	Maximum	Median	Maximum	Richmond
Particulate matter and related carbon fractions[d]									
$PM_{2.5}$	10	17	5.5	9.5	11	28	7.1	18	0.54*
Elemental carbon fraction 1	0.32	0.85	0.11	0.47	0.56	2.1	0.35	1.8	0.52*
Elemental carbon fraction 2	0.15	0.57	< MRL	0.45	0.23	0.70	0.088	0.46	0.78*
Total elemental carbon	0.35	0.94	0.067	0.63	0.54	1.6	0.20	0.95	0.58*
Organic carbon fraction 1	< MRL	0.79	< MRL	1.1	2.1	6.3	1.8	3.5	0.03
Organic carbon fraction 2	0.88	1.9	< MRL	1.4	3.4	5.2	2.6	6.5	0.20
Organic carbon fraction 3	0.87	1.9	0.55	1.2	3.2	8.0	2.8	9.0	< 0.01
Organic carbon fraction 4	0.32	0.77	0.11	0.61	1.1	2.9	0.87	2.3	0.01
Total organic carbon	2.1	4.8	< MRL	4.7	10	17	8.3	22	0.08
Total carbon	2.7	5.7	< MRL	5.3	11	19	8.5	23	0.14
Metals and ions[e]									
Aluminum	0.021	0.090	0.0092	0.086	0.023	0.11	0.030	0.25	0.50*
Calcium	0.090	0.37	< MRL	0.053	0.093	0.28	0.045	0.11	0.81*
Copper	< MRL	0.055	< MRL	0.0094	< MRL	0.054	< MRL	0.023	0.05
Iron	0.063	0.24	< MRL	0.034	0.055	0.32	0.028	0.15	0.79*
Lead	0.0010	0.0040	< MRL	0.0017	0.0012	0.0041	< MRL	0.0015	0.62*
Manganese	0.0020	0.0080	< MRL	0.0012	0.0015	0.0062	0.0013	0.0030	0.78*
Nitrates	1.0	3.2	0.34	0.84	0.95	3.3	0.25	1.1	0.62*
Potassium	0.051	0.11	0.018	0.062	0.050	0.21	0.033	0.11	0.46*
Sulfates	2.2	3.9	1.3	2.5	1.6	3.6	1.4	3.4	0.91*
Vanadium	0.0050	0.023	0.0018	0.0028	0.0035	0.020	0.0016	0.0023	0.93*
Polycyclic aromatic hydrocarbons[f]									
1-Methylphenanthrene	0.00042	0.0010	< MRL	0.00068	0.0013	0.0041	0.0010	0.0030	0.17
2-Methylphenanthrene	0.00076	0.0020	< MRL	0.0012	0.0021	0.0060	0.0015	0.0047	0.30
3-Methylphenanthrene	0.00069	0.0020	< MRL	0.0012	0.0019	0.0066	0.0017	0.0048	0.27
9-Methylphenanthrene	0.00034	0.00091	< MRL	0.00046	0.0013	0.0043	0.0012	0.0032	0.27
Acenaphthene	0.0048	0.011	0.00081	0.0046	0.0061	0.029	0.0034	0.0089	0.57*
Fluoranthene	0.0010	0.0027	< MRL	0.0038	0.00098	0.012	0.00068	0.0015	0.49*
Fluorene	0.0055	0.011	0.0011	0.0056	0.0081	0.028	0.0052	0.012	0.54*
Phenanthrene	0.0086	0.017	0.0022	0.015	0.012	0.044	0.0097	0.018	0.48*
Pyrene	0.00063	0.0019	< MRL	0.0019	0.00090	0.028	0.00071	0.00097	0.22
Phthalates[g]									
Bis(2-ethylhexyl) adipate	0.0023	0.0087	0.0015	0.0021	0.032	0.075	0.023	0.069	0.13
Bis(2-ethylhexyl) phthalate	0.017	0.24	< MRL	0.024	0.079	0.21	0.056	0.11	−0.02
Other									
Ammonia[b]	3.0	32	0.67	2.0	24	180	7.9	32	0.03
o-Phenylphenol[c]	0.0012	0.0048	0.00052	0.0010	0.0083	0.061	0.013	0.019	−0.04

Note. MRL = method reporting limit (defined as the maximum of the analytical detection limit and the 90th percentile of the field blanks. Estimated values (i.e., quantified by the laboratory but below the MRL) were used in the calculation of summary statistics. Sources listed for each chemical class are based on cited literature and not specifically characterized in this study. Included are compounds measured at significantly ($P \leq .05$; Wilcoxon rank-sum test) higher concentrations in Richmond outdoor air than in Bolinas outdoor air.

[a]Spearman rank correlation coefficients.

[b]Sources: petroleum refining, agricultural activity, human and pet metabolic processes, and household cleaning products.[57,58]

[c]Sources: pesticides, disinfectants, preservatives, and other uncharacterized sources.[60,62]

[d]Sources: combustion sources including traffic, home heating, cigarette smoke, cooking, and candle burning.[52,55,56]

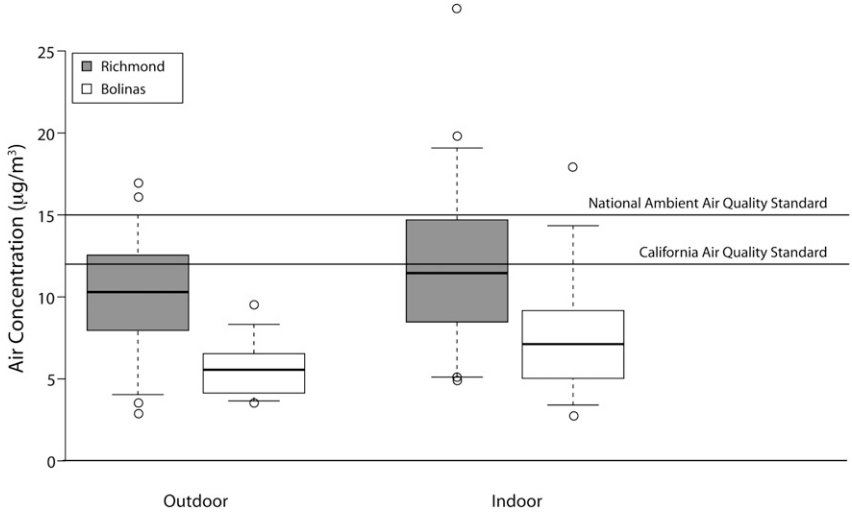
[e]Sources: petroleum refining, shipping, power generation, and other industrial activity; traffic; and crustal/soil.[52]

[f]Sources: combustion sources including traffic, power generation, home heating, cigarette and incense smoke, and cooking.[59]

[g]Sources: plastics, consumer products including cosmetics and pesticides, and other uncharacterized sources.[60,61]

*$P \leq .05$.

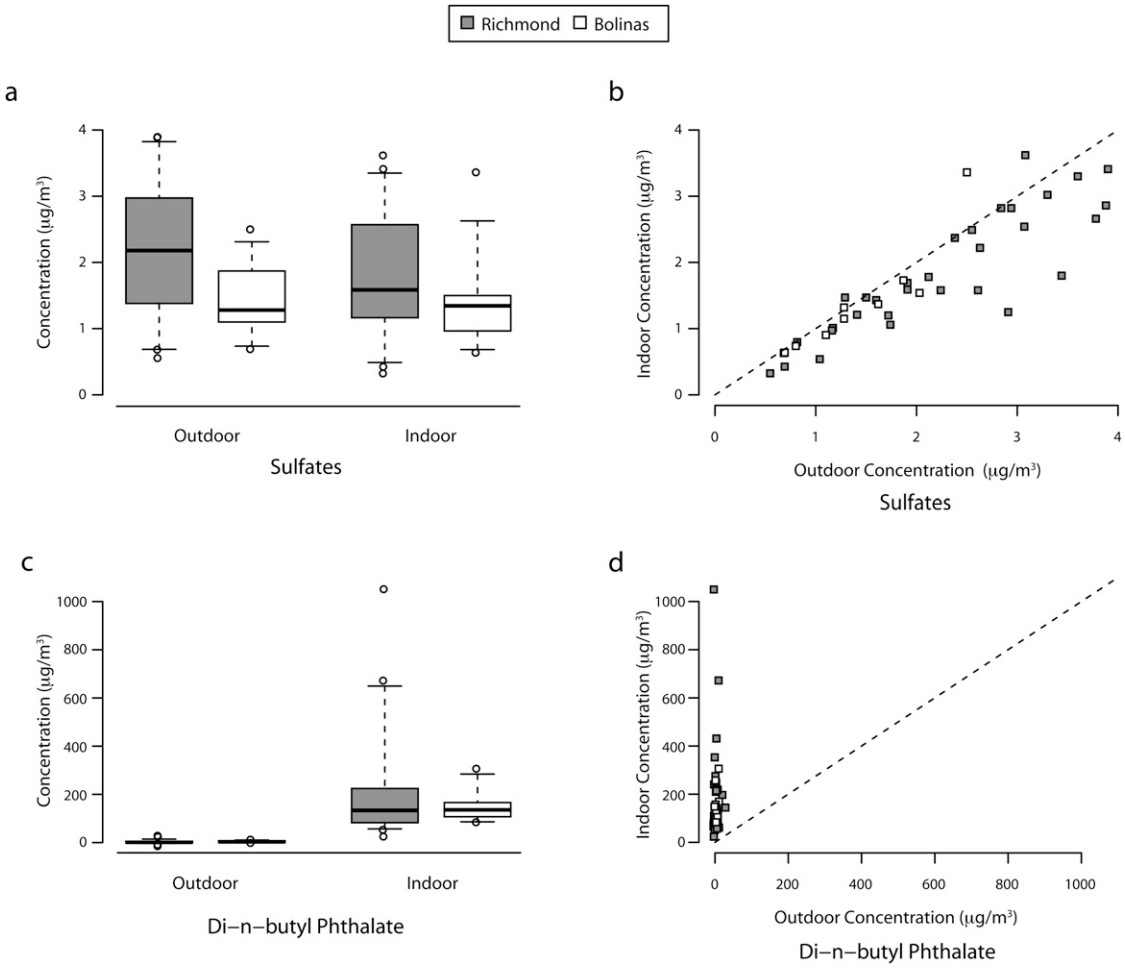
On page S604, Figure 1 should be:



Note. Solid lines are medians; boxes are interquartile ranges; vertical lines are 5th and 95th percentiles; circles are extreme data points below the 5th percentile and above the 95th percentile; and horizontal dotted lines represent annual federal and state ambient air quality standards for $PM_{2.5}$.

FIGURE 1—Levels of fine particulate matter ($PM_{2.5}$) in homes in Richmond and Bolinas, CA: 2006.

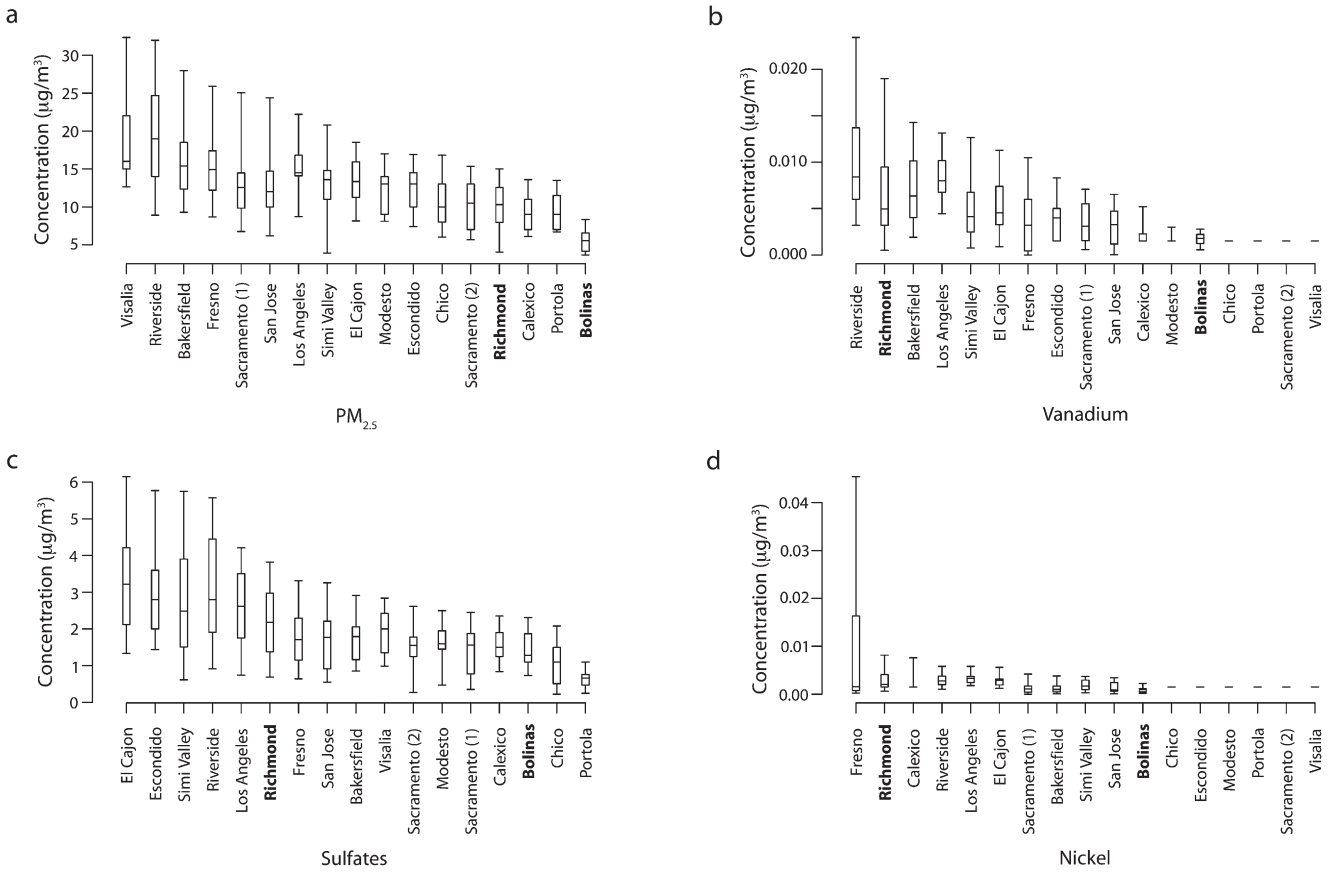
On page S605, Figure 2 should be:



Note. Panels a and c are box plots comparing distributions for sulfates and di-*n*-butyl phthalate, respectively. Panels b and d show the correlations between indoor and outdoor concentrations across both communities. The dotted line represents 1:1.

FIGURE 2—Relationships between (a) sulfates, (b) indoor and outdoor concentrations of sulfates, (c) di-*n*-butyl phthalate, and (d) indoor and outdoor concentrations in di-*n*-butyl phthalate: Richmond and Bolinas, CA, 2006.

On page S606, Figure 3 should be:



Note. Monitor locations are ranked according to the 95th percentile concentration in order of highest to lowest from left to right. Solid lines are medians; boxes are interquartile ranges; and vertical lines are 5th and 95th percentiles.

FIGURE 3—Comparison of Richmond and Bolinas, CA, with California state monitor outdoor air pollution levels of (a) $PM_{2.5}$, (b) vanadium, (c) sulfates, and (d) nickel: 2006.

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