PROCEEDINGS OF THE 2015 AEHS FOUNDATION ANNUAL INTERNATIONAL CONFERENCES

Volume 21

The 25th Annual International Conference on Soil, Water, Energy and Air,
March 23 – 26, 2015
San Diego, CA

The 31st Annual International Conference on Soils, Sediments, Water, and Energy,
October 19 – 22, 2015
Amherst, MA

PROCEEDINGS OF THE 2015 AEHS FOUNDATION ANNUAL INTERNATIONAL CONFERENCES

Volume 21

Site Characterization
Risk Assessment
Remediation
Sustainability
Technology

Selected manuscripts from

25th Annual International West Coast Conference on
Soil, Water, Energy, and Air
San Diego, California
March 23 – 26, 2015

31st Annual International East Coast Conference on
Soils, Sediments, Water, and Energy
Amherst, Massachusetts
October 19 – 22, 2015

Edited by:
Paul T. Kostecki
Christopher Teaf
Edward J. Calabrese

ISBN: 978-0-9888932-4-5

AEHS Foundation
150 Fearing Street
Amherst, MA 01002
www.AEHSFoundation.org

Dedication

We dedicate these Conference Proceedings to Dr. David Ludwig, a true friend of these international conferences, and 2013 recipient of a Lifetime Achievement Award from the East Coast Conference. Dave was a remarkable, broadly based, modern Renaissance man: scientist, teacher, writer, herpetologist, father, husband, and musician. Rock On, Viper! You will be missed, but your legacy will live on…

Table of Contents

REMEDIATION

SUSTAINABILITY

TECHNOLOGY

Contributing Authors

M.S. Al-Suwaiyan, Civil and Environmental Engineering Department, King Fahd University of Petroleum and Minerals, Dhahran, Saudi Arabia
Nakisa Alborz, Interdisciplinary Engineering Department, Wentworth Institute of Technology, Boston, Massachusetts
John Boilard, Northeast Test Consultants, 587 Spring Street, Westbrook, Maine 04092
Seyit Alperen Çeltek, Department of Energy Systems Engineering, Karamanoglu Mehmetbey University, Karaman, Turkey 70100
Gregory W. Council, Tetra Tech, 1165 Sanctuary Parkway, Suite 270, Alpharetta, Georgia 30009
Douglas J. Covert, Hazard Substance & Waste Management Research, Inc., 2976 Wellington Circle West, Tallahassee, Florida 32309
H. R. Dankwa, CSIR-Water Research Institute, Accra, Ghana
David Falatko, Innovative Engineering Solutions, Inc., 95A Ocean Street, South Portland, Maine 04106
Sami A. Fam, Innovative Engineering Solutions, Inc., 25 Spring Street, Walpole, Massachusetts 02081
J. Huang, College of Environmental Science and Engineering, Hohai University, Nanjing, China 210098
Talaat Ijaz, Tetra Tech, One Salem Square, 295 Route 22 East, Suite 104E, Whitehouse, New Jersey 08889
Mehmet Numan Kaya, Department of Mechanical Engineering, Karamanoglu Mehmetbey University, Karaman, Turkey 70100
Faruk Köse, Department of Mechanical Engineering, Selcuk University, Aladdin Keykubat Campus, Konya, Turkey
Matthew W. Muzzy, Sevee & Maher Engineers, Inc., 4 Blanchard Road, P.O. Box 84A, Cumberland, Maine 04021
Linda A. Nuamah, College of Environmental Science and Engineering, Hohai University, Nanjing, China 210098
Scott C. Simpson, Tetra Tech, 1165 Sanctuary Parkway, Suite 270, Alpharetta, Georgia 30009
Christopher M. Teaf, Center for Biomedical and Toxicological Research, Florida State University, 2035 E. Dirac Drive, Tallahassee, Florida 32310
Kevin W. Teaf, Hazardous Substance & Waste Management Research, Inc., 2976 Wellington Circle West, Tallahassee, Florida 32309
Muhammed Emin Tolu, Department of Mechanical Engineering, Karamanoglu Mehmetbey University, Karaman, Turkey 70100

Foreword

In the spring and fall of each year, a pair of complementary international symposia grace the U.S. West Coast (San Diego, CA) and East Coast (Amherst, MA), respectively. San Diego in the spring and Amherst in the fall are extraordinary destinations that annually host a broad range of scientists and environmental practitioners, many of whom return year after year for what have become highly anticipated events.

With the West Coast Annual Conference well into its third decade at age 25, and the East Coast now into its 32nd year, the participants meet to present, understand, challenge and implement a wide variety of environmental investigations, solutions, and approaches that have been successful in one context or another around the U.S. and the world. The Conferences exhibit a remarkable combination of scientific diversity, represented in 2015 by a total over 400 platform and poster presentations, 21 participatory workshops on highly specialized areas of technical interest, and featuring nearly 100 commercial exhibitors of innovative environmental and analytical services and products. It is fascinating to watch and listen as those within a subject area bat around extremely fine technical points and new paradigms, while cross-disciplinary conversations often yield new insights into old problems. The discussions and technical sessions chew on the results of yet another year's worth of struggles to identify, assess, and manage subtle but critical environmental issues. One only needs to get knee deep in the daily discussions to see clearly why public organizations, governments, regulatory agencies, and companies contribute skills, time, funds, and life experiences to understanding and managing the only environment we have.

A number of categories of excellent papers are presented in this compendium, including from the fields of Site Characterization, Risk Assessment, Remediation, Technology, and Sustainability. It is a pleasure to present this group to you, and we look forward to revisiting San Diego and Amherst yet again this coming year with other biologists, chemists, engineers, geologists, planners, toxicologists, regulatory personnel, other scientists, attorneys and those in related fields who are professionally working to understand and improve our natural environment.

Dr. Edward Calabrese
Dr. Paul Kostecki
Dr. Christopher Teaf

April, 2016

About the Editors

Dr. Paul T. Kostecki's professional career has focused on research, education and training in environmental contamination with an emphasis on human and ecological risk assessment and risk management of soils. His work includes soil ingestion estimates for children and adults; establishment of scientifically sound cleanup levels for soil; bioavailability of soil contaminants; fish as toxicological models for contamination assessment; and assessment and management of petroleum contaminated soils. Dr. Kostecki has developed and conducted over 55 conferences, workshops and courses both nationally and internationally, and has made presentations at over 100 national and international meetings. Since 1985, his conference at the University of Massachusetts Amherst on Contaminated Soils, Sediments and Water has attracted over 10,000 environmental professionals from over 50 countries. Dr. Kostecki has published over 100 articles and reports, co-edited/co-authored 35 Books and secured over $15M in research support.

Dr. Kostecki co-created the Association for Environmental Health and Sciences (AEHS) in 1989 and served as its Executive Director until 2009. In 2009, he established the AEHS Foundation. He helped found Amherst Scientific Publishers and co-created seven peer-reviewed journals: Journal of Soil and Sediment Contamination (1990); Human and Ecological Risk Assessment (1994); Journal of Phytoremediation (1998); Journal of Environmental Forensics (1999); Journal of Children's Health (2003); Non-Linearity Journal (2003); and Journal of Medical Risks (2004). In addition, Dr. Kostecki co-created the International Society for Environmental Forensics in 2002.

Dr. Kostecki served as Vice Provost for Research and Vice Chancellor for Research and Engagement at the University of Massachusetts Amherst from 2003 to 2009. He served as Special Advisor for the Clean Energy China Initiative, Office of the President, University of Massachusetts from 2009–2011. He briefly left the University of Massachusetts Amherst to establish the online education program for Simmons College, Boston, MA (2011 -2012). He is presently Professor Emeritus in the School of Public Health and Health Sciences, University of Massachusetts, Amherst.

Dr. Christopher M. Teaf is a Board-certified toxicologist with broad experience in evaluation of potential effects from chemical exposures related to industrial facilities, agriculture, waste management facilities, power generation, educational institutions, and products in general

commerce. Dr. Teaf has served on the faculty of the Center for Biomedical & Toxicological Research at Florida State University since 1979, and as Director of Toxicology for Hazardous Substance & Waste Management Research since 1985.

Chris' areas of interest include risk assessments under environmental and occupational elements of federal, state or local regulations, risk communication, and development of risk-based targets to guide remedial actions. He has extensive experience in evaluation of environmental fate and potential health effects from petroleum, solvents, metals, pesticides, pharmaceuticals, biological agents (e.g., mold, microbes) and physical agents (e.g., particulates, asbestos. For over 30 years, he has directed or conducted research in environmental and occupational toxicology for the World Health Organization, NATO, U.S. EPA, U.S. Air Force, U.S. Department of Agriculture (USDA), Florida Department of Environmental Protection, Florida Department of Health, Florida Department of Community Affairs, and Agency for Toxic Substances & Disease Registry (ATSDR), among others. He served as Toxicologist for the Florida Landfill Technical Advisory Group and the state Petroleum Technical Advisory Committee. He served on the Florida Governor's Financial and Technical Advisory Committee and was Chair of the Toxic Substances Advisory Council for the Florida Department of Labor. Chris has organized and taught many graduate and undergraduate courses and technical seminars for presentation to universities as well as international, federal, state and local agencies. He has served as Chair of the Dog Island Conservation District since 2004.

Dr. Teaf has served on editorial boards or as peer reviewer for a variety of journals and is Senior Editor for Human Health of the international journal *Human & Ecological Risk Assessment*. In addition to training, research and advisory services to many environmental agencies and private sector firms, he has provided environmental and toxicological services to the U.S. Attorney, Florida State Attorney, and Attorneys General of FL, OK, and WA. Chris has been qualified as an expert in federal and state courts, as well as administrative proceedings, in a number of states regarding toxicology, health risk assessment, and environmental chemistry.

Dr. Edward J. Calabrese is a Professor of Toxicology at the University of Massachusetts, School of Public Health and Health Sciences, Amherst. Dr. Calabrese has researched extensively in the area of host factors affecting susceptibility to pollutants, and is the author of over 750 papers in scholarly journals, as well as more than 10 books, including Principles of Animal Extrapolation; Nutrition and Environmental Health, Vols. I and II; Ecogenetics; Multiple Chemical Interaction; Air Toxics and Risk Assessment; and Biological Effects of Low Level Exposures to Chemical and Radiation. Along with Mark Mattson (NIH) he is a co-editor of the recently published book entitled Hormesis: A Revolution in Biology, Toxicology and Medicine. He has been a member of the U.S. National Academy of Sciences and NATO Countries Safe Drinking Water committees, and on the Board of Scientific Counselors for the Agency for Toxic Substances and Disease Registry (ATSDR). Dr. Calabrese also serves as Chairman of the Biological Effects of Low Level Exposures (BELLE) and as Director of the Northeast Regional Environmental Public Health Center at the University of Massachusetts. Dr. Calabrese was awarded the 2009 Marie Curie Prize for his body of work on hormesis. He was the recipient of the International Society for Cell Communication and Signaling-Springer award for 2010. Dr. Calabrese received an honorary Doctor of Science from McMaster University, Hamilton, Ontario in 2013. Over the past 20 years Professor Calabrese has redirected his research to understanding

the nature of the dose response in the low dose zone and underlying adaptive explanatory mechanisms. Of particular note is that this research has led to important discoveries which indicate that the most fundamental dose response in toxicology and pharmacology is the hormetic-biphasic dose response relationship. These observations are leading to a major transformation in improving drug discovery, development, and in the efficiency of the clinical trial, as well as the scientific foundations for risk assessment and environmental regulation for radiation and chemicals.

CHARACTERIZATION OF RIVERBED SEDIMENTS CONTAINING LINOLEUM WASTE LADEN WITH ASBESTOS

East Coast Conference, October 2015

Matthew W. Muzzy[1§], John Boilard, RIHT[2§]
[1]*P.E., Principal, Sevee & Maher Engineers, Inc.,* [2]*Operations Manager, Northeast Test Consultants*

ABSTRACT

A former borrow pit was used for disposal of linoleum waste. The borrow pit was situated alongside a riverbank, and a portion of the linoleum waste encroached upon the river. Large quantities of linoleum waste became mixed with the sediments in the riverbed. Analysis of samples of the linoleum showed it to be an asbestos-containing material. Studies of the nature and extent of the linoleum waste in the riverbed sediments were conducted and summarized. Descriptions of initial cleanup efforts are presented as well as the planned approach for final mitigation. A discussion of riverbed ecology relative to the asbestos-containing linoleum waste presence is also presented.

1. INTRODUCTION

In the 1930s, a textile mill in Lisbon, Maine was converted to manufacture linoleum flooring. The mill was located along the bank of a small river. On the other side of the river was an abandoned sand and gravel borrow pit (see Figure 1). During periods of normal flow, the base of the borrow pit was at approximately the same elevation as the river. Eventually, the borrow pit became the disposal site for linoleum scrap and remnants from the mill, and at some point, became filled with linoleum waste. The resulting landfill was approximately 4 acres in size with waste thicknesses up to 20 feet. Portions of linoleum waste along the western edge of the landfill encroached upon the river. Unknown volumes of the linoleum waste were carried into the river during periodic high water conditions. Photos 1, 2, and 3 show the linoleum waste that forms much of the western edge of the landfill.

Large pieces of linoleum waste were apparently dislodged from the edge of the landfill, transported downstream, and subsequently accumulated in back eddies and quiescent flow areas (see Photos 4 and 5). Small pieces of linoleum waste were also carried from the landfill and transported downstream, leaving sections of riverbed coated with multi-colored waste fragments. In some riverbed areas, the linoleum waste became part of the sediment deposition process and became intermixed with sand and silt attaining thicknesses of several feet. Photos 6 and 7 show the riverbed and the scattered linoleum waste.

For many years the river level was elevated and held at a near-constant level by a dam located downstream. Most of the accumulated linoleum waste on the riverbed remained submerged by the dammed river. Eventually, the dam fell into disrepair and the river level dropped several

[§]Corresponding authors: Matthew W. Muzzy, SME, 4 Blanchard Road, P.O. Box 84A, Cumberland, ME 04021; Tel: 207-829-5016, mmuzzy@smemaine.com; John Boilard, NTC, 587 Spring St, Westbrook, ME 04092; Tel: 207-331-4796, johnb@netest.com

feet. The lowered water surface resulted in large portions of the waste-covered riverbed becoming exposed during much of the year. For property owners along the river, the exposed linoleum waste represented a degradation of local aesthetics. More importantly, because the linoleum waste was determined to contain asbestos, it was considered to be an Asbestos Containing Material (ACM) and subject to various environmental safety rules.[1]

Sevee & Maher Engineers Inc. (SME), of Cumberland, Maine was tasked to determine the horizontal and vertical extent of the linoleum waste within the riverbed as well as characterize the physical, chemical, and environmental properties of the linoleum waste with respect to: its possible ecological effects on the riverbed sediments; the potential hazard the linoleum waste presented to humans using the riverbed; and what requirements would be necessary for safe handling of the linoleum waste in the event a riverbed clean-up was undertaken.

2. INITIAL CLEAN-UP OF LINOLEUM WASTE

A work plan was prepared to clean up as much of the large-sized linoleum waste (e.g., rolls, sheets, and tile fragments) as possible from the riverbed areas near the residences, as well as from areas where the landfill had encroached on the river (SME 2010a,b). To address the potentially hazardous nature of the ACM, all clean-up activities involving the linoleum waste were supervised by a Maine licensed asbestos management professional and personnel trained in accordance with applicable OSHA and Maine Department of Environmental Protection (MEDEP) Rules.[2] Each worker handling the linoleum waste wore personal protective clothing to minimize contact with the linoleum waste as well as an air purifying respirator to limit risk of particulate inhalation. Air monitoring of the linoleum waste clean-up areas was continuously performed to document the presence of airborne asbestos fibers. The air monitoring showed no asbestos fiber presence in the breathing zone during the initial linoleum waste clean-up activities. Photos 8 and 9 show views of the initial clean-up.

3. INVESTIGATION OF LINOLEUM WASTE IN RIVERBED SEDIMENT

3.1 Nature and Extent of Linoleum Waste in the Riverbed Sediment

The nature and extent of linoleum waste in the riverbed were characterized by collecting sediment samples from more than 50 depth probes (probes) across the river's bottom (SME 2012). Sediment samples from various depths were examined to characterize the composition (i.e., mineral soil content, linoleum waste content, and asbestos content). The probes generally encountered one of three distinctly different soil conditions: 1) granular to mucky sediment, often containing linoleum waste; 2) dense glacial till; and 3) stiff silty clay. The silty clay and glacial till are considered to be the native soils of the riverbed. Figure 2 shows a map of the sediment probe locations and the associated sediment thickness measured by each probe.

[1] Linoleum contains predominantly linseed oil, pine resin, cork, wood, pigments, and limestone filler. For some varieties of linoleum, asbestos was used to help bind the linoleum together.

[2] Asbestos I Construction", 29 CFR 1926.1101, U.S.EPA, "Asbestos Hazard Emergency Response Act"(AHERA), 40 CFR Part 763 and Chapter 425 of the Maine Solid Waste Rules.

Figure 3 shows the surficial extent of the sediment and native soils in the riverbed. Table 1 includes a summary of the measured sediment depths for each of the probe locations.

Thirty-seven sediment samples were selected from the probe locations for measurement of linoleum waste content; first by hand sorting and subsequently by separation using specific gravity. Hand-sorting the sediment was accomplished by passing the sediment and linoleum waste over a No. 4 sieve (0.2-inch opening). The linoleum waste and stones remaining on the sieve were then sorted by hand. The material passing through the sieve (i.e., silt, sand, and small fragments of linoleum waste) was submerged in a water bath adjusted to a specific gravity of 2.0.[3] Published literature indicates linoleum/vinyl flooring generally exhibits a specific gravity in the range of approximately 1.2 to 1.4; therefore, the adjusted specific gravity of the water bath allowed the linoleum waste to float and the mineral soil portion of the sediment to remain submerged. With adequate depth of water in the bath, the soil and linoleum waste became physically separated, allowing the linoleum waste to be recovered and weighed. Table 1 summarizes the linoleum waste content determined for the sediment samples and Figure 4 shows the spatial variation of linoleum waste content across the expanse of the riverbed.

3.2 Asbestos Content of Riverbed Sediment and Linoleum Waste

To quantify the asbestos content of the soil portion of the riverbed sediment, 34 sediment samples were selected for asbestos fiber testing using polarized light microscopy dispersion staining techniques and gravimetric reduction procedures (i.e., U.S.EPA Method 600/R-93/116). Figure 5 shows the asbestos content measured in the riverbed sediment samples, and Table 1 lists the asbestos content measurement by location. In conducting the sediment asbestos content testing, precautions were taken to exclude as much linoleum waste as possible from the sediment. To help compare/contrast the asbestos content of the sediment relative to the linoleum waste, five linoleum waste samples (rinsed free of sediment) were collected from the riverbed (see Figure 5) and also subjected to U.S.EPA 600/R-93/116 testing. All five linoleum waste samples contained asbestos with percentages ranging from 3.8 to 7.5, which was several to many times greater than the asbestos contents measured in the soil portion of the sediment. Table 1 includes the measured asbestos contents for the linoleum waste samples.

3.3 Linoleum Waste and Riverbed Ecology

Asbestos was detected in 13 of the 34 riverbed samples at concentrations ranging from 1.1 to 2.9 percent. To evaluate the potential ecological effect the linoleum waste presented to living organisms in the sediment, a Streamlined Ecological Risk Evaluation (SERE) was performed, which included quantification of the riverbed's benthic community (i.e., macroinvertebrates) using rock-basket sampling (SME 2012). The rock basket sampling confirmed the presence of a benthic community. The favorable benthic community finding, along with research performed for the SERE, relative to microorganisms and asbestos, led to the conclusion that no ecological threat was posed to receptors living in the sediments containing asbestos-laden linoleum waste.

[3] Specific gravity of water bath adjusted by addition of sodium polytungstate.

3.4 Clean-up Considerations and Airborne Asbestos Fiber Evaluation

The SERE showed clean-up of the linoleum waste-laden sediment would not be necessary from an ecological perspective;[4] however, it was clear that the linoleum waste in the riverbed presented an aesthetic degradation of the environment, and that status would not change until clean-up of the linoleum waste in the sediment was performed and areas where exposed linoleum waste formed the riverbank was mitigated. In either case, clean-up of the linoleum waste would involve handling an ACM and a potential for human exposure to airborne asbestos fibers.

To begin preparation for handling the linoleum waste on a mass scale, a study was conducted to evaluate the presence of airborne asbestos fibers in the riverbed and riverbank areas under conditions simulating light construction work (NTC 2012). The study focused on (1) exposed (i.e., low water) riverbed areas where linoleum waste was clearly a constituent of the sediment (see Photo 10) and (2) a portion of the riverbank where the linoleum waste had encroached upon the river (see Photo 1).

The airborne asbestos fiber study was performed by establishing air monitoring stations upwind, downwind, crosswind, and within the selected riverbed and riverbank study areas. Air samples were obtained at vertical distances of 5 feet and 3 feet above the ground surface to simulate a standing adult person's breathing zone and the breathing zone for a sitting/kneeling adult and/or a standing child, respectively. Breathing zone air samples from within the study areas were also collected by equipping field personnel with continuous air sampling devices which were worn throughout the activities completed on the riverbed and riverbank. The study included agitation of exposed sediment and linoleum waste by activities such as walking, running, raking, and shoveling within the study area (see Photos 11 and 12). All agitation of the sediment and linoleum waste occurred on a calm and clear day at mid-day to minimize potential effects of dampness (i.e., dew) and wind. At the conclusion of the study, bulk samples of the sediment and linoleum waste were recovered from the study areas and tested to confirm that the study area did indeed contain asbestos.

[4] Research performed for the SERE indicated that there are no known ecological screening benchmark(s) available for asbestos as related to a riverbed sediment setting. It was recognized that asbestos, in of itself, can however be a source of trace metals (particularly nickel, cobalt, chromium, and manganese). These metals were detected in the groundwater, pore-water, surface water, and sediment analyzed for the overall landfill project and at concentrations sometimes considerably greater than "trace." The benthic community findings for the riverbed clearly showed that the linoleum waste presence in the sediment had no significant effect on the riverbed ecology. Moreover, the metal concentrations measured in the riverbed, as compared to any would-be asbestos related trace metal concentrations, suggested the metals contribution to the sediment from the asbestos was negligible terms of presenting an ecological threat.

The air, linoleum waste, and sediment testing completed for the airborne asbestos fiber study showed the following:[5]

- Ambient air samples collected from simulated light construction work performed on the exposed riverbed and riverbank study areas (where linoleum waste was present) showed no asbestos fiber presence above the air test method detection limits.
- Air samples collected from the breathing zone within the study areas showed no asbestos fiber concentrations greater than or equal to 0.1 fiber per cubic centimeter (f/cc). An asbestos fiber concentration of less than 0.1 f/cc is below the personal exposure limit (PEL) set forth by OSHA.[6]
- Bulk samples of linoleum waste collected from the airborne asbestos fiber study areas showed the linoleum waste to consist of a matrix of non-friable, organically-bound material, including asbestos with approximately two-thirds of the samples analyzed having an asbestos content (by volume) of over 1 percent.
- Sediment samples collected from the riverbed and riverbank study areas showed no presence of unbound asbestos fibers or asbestos bundles.

From the airborne asbestos fiber study, it was concluded that the linoleum waste encountered in the riverbed and riverbank area is mainly comprised of non-friable, organically-bound material, and that casual visitation to the riverbed and areas or riverbank by persons would not be expected to cause a condition of significant elevated health risk from asbestos exposure. The study further led to the conclusion that if the linoleum waste in the riverbed or riverbank was to be disturbed by significant mechanical methods (such as by heavy construction equipment) then a risk for airborne asbestos fiber concentrations in excess of ambient conditions could occur and that such risk could be minimized by implementing engineered control measures.

4. STATUS OF LINOLEUM WASTE CLEAN-UP

Clean-up of the riverbed and riverbank is tentatively scheduled for 2016/2017. A work plan describing the clean-up is in preparation and will soon be submitted to state, local, and federal regulatory agencies for review and approval. The asbestos component of the linoleum waste will necessitate special consideration relative to handling and limiting exposure to workers, as well as occupants of nearby properties. An air monitoring program for the work area and surrounding site area will be established and evaluated on an ongoing basis. Engineered control measures to minimize potential for airborne asbestos fibers in the work area will be an important part of the clean-up and are expected to utilize various wetting methods to suppress workplace dust and airborne particulates. Other elements of consideration for the clean-up will be temporary river

[5] Ambient air sampling consisted of the collection and analysis of samples in accordance with U.S.EPA Method 68-02-3266, Level II Transmission Electron Microscopy (TEM) and NIOSH Method #7402. Air samples were collected for a 2-hour period at 10 liters per minute for a total volume of 1,200 liters. Fibrous asbestos structures in the samples were identified, counted, and sized by morphology, visual Selected Area Electron Diffraction (SAED) pattern recognition, and elemental analysis using Energy-Dispersive X-ray Spectroscopy (EDS) microanalysis. Soil/sediment samples were analyzed in accordance with U.S.EPA Region 1 Methodology for PLM Macroscopic/Microscopic Evaluation with 400 point counting for detection of asbestos fibers and/or fiber bundles present at sizes of >10 mm, 10 mm – 1.0 mm, and <1.0 mm, if necessary.

[6] OSHA's PEL (29 CFR 1910.1001 and 29 CFR 1926.1101) for asbestos in the workplace is 0.1 fiber/cc of air based on an 8-hour, time-weighted average.

diversion, replacement of sediment removed from the riverbed, erosion/scour resistance of the riverbed and riverbank following linoleum waste clean-up, and overall closure of the landfill including those areas encroaching upon the river.

5. ACKNOWLEDGEMENTS

The authors wish to thank Miller Industries, Sevee & Maher Engineers, Inc., and Northeast Test Consultants for their support in preparation and production of this paper.

6. REFERENCES

Northeast Test Consultants (NTC). 2012 September 25. Airborne Asbestos Fiber Evaluation, Bonafide Landfill, Sabattus River Exposed Riverbed and Riverbank Areas, Lisbon, Maine.

Sevee & Maher Engineers, Inc. (SME). 2010a. Report for Partial Cleanup of Flooring Waste Landfill Toe Area, Bonafide Landfill, Lisbon, Maine. Revised 2010 December.

Sevee & Maher Engineers, Inc. (SME). 2010b. Report for Partial Cleanup of Flooring Waste Riverbank Area Downstream of Landfill, Bonafide Landfill, Lisbon, Maine. Revised 2010 December.

Sevee & Maher Engineers, Inc. (SME). 2012 February. Site Investigation Report for Bonafide Landfill, Lisbon, Maine.

Table 1. Sediment characterization details

Location[1,2]	Probe Depth (feet)	Probe Refusal (yes/no)	Sample Depth Interval (feet)	Linoleum Waste Content of Sediment (Percent by Dry Weight)	Asbestos Content of Sediment (Percent by Volume)[3]	Location[1,2]	Probe Depth (feet)	Probe Refusal (yes/no)	Sample Depth Interval (feet)	Linoleum Waste Content of Sediment (Percent by Dry Weight)	Asbestos Content of Sediment (Percent by Volume)[3]
P-1	2	yes	0 - 1			P-27	-		1 - 2	50	Trace
P-2	-		0 - 1			P-27	2.5	yes	2 - 2.5		
P-2	3	yes	1 - 2			P-28	1	yes	0 - 1		
P-3	-		0 - 1			P-29	-		0 - 1	0	None Detected
P-3	6+	no	1 - 2			P-29	2	yes	1 - 2		
P-4	-		0 - 1			P-30	-		0 - 1	0	None Detected
P-4	3	yes	1 - 2			P-30	1.5	yes	1 - 1.5		
P-5	-		0 - 1			P-31	-		0 - 1	0	None Detected
P-5	3	yes	1 - 2			P-31	1.5	yes	1 - 1.5		
P-6	-		0 - 1			P-32	-		0 - 1		
P-6	6	yes	1 - 2	0		P-32	-		1 - 1.5		
P-7	-		0 - 1			P-32	3	yes	2 - 2.5		
P-7	3	yes	1 - 2			P-33	-		0 - 0.5		
P-8	-		0 - 1			P-33	-		0.5 - 1		
P-8	-		1 - 2			P-33	1.5	yes	1 - 1.5		
P-8	3	yes	2 - 3			P-34	-		0 - 1	27	2.2
P-9	-		0 - 1			P-34	2+	no	1 - 1.5		
P-9	6+	no	1 - 2	23		P-35	-		0 - 1		
P-10	-		0 - 1			P-35	1.5		1 - 1.5	95	2.0
P-10	2.5	yes	1 - 2			P-36	-		0 - 1		
P-11	-		0 - 1			P-36	3.5		1 - 1.5	7	1.4
P-11	2.5	yes	1 - 2			P-37	-		0 - 1		
P-12	-		0 - 1			P-37	2	yes	1 - 2	11	1.55
P-12		no	1 - 2			P-38	-		0 - 1		
P-12	yes		2 - 2.5			P-38	3	yes	1.5 - 2	0	None Detected
P-13		yes	0 - 1			P-39	-		0 - 1		None Detected
P-13	yes		1 - 2	16	1.5	P-39	6+	no	1 - 2		
P-14		yes	0 - 1		Trace	P-40	-		0 - 1		
P-14			1 - 2	46		P-40	6+	no	1 - 2		
P-14	no	no	2 - 3			P-101	-		0 - 0.25		
P-15			0 - 1			P-101	1 - 1.5	no	0.25 - 0.75	0	None Detected
P-15	yes		1 - 2			P-102	1	yes	0 - 1	22	2.9
P-16		yes	0 - 1	15		P-103	-		0 - 1		
P-16		yes	1 - 2			P-103	3+	no	1 - 1.5		
P-16	yes		2 - 2.5			P-104	-		0 - 0.5		2.5
P-17		yes	0 - 1			P-104	3+	no	0.5 - 1.0	21	
P-17	yes		1 - 2			P-105	-		0 - 1		
P-18		yes	0 - 1			P-105	-		1 - 1.5	20	1.1
P-18			1 - 2			P-105	2	yes	1.5 - 2		
P-18	yes	yes	2 - 2.5	34	Trace	P-106	-		0 - 1		
P-19			0 - 1			P-106	4	yes	1 - 1.5		Trace
P-19	yes		1 - 2	5	1.8	P-107	-		0 - 1		
P-20		yes	0 - 1			P-107	1.5	no	1 - 1.5	0	None Detected
P-20			1 - 2			P-108	-		0 - 0.5		
P-20			2 - 2.5			P-108	1	no	0.5+		
P-20	yes	yes	2.5 - 3			P-109	-		0 - 0.5		
P-21			0 - 1	26	None Detected	P-109	0.5 - 1	yes	0.5 - 1	0	None Detected
P-21		no	1 - 2			P-110	1	yes	0 - 0.5		
P-21	yes		2 - 3			P-111	1	yes	0 - 1		
P-22			0 - 1			P-112	1.5	yes	0 - 1	5	1.2
P-22	yes		1 - 2	43		S-111	-	-	0 - 1	4	Trace
P-23	-		0 - 1			S-112	-	-	0 - 1	6	1.1
P-23	6+	no	1 - 2			S-113	-	-	0 - 0.5	6	-
P-24	-		0 - 1			S-114	-	-	0 - 0.5	5	Trace
P-24	3	yes	1 - 2			S-115	-	-	0 - 0.5	4	Trace
P-25	-		0 - 1			S-116	-	-	0 - 0.5	15	1.5
P-25	3	yes	1 - 2			S-117	-	-	0 - 1	39	Trace
P-26	-		0 - 1	0	None Detected	S-118	-	-	0 - 1	14	1.6
P-26	6+	no	1 - 2			S-119	-	-	0 - 0.5	18	Trace
						S-120	-	-	0 - 1	9	-
						S-121	-	-	0 - 1	7	Trace
						S-122	-	-	0 - 1	13	-

Notes:
1. P denotes probe sample location.
2. S denotes surficial sediment sample location.
3. Visible fragments of linoleum waste were removed from sediment prior to testing for asbestos content.

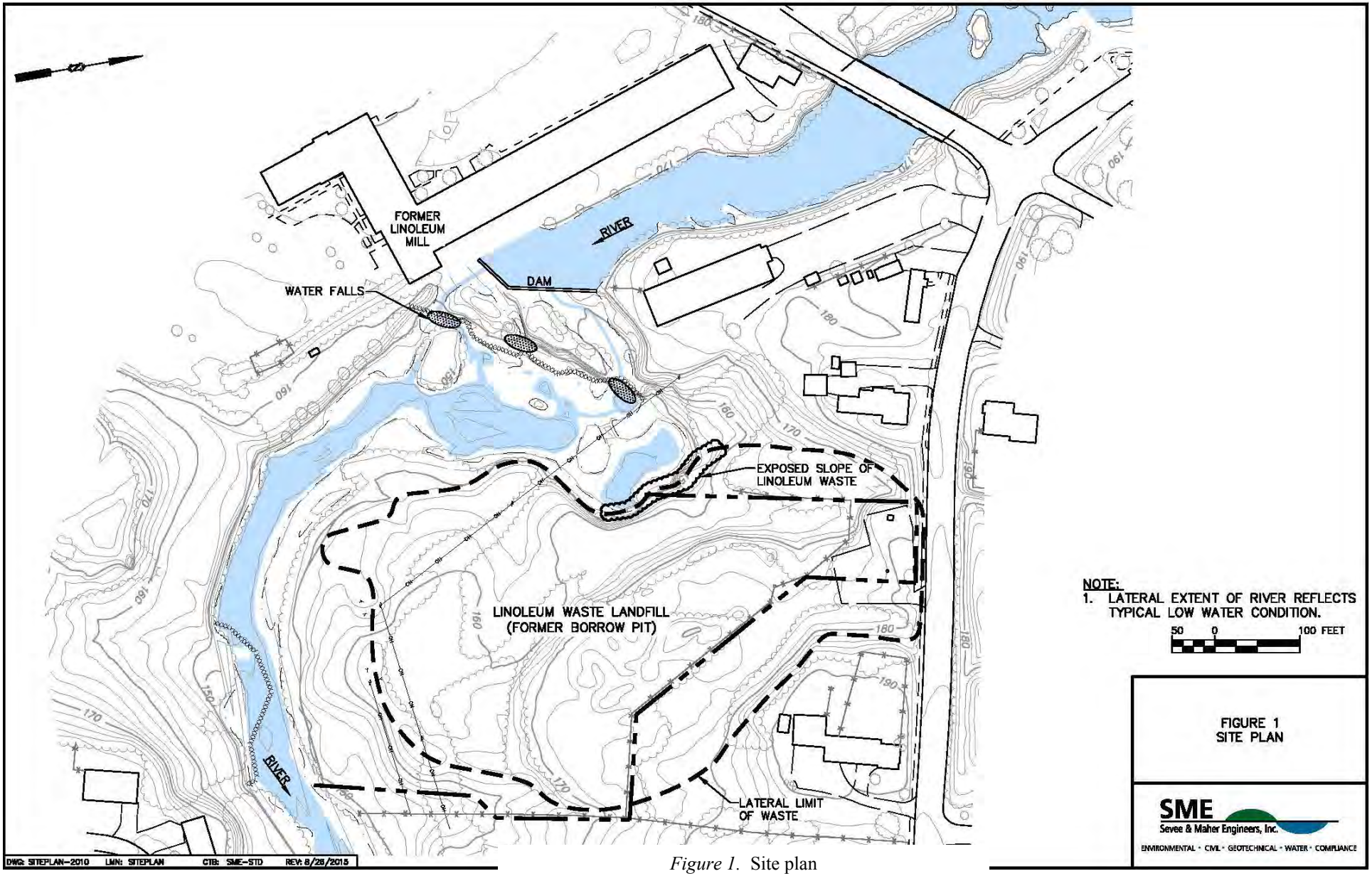


Figure 1. Site plan

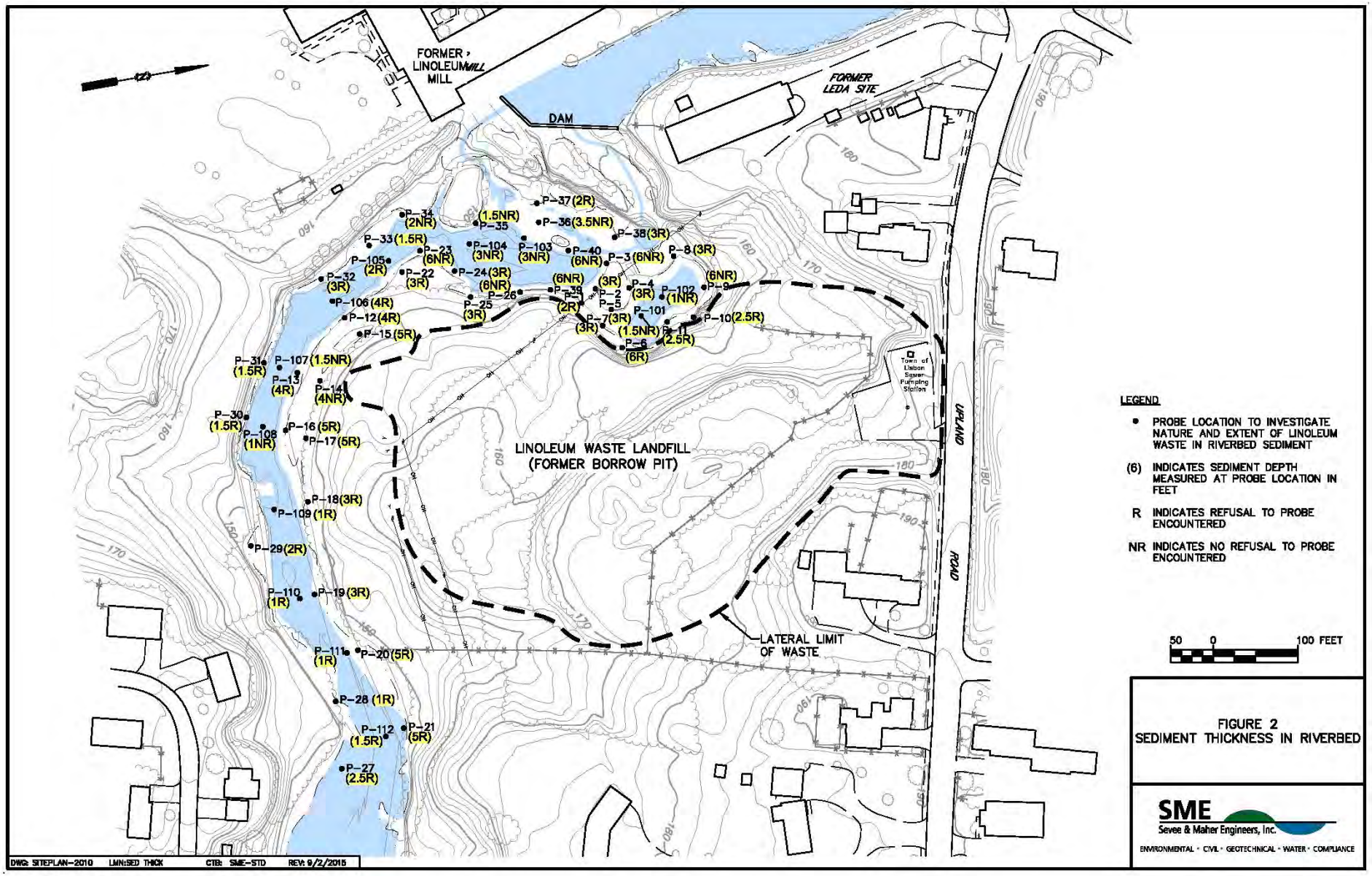


Figure 2. Sediment thickness in riverbed

Figure 3. Surficial extent of sediment in riverbed

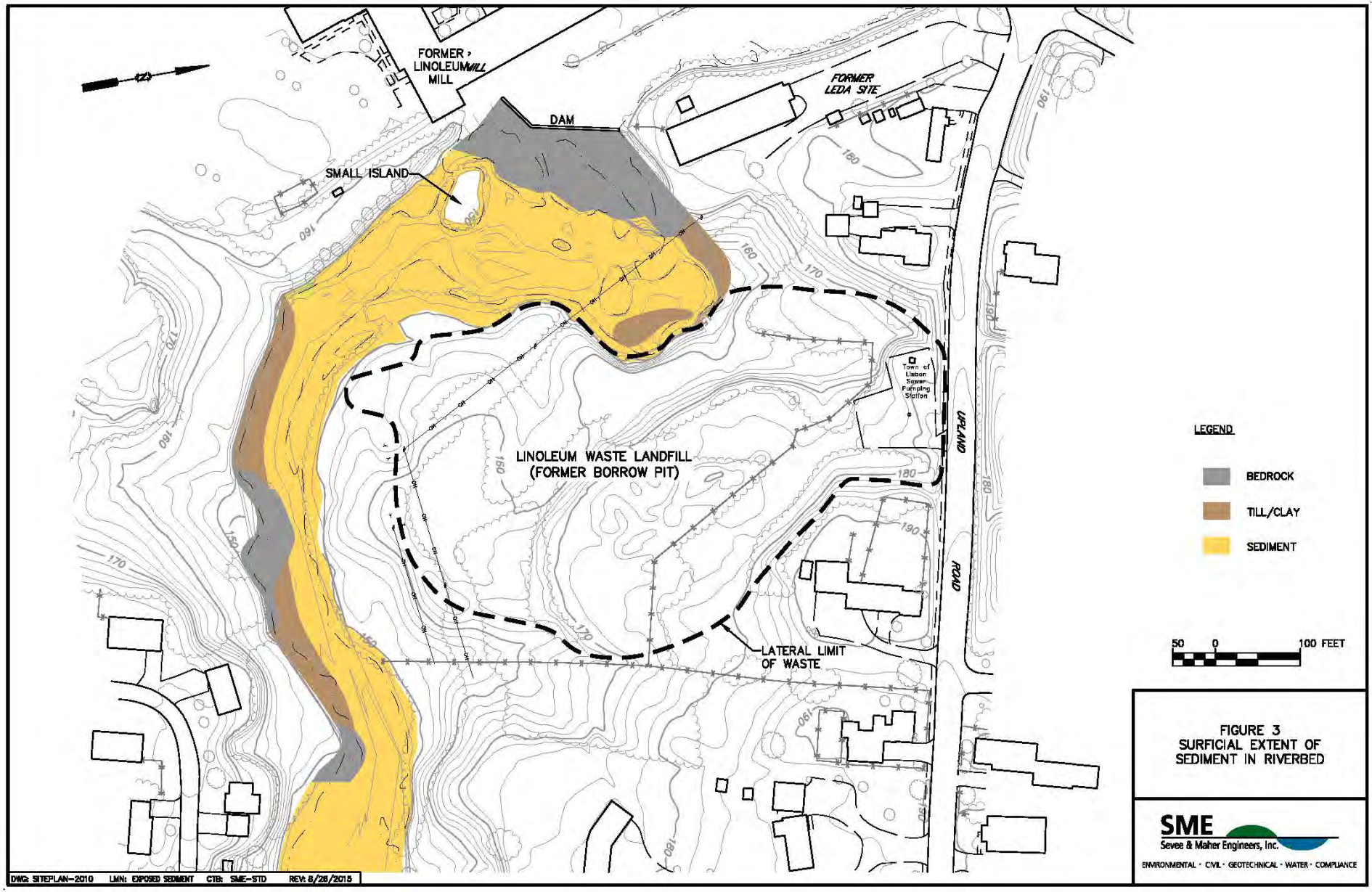
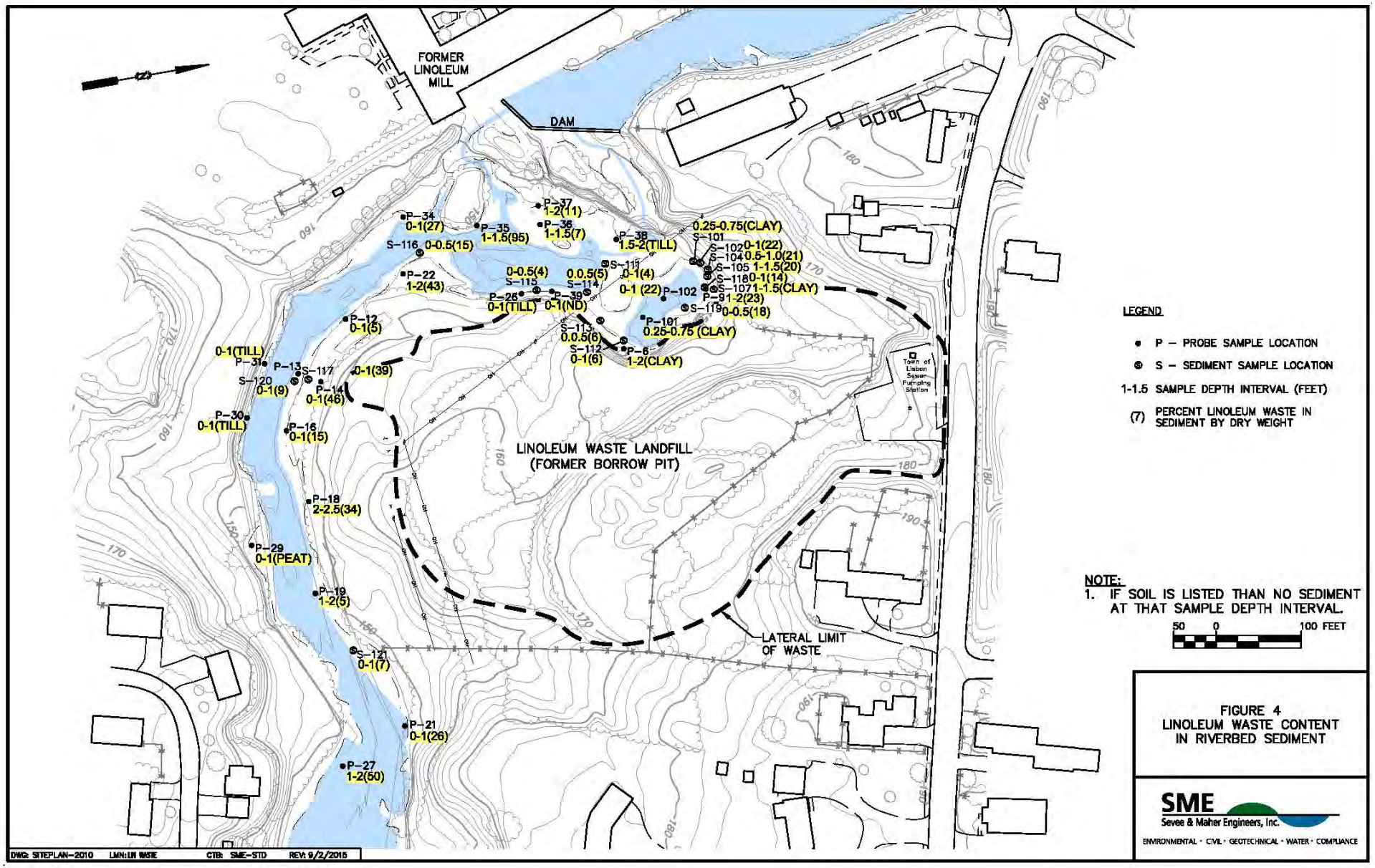


Figure 4. Linoleum waste content in riverbed sediment



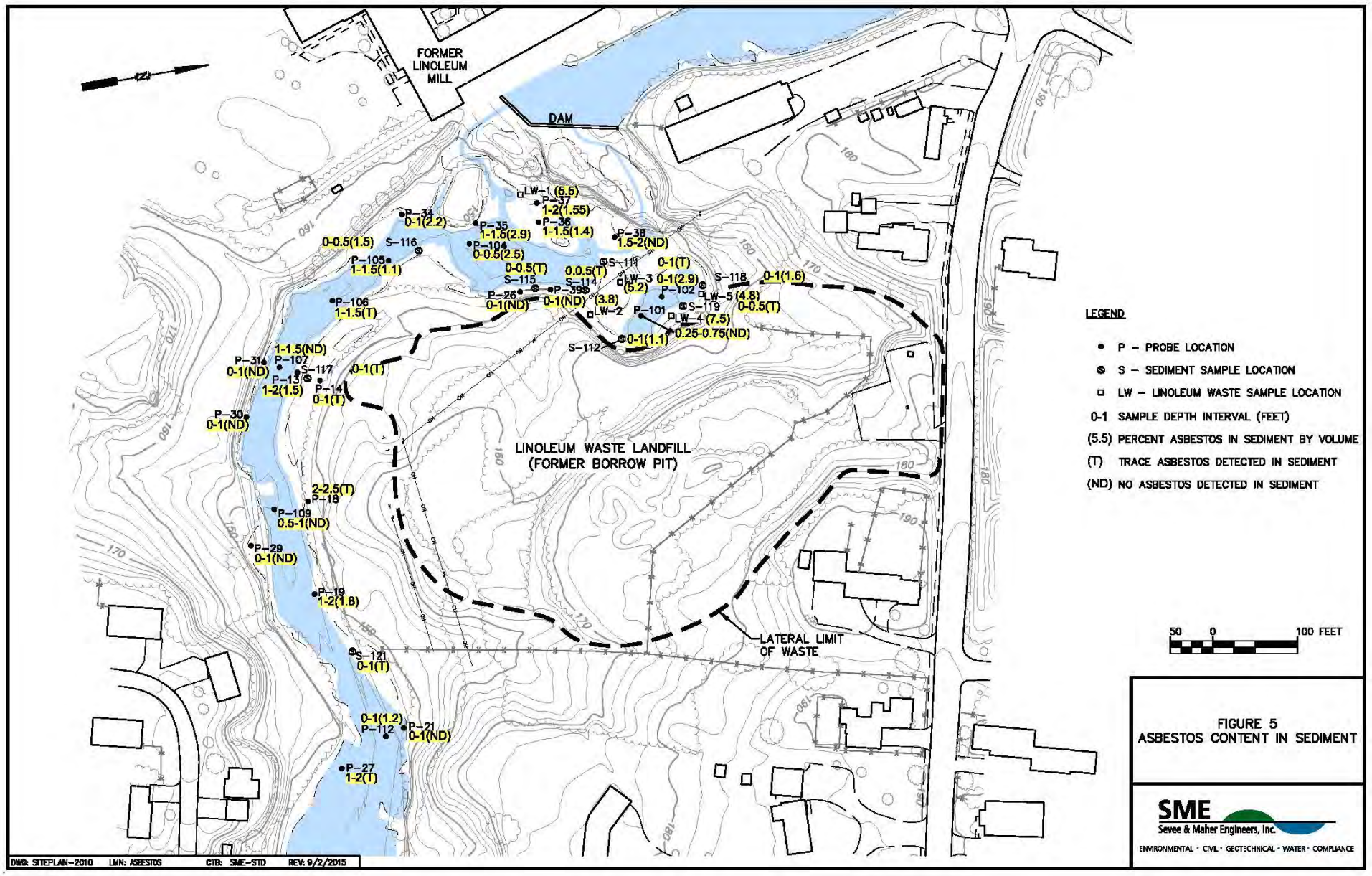


Figure 5. Asbestos content in sediment

Photo 1. Riverbank consisting of exposed linoleum waste.

Photo 2. Rolls of linoleum waste protruding from landfill slope at riverbank.

Photo 3. Rolls of exposed linoleum waste on riverbank area.

Photo 4. Pieces and rolls of linoleum waste on riverbank area.

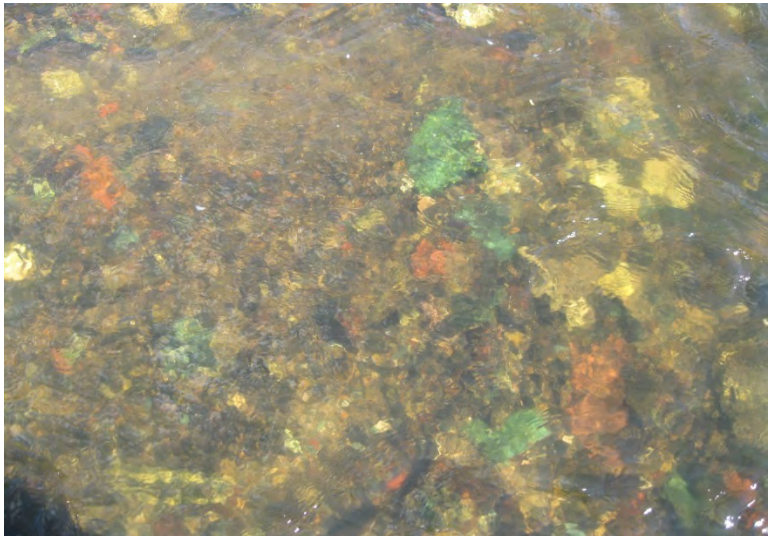


Photo 5. Linoleum waste viewed through shallow river flow.

Photo 6. Linoleum waste along exposed riverbank/riverbed.

Photo 7. Linoleum waste on riverbank/riverbed opposite of landfill.

Photo 8. Limited clean-up of linoleum waste in shallow water near private residences.

Photo 9. Limited clean-up of linoleum waste in exposed riverbed area near private residences.

Photo 10. Linoleum waste covered riverbed study area.

Photo 11. Agitating linoleum waste by raking sediment.

Photo 12. Agitating linoleum waste by shoveling/casting sediment.

MODEL DEVELOPMENT FOR PREDICTING TEMPORAL CHARACTERISTICS OF LEACHATE

West Coast Conference, March 2015

M.S. Al-Suwaiyan[1§]
[1]*Civil and Environmental Engineering Department, King Fahd University of Petroleum and Minerals, Dhahran, Saudi Arabia; msaleh@kfupm.edu.sa*

ABSTRACT

Accidental spills and leaks of petroleum products eventually result in a zone in the subsurface with residual amounts of the light nonaqueous phase liquid (LNAPL) held by capillary forces in the unsaturated zone. This zone contains three phases, namely: air, water and LNAPL, and in most cases it acts as a source of continuous contamination of the surrounding groundwater. Removal of the residual LNAPL is a very important aspect of any subsurface remediation project. In pump-and-treat technology or soil extraction treatment of a contaminated zone, the residual LNAPL is essentially leached slowly into the flowing water or air eventually leading to the reduction and hopefully depletion of the residual LNAPL mass. Since common organics including gasoline and crude oil are actually mixtures of many individual organic products that have different physical and chemical properties that influence their partitioning into the various phases, their concentration in the leachate will vary with time as a result of the change in the LNAPL composition. An outline for the development of a compositional model is presented with the objective of coming up with the concentration of the various components in a leachate through uniform soil contaminated by crude oil. The model will be able to clearly demonstrate the difference in behavior of the various compounds of the LNAPL. The model will demonstrate that the concentrations of some compounds will decrease continuously with time, while other compounds would exhibit a totally different behavior. The model can be used to examine the influence of critical chemical properties of the LNAPL compounds on their leaching aqueous concentration with time. Such a model can also be useful for assessing the progress of the remediation process and its degree of effectiveness, as well as an environmental forensic tool to perhaps determine the source and nature of spills that happened several years back.

1. INTRODUCTION

Groundwater represents the most significant potential source of fresh water on earth (Van der Leeden et al. 1990). Subsurface contamination from accidental spills and leaks of petroleum products from underground or above ground storage, or transport of such products, is a very common environmental problem (USEPA 2011). When such incidents take place, the free product, which is essentially a mixture of many organic compounds that have different physical and chemical properties, is treated as one entity. The spilled hydrocarbon starts to percolate slowly through the unsaturated zone due mainly to gravitational effect that must overcome viscous or interfacial forces. Depending on the spilled volume and the subsurface hydraulic

[§]Corresponding author: M.S. Al-Suwaiyan, Civil and Environmental Engineering Department, King Fahd University of Petroleum and Minerals, Dhahran, Saudi Arabia; msaleh@kfupm.edu.sa

properties, it may continue its downward migration until it reaches the water table and starts to form some sort of a pool of the free product above the capillary fringe. During its downward migration residual amounts remain trapped in the vadose zone due to capillary forces in the unsaturated zone. Factors related to soil properties, as well as fluid properties that affect the distribution in the subsurface were presented in detail from various aspects in previous studies (Al-Suwaiyan et al. 2002; Saleem et al. 2004; Al-Suwaiyan et al. 2006).

2. MODEL DEVELOPMENT

To examine the characteristics of the leachate, a bulk mass balance is carried out over the contaminated zone that accounts for the various contaminant transport mechanisms that affect the process. This approach does not consider the expected changes in the hydrocarbon properties with time due to the continuous change in its composition. Instead, the polluting hydrocarbon is treated as a mixture of various compounds that can have different characteristics and carrying the mass balance on each of these compounds using adequate size of the time step. At the end of each time step, the mass of each compound is calculated, therefore, the new composition is determined along with the change in void space occupied by the polluting hydrocarbon.

For any compound *i*, it exists in four phases, namely: in the free product, in the air voids, in the water voids, or adsorbed on the solid grains. Therefore, its mass per unit bulk volume for constituent *i* will be:

$$m_i = \varphi_{ai} C_{ai} + \varphi_{w_i} C_{w_i} + \rho_i C_{si} + K_{io} C_{oi} \qquad (1)$$

In this equation, the mass of a particular compound is calculated by adding its mass in air, water, adsorbed phase, and in the free product. Each of these is calculated by multiplying the volume by the concentration.

If water flows through the contaminated zone it will leach out the residual phase gradually and the concentrations of the various compounds can be predicted by applying and then solving the mass balance equations for each constituent *i*.

Considering the shown zone with residual amounts of an organic mixture like gasoline, the mass balance for compound *i* is:

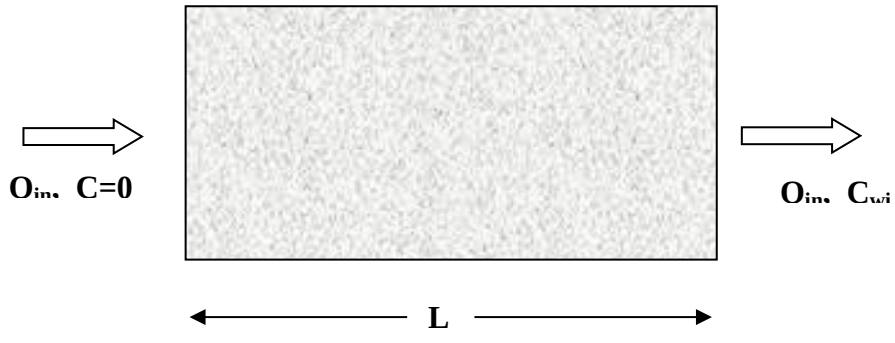


Figure 1. Conceptual model for leaching contaminants for residual hydrocarbon

After a hydrocarbon spill in the vadose zone its components will partition between four possible phases, namely: water, air, soil, and free hydrocarbon. For each component the concentration in water is related to the bulk concentration through (Charbeneau 2000):

$$m_i = B_{w_i} C_{w_i} \qquad (1)$$

Where:

m_i = bulk concentration of compound *i*

B_{w_i} = bulk water partitioning coefficient for compound *i*

C_{w_i} = concentration of compound *i* in water

The bulk water partitioning coefficient will be influenced by component volatility, adsorption properties, and the distribution of compound in the free hydrocarbon.

Applying principle of mass balance will allow us to develop the concentration of pollutants in leachate as a function of time as outlined by Charbeneau [6].

Referring to Fig.1, the total mass present for compound *i* is given by:

$$M_i = A\, L_o\, B_{w_i}\, C_{w_i} \qquad (2)$$

Where:

M_i = total mass of compound *i*

A = area of contaminated zone

L_0 = depth of contaminated zone

Neglecting volatility and degradation and assuming mass is lost only with leaching water, the mass balance equation becomes:

$$\frac{d\,M_i}{d\,t} = \frac{d}{d\,t}\left[A\, L_o\, B_{w_i}\, C_{w_i}\right] = -\,q\,A\, C_{w_i} \qquad (3)$$

The above equations can be solved numerically to come up with concentrations of the various hydrocarbon compounds at various times. Al-Suwaiyan (2011) presents a model restricted to an LNAPL composed of only three compounds.

3. MODEL RESULTS

The temporal variations of sample compounds in leaching water are shown qualitatively in Figure 2. It clearly shows qualitatively that different compounds have different degrees of leachability which varies with time. For example, Benzene is leached much quicker than Toluene and other compounds while Xylene would remain in the contaminated zone for a much longer

time. Such behavior is explained by the relative high solubility of Benzene compared to other BTX compounds apparently controlling at early times, however, as time passes the effective solubility is lowered continuously as the molar fraction of the compound is lowered as its mass in the oil phase in the contaminated zone is lost through leaching.

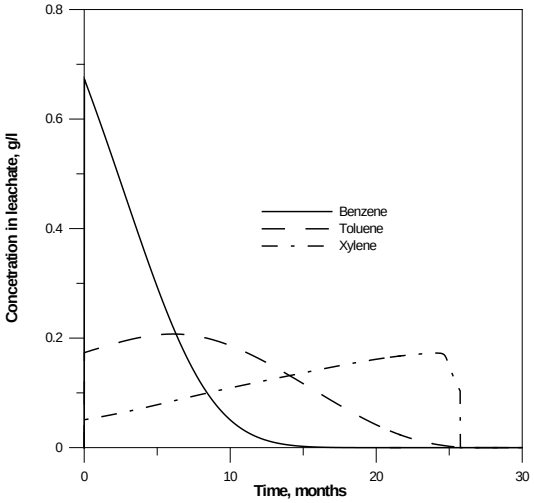


Figure 2. Temporal variations of sample compounds in leaching water

Tables 1 through 3 give the concentration of BTX in leachate, mass of BTX leached, and percentage oil remaining in the contaminated zone, respectively, at different time levels.

Table 1. Concentration in leachate (mg/l)

	Initial	5 months	10 months	15 months	20 months
Benzene	0.68	0.3	0.05	0	0
Toluene	0.18	0.21	0.19	0.12	0.04
Xylene	0.05	0.08	0.12	0.14	0.18

Table 2. Leached mass

	5 months	10 months	15 months	20 months
Benzene	14.2 (64%)	19 (53%)	19.5 (44%)	19.5 (38%)
Toluene	6 (27%)	12 (33%)	16.5 (37%)	18.5 (36%)
Xylene	2 (9%)	5 (14%)	8.5 (19%)	13 (25%)

Table 3. Remaining oil

	5 months	10 months	15 months	20 months	5 months
Oil content	0.0675	0.042	0.027	0.017	0.009
%	100%	62%	40%	25%	13%

The model can produce future predictions for the concentration of any specific compound as a function of time as well as predicting the mass of that specific compound remaining in the contaminated zone. Comparison between the behaviors of the various compounds can be used to make conclusions with respect to the extent of conclusion, as well as the efficiency of the cleanup process.

4. CONCLUSIONS

An outline for the development of a compositional model is presented with the objective of coming up with the concentration of the various components in a leachate through uniform soil contaminated by crude oil. The model is able to clearly demonstrate the difference in behavior of the various compounds of the LNAPL. The model also demonstrates and explains why the concentrations of some compounds will decrease continuously with time while other compounds would exhibit a totally different behavior. The model can be used to examine the influence of critical chemical properties of the LNAPL compounds on their leaching aqueous concentration with time. Such a model can also be useful for assessing the progress of the remediation process and its degree of effectiveness, as well as an environmental forensic tool to perhaps determine the source and nature of spills that happened several years back.

5. ACKNOWLEDGEMENT

The support provided by the Civil and Environmental Engineering Department at King Fahd University of Petroleum and Minerals is highly appreciated.

6. REFERENCES

Al-Suwaiyan MS. 2011. Evaluating and managing potential groundwater pollution from leaking USTs at a coastal city in Saudi Arabia. The International Journal for Sustainable Development and Planning, 6(3):360-376.

Al-Suwaiyan MS, Bashir K, Aiban SA, Ishaq AM. 2002. Analytical model to quantify crude oil spills in sandy layered aquifers. Journal of Environmental Engineering, 128(4):320-326.

Al-Suwaiyan MS, Ishaq AM, Essa MH, Saleem M. 2006. Assessment of the status of USTs at service stations in eastern Saudi Arabia. Proc. of the Fourth Saudi Technical Conference, Riyadh, pp. 286-293.

Charbeneau RJ. 2000. Groundwater Hydraulics and Pollutant Transport. Upper Saddle River (NJ): Prentice–Hall.

Saleem M, Al-Suwaiyan MS, Aiban SA, Ishaq AM, Al-Malack M, Hussain M. 2004. Estimation of spilled hydrocarbon volume – The state-of-the-art. Environmental Technology, 25(9):1077-1090.

United States Environmental Protection Agency (USEPA). Washington (DC): The Office of Ground Water and Drinking Water [Internet] [cited 2011 January 10] Available from: http://www.epa.gov/OGWDW.

Van der Leeden F,Troise FL, Todd DK (Eds.). 1990. The Water Encyclopedia, 2nd ed. Chelsea (MI): Lewis Publishers

ASSESSMENT OF WATER QUALITY BY DETERMINING THE DIVERSITY AND ABUNDANCE OF BENTHIC MACRO-INVERTEBRATES IN THE NIMA CREEK IN GHANA

West Coast Conference, March 2015

L. A. Nuamah[1§], J. Huang[2], and H. R. Dankwa[3]
[1]College of Environmental Science and Engineering, Hohai University, Nanjing, 210098, China, nuamahlinda@yahoo.com; [2]College of Environmental Science and Engineering, Hohai University, Nanjing, 210098, China, huangjingyu120120@126.com; [3] CSIR-Water Research Institute, Accra, Ghana, hr_dankwa@yahoo.com

ABSTRACT

Biological monitoring and assessing the quality of fresh waters has been one of the environmental concerns for many countries of which Ghana is not exempted. This study was undertaken in the Nima creek in Ghana to assess the abundance, composition, and diversity of the benthic macro-invertebrate fauna. It also aimed at determining the quality of water in the creek based on the type of macroinvertebrates found in the creek. Samples were collected at 8 different riffles with a surber sampler. The benthic macroinvertebrates sampled consisted of 6 taxa and 5891 individuals belonging to Nematoda, Oligochaeta, Gastropoda, and Insecta classes. Chironomini were the predominant group with 99.04% on the average, followed by Psychoda sp (0.44%), Rhabitidae (0.26%), and Tubifex (0.26%) at the upstream stretch of the creek. The downstream was dominated by Chironomini forming 97.30% on the average, followed by Tubifex (1.52%), Rhabitidae (1.08%), and Psychoda sp (0.05%). The estimated diversity of the sampling area for both upstream and downstream was assessed by using the Simpson Diversity Index and was found to be 0.53, indicative of a fairly diversified community structure. The Family Biotic Index (FBI) was used to determine the water quality of the creek and found to be 9.92, which indicates severely polluted water. The distribution and occurrences of taxa in the upstream and downstream showed that the macro-invertebrates appeared in both reaches with the exception of *Melanoides tuberculata,* which occurred only at the downstream reach due to low current, the formation of rocky substratum, and the absence of riparian vegetation making it a suitable habitat.

Keywords: Benthic Macroinvertebrate, Riffles, Simpson Diversity Index, Family Biotic Index (FBI)

1. INTRODUCTION

Streams, lakes, and rivers provide a home for the most diverse communities of plants and animals comprising invertebrate forms such as larva insects, annelids(worms), and Mollusca and crustaceans (Johnson et al. 1999; USEPA 2004). Fresh water resources in many developing countries, including Ghana, have not been adequately utilized despite its abundance. Human activities result in widespread and high level stresses of pollution in the water bodies,

[§]Corresponding author: L. A. Nuamah, College of Environmental Science and Engineering, Hohai University, Nanjing, 210098, China; Tel: +86 18705154104; Email: nuamahlinda@yahoo.com.

particularly from sewage discharge, agricultural runoffs, etc. (Asante and Amakye 1998). This disturbance has caused organic enrichment that has an extensive influence on the macrobenthic community structure and composition (Pearson and Rosenberg 1978; Sampaio et al. 2010).

Benthic macroinvertebrates are an essential part of the food web, the interconnection of a food chain, because of their abundance and position in the aquatic web. They play a critical role in the natural flow of energy and nutrient (Lea 1994; Gordon 2000). In most streams, the energy available to organisms are stored in plants and made available to animal life in the form of leaves, algae and bacteria. Macroinvertebrate feed on these leaves, algae, and bacteria, which are at the lower end of the food web. In turn, macroinvertebrates serve as a source of energy (food) for larger animals such as fish, which are a source of energy (food) for birds and amphibians (Aggrey-Fynn et al. 2011). In addition, they serve as bioindicators, which is used to assess the quality of water. Some stream macroinvertebrates may tolerate high levels of pollution, while others cannot survive or even thrive in polluted water (Acharyya and Mitsch 2001; Nazarova et al. 2004; Mekong River Commission 2010). Although water resources with high water quality generally have diverse and rich macroinvertebrate fauna, certain pristine environments have low diversity of macroinvertebrate fauna because of the cold temperature and/or relatively low nutrient levels. Stonefly larvae (Plecoptera), for example, require high dissolved oxygen concentrations and tend to be found in cold, flowing water with a gravel or stone bottom(Peckarsky et al. 1990). Thus, in a polluted stream there are usually large numbers of a few species, while in a clean stream there are moderate numbers of many species (Zimmerman 1993). Since pollution sensitive and tolerant forms are present in "clean" waters, it is the absence of the former coupled with the presence of the latter which may indicate damage. The use of a single species or group of species to provide information on the degree of pollution or the overall water quality has a long history in freshwater systems (Pearson and Rosenberg 1978).

Studies on benthic macroinvertebrates in response to pollution have been carried out in quite a number of countries including Ghana. Baa-Poku *et al.* (Asante and Amakye 1998) observed that the Nima Creek was an indicative of a disturbed urban creek with the impact of effluents on the macroinvertebrate communities. Thorne and Williams (1997) also observed that the macroinvertebrate communities in many developing countries in the tropics displayed a similar response to pollution to that observed in temperate areas. Benbow et al. (2014) observed that specific macroinvertebrate taxa may be used as aquatic biological indicators of the pathogen *Mycobacterium ulcerans* of Buruli ulcer (BU), a tropical disease transmitted by the aquatic macroinvertebrate vector the biting Hemiptera. In addition, the control of causative organisms, such as Simulium and Bulinus, has been one of the main macroinvertebrate studies in Ghana (Hynes 1975a). Related studies carried out include: the annual cycles of macro-invertebrates of the Pawmpawm River in Southern Ghana (Hynes 1975b); the macro-invertebrate fauna of the Ankobra basin (Osafo and Paintsil 1994), and macroinvertebrate communities in the Odaw stream running through Accra (Thorne et al. 2000). Amuzu (1995) and Dartey. (1999) also assessed the impact of urbanization and microbial populations of the Nima Creek in Accra.

The Nima creek, which is not used for drinking purposes, is of economic importance to the inhabitants living in its catchment area. The creek, which serves as the main source of water for irrigation for vegetable farmers along its banks, also receives effluent discharges from the waste treatment plants of some public buildings. The study, therefore, aimed to evaluate and assess the water quality of the Nima creek using the occurrence of macroinvertebrates; to examine the

distribution or occurrence of taxon in upstream and downstream macroinvertebrates in the Nima creek; and to determine the diversity and abundance of macroinvertebrates in the Nima creek.

2. MATERIALS AND METHODS

The study was conducted in the Nima creek located within the Greater Accra region of Ghana. The Nima creek stretches from the Kotoka International Airport and discharges into the Odaw River at Kwame Nkrumah Circle (Figure 1). Drains from the nearby environs such as the Opeibea house, Golden Tulip Hotel, and the Council for Scientific and Industrial Research (CSIR) are discharged into the creek. Farmers nearby use the stream for irrigation activities, and run-offs from the farm enter directly into the stream leading to a high level of nutrients in the stream. The Nima creek has a catchment of about 6.7 km^2 . Its topography is gently rolling except at the headwaters where it is slightly hilly. A large proportion of the basin ranges in elevation from 5.4 to 55.6 m above sea level (Asante and Amakye 1998). The creek is a fast flowing shallow water body with its bed consisting of large rocks and stones. It is heavily shaded by riparian vegetation, particularly vegetation such as rushes and sedges, which balance the water flow, light availability, and temperature levels of the stream.

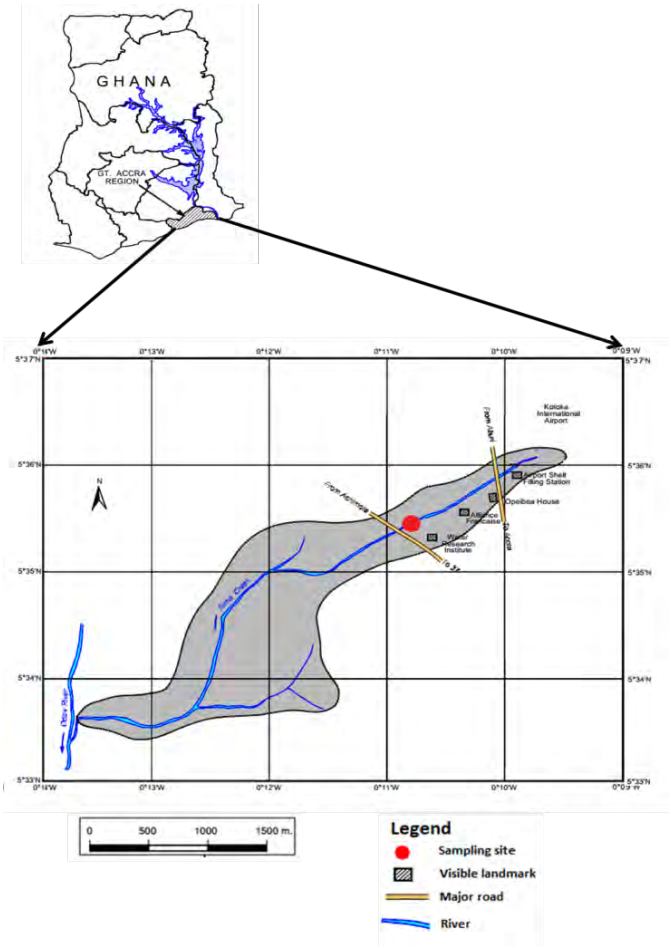


Figure 1. Study area

The depth of the stream was measured using the meter stick and found to be 0.3 m deep and 1.12 m wide. The surber sampler was used to collect the samples. The samples were collected randomly at eight different riffles extending from upstream to downstream. The bridge was used as the midpoint of the stream. The surber sampler was positioned firmly on the floor of the steam facing upstream. Sampling was done by holding the surber firmly against the substrate, and then disturbed by stirring the sediments. The net was then rinsed several times in the water body to allow excess sediments stuck in the net to pass through into the container. Each sample was labeled with a white paper with the following information: the stream name, location, date, and sampler name, and was placed inside the containers. 10% formalin was used to preserve the macroinvertebrates, which then were taken to the lab to be sorted and identified with the aid of the dissecting microscope. Each organism was identified to a major taxonomic group and counted. The percentage composition of macroinvertebrates upstream and downstream was determined using Simpson's Diversity Index (D), which considers not only the number of species (richness) and the total number of individual, but also the proportion of the total that occur in each species (evenness). Thus, this index accounts for both species richness and evenness, and was calculated according to Simpson (1949):

$$D = 1 - \sum \frac{n_i(n_i-1)}{N(N-1)} \quad (1)$$

Where n_i is the number of individuals in the i^{th} species, N is the total number of individuals in the sample, and s is the total number of species in the sample.

The Family Biotic Index (FBI), which is based on categorizing macroinvertebrates depending on their response to organic pollution, was used to assess the quality of the creek. Using the Hilsenhoff (1987) equation, the FBI was calculated as follows:

$$FBI = -\sum \frac{x_i * t_i}{n} \quad (2)$$

Where x_i is the number of individuals in the i^{th} taxon, t_i is the tolerance value of the i^{th} taxon, and n' is the total number of organisms in the sample.

3. RESULTS

The results for the upstream (Table 1) and downstream (Table 2) sampling were compared with the standards outlined by Hilsenhoff (1987).

Table 1. Upstream sampling results

Species	Surber net					Total	% Abundance
	1	2	3	4	5		
Chironomus formosipennis	711	892	701	816	677	3797	98.52
Polypedilum abyssinae	5	2	3	3	7	20	0.52
Psychoda sp	0	0	2	1	14	17	0.44
Rhabitidae	2	2	1	0	5	10	0.26
Tubifex	3	3	1	0	3	10	0.26

Table 2. Downstream sampling results

Species	Surber net			Total	% Abundance
	1	2	3		
Chironomus formosipennis	738	595	639	1972	96.81
Melanoides tuberculata	0	0	1	1	0.05
Polypedilum abyssinae	2	4	4	10	0.49
Psychoda sp	0	1	0	1	0.05
Rhabitidae	2	0	20	22	1.08
Tubifex	3	16	12	31	1.52

Chironomus formosipennis, polypedilum abyssinae, psychoda sp, rhabitidae and *tubifex* were present in the upstream. *Melanoides tuberculate* was not present in the upstream. In the downstream, *chironomus formosipennis, polypedilum abyssinae, psychoda sp, rhabitidae, tubifex* and *melanoides tuberculate* were present.

The percentage abundance of macroinvertebrates in upstream and downstream communities are respectively shown in Figure 2 and Figure 3.

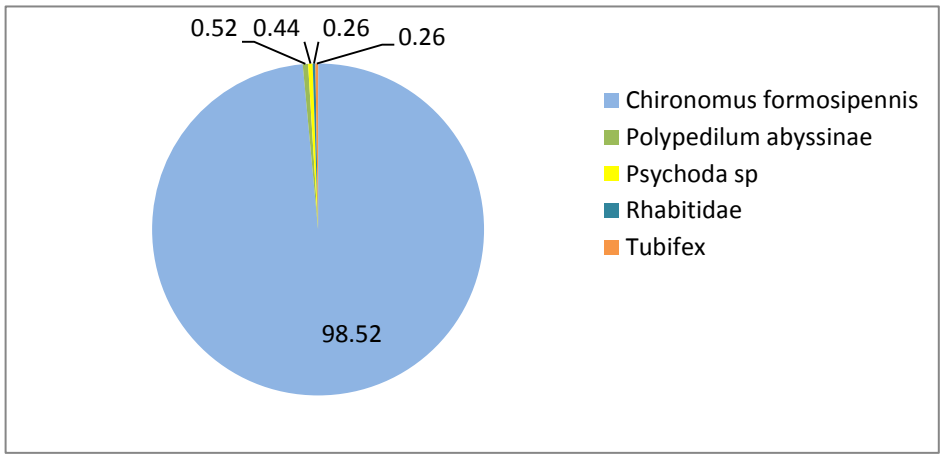


Figure 2. A bar chart showing the percentage abundance of macroinvertebrates in the upstream community

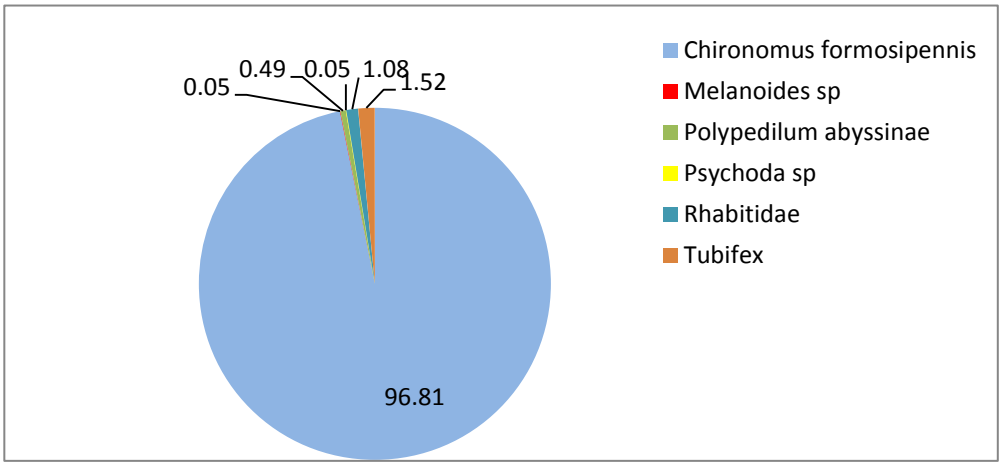


Figure 3. A bar chart showing the percentage abundance of macroinvertebrates in the downstream community

The result for the calculation of FBI is shown in Table 3.

Table 3. Results of Family Biotic Index

Taxa	Suber net								x_i	Tolerance Value (t_i)	$t_i * x_i$
	1	2	3	4	5	6	7	8			
Chironomus formosipennis	711	892	701	816	677	738	595	639	5769	10	57690
Melanoides tuberculata	0	0	0	0	0	0	0	1	1	0	0
Polypedilum abyssinae	5	2	3	3	7	2	4	4	30	6	180
Psychoda sp	0	0	2	1	14	0	1	0	18	10	180
Rhabitidae	2	2	1	0	5	2	0	20	32	0	0
Tubifex	3	3	1	0	3	3	16	12	41	10	410

4. DISCUSSION/CONCLUSIONS

A total of 6 macroinvertebrate taxa were found in the Nima creek. These 6 taxa consisted of 3 *diptera*, which are *mollusca, oligochaeta,* and *nematoda*. The macroinvertebrate benthos indicated by samples examined in the study was significantly similar in the upstream and downstream. A total of 5 taxa were recorded in the upstream and 6 in the downstream. The samples tended to contain a similar proportion of the same macroinvertebrate taxa (Lenz and Miller 1996), indicating no significant difference between the upstream and downstream reaches. However, both reaches contain varying populations. *Melanoides tuberculata* was present in downstream, but absent in upstream reach and is noted to be associated with extremely high levels of pollution. Both upstream and downstream reaches are statistically similar to one another in terms of their diversity, which is the number of different species of macroinvertebrates (richness).The most frequent and dominant species is *Chironomus formosipennis* with a total of 5,769 macroinvertebrates and a percentage abundance of 98.52% and 96.81% representing the upstream reach and downstream reach, respectively. The genus Chironomus, which is known to be tolerant of organic pollution, predominated the fauna. Therefore, the high numbers of Chironomini in the creek confirmed the polluted state of the creek (Asante and Amakye 1998). Simpsons Diversity Index was found to be 0.53 which indicates that the benthic community of the stream is fairly diverse. From Table 3, the Family Biotic Index (FBI) was found to be 9.92, which indicates that the water is of very poor quality and has severe organic pollution. The poor water quality may be attributed to agricultural field runoff that includes nutrients and pesticides. Both may degrade the water quality dramatically, but are present for only a few hours after heavy rainfall. Industrial and urban discharge may also greatly affect the water quality. This has also resulted in the abundance of few species in polluted water and a moderate number of many species in clean water.

Reduced macroinvertebrate fauna at both upstream stations and downstream stations was observed. The impact of the run-off probably caused a disturbance in the life cycle and migration of less tolerant benthic macroinvertebrates, resulting in the non-sensitive species increasing in population density due to the decline of competition with the more sensitive species. It is apparent from the study that the quality of the creek's water deteriorated as one moved downstream, and this was mainly due to the untreated organic waste discharges. This has resulted in the loss of species diversity, a situation that may have adverse effects on the proper functioning of the creek's ecosystem. The impact of effluents on the Nima creek must be

monitored to avoid further extinction of sensitive species, which are already declining in population, as this study has pointed out.

Based on the research findings it is recommended that there must be an effective regular assessment and monitoring of effluents from waste treatment plants before discharging into freshwater bodies by the appropriate regulatory agencies and institutions; enforcement of Environmental Impact Assessment laws by the Environmental Protection Agency of Ghana on all new developmental projects along the catchment area of the creek, including those that are ongoing at the time of this study; and education of members of the communities along the creek on the negative impact of their activities on the creek and its effect on the benthos of our freshwaters. Further studies should be extended to cover other parts of the creek in order to fully document changes in water quality and community structure, and the extent and duration of such changes in order to understand the process of pollution in this creek.

5. ACKNOWLEGEMENT

This work was supported by the Hydrobiology Division of the Council for Scientific and Industrial Research Institute (CSIR). Special thanks go to Dr. J.A Quaye-Ballard of Kwame Nkrumah University of Science and Technology (KNUST) for logistics support; Mr .G. Amegbe of the CSIR for his assistance during the field and Laboratory work; and The College of Environmental Science of Hohai University for office and Lab space.

6. REFERENCES

Acharyya S, Mitsch WJ. 2001. Macroinvertebrate diversity and its ecological implications in two created wetland ecosystems. In: WJ Mitsch; L Zhang. (Ed.), The Olentangy River Wetland Research Park at The Ohio State University Annual Report 2000. The Ohio State University, Columbus, p. 65-76.

Aggrey-Fynn J, Galyuon I, Aheto DW, Okyere I. 2011. Assessment of the environmental conditions and benthic macroinvertebrate communities in two coastal lagoons in Ghana. Ann Biol Res 2.

Amuzu AT. Nana-Amankwaah E, and Bosque- Hamilton EK. 1995. Impact of Development and Urbanization on Urban River Water Quality – The Nima Creek Example. Water Research Institute (Technical Report). p. 33.

Asante F, Amakye JS. 1998. Impact of Urban Effluents on the Macroinvertebrates of 97 a Creek in Accra (Ghana). p. 97–109.

Benbow ME, Kimbirauskas R, McIntosh MD, Williamson H, Quaye C, Boakye D, Small PLC, Merritt RW. 2014. Aquatic macroinvertebrate assemblages of Ghana, West Africa: Understanding the ecology of a neglected tropical disease. Ecohealth, 11:168–183.

Dartey GA. 1999. Assessment of microbial population of a stream and a hotel effluent(Higher diploma in microbial techniques). Institute of Science Technology. Water Research Institute. p. 43

Gordon C. 2000. Hypersaline lagoons as conservation habitats: Macro-invertebrates at Muni Lagoon, Ghana. Biodiversity and Conservation, 9(4):465– 478.

Hilsenhoff WL. 1987. An improved biotic index of organic stream pollution. Gt. Lakes Entomol, 20:31–40.

Hynes JD. 1975a. Downstream drift of invertebrates in a river in southern Ghana. Freshw Biol, 5:515–532.

Hynes JD. 1975b. Annual cycles of macro-invertebrates of a river in southern Ghana. Freshw Biol, 5:71–83.

Johnson BR, Stribling JB, Paul JEFMJ. 1999. Benthic Macroinvertebrates. Bedömningsgrunder för miljökvalitet. Sjöar och vattendrag. Bakgrundsrapport, 2:86-166

Lae R. 1994 Changes in fish and crustacean communities of a tropical lagoon, Lake Togo, submitted to alternate phases of opening and closing belt. Auatic Living Resources, 7:165–179.

Lenz BN, Miller MA. 1996. Comparison of aquatic macroinvertebrate samples collected using different field methods. US Geological Survey, National Water Quality Assessment Program, Madison, Wisconsin. Fact Sheet FS-216-96.

Mekong River Commission VP. 2010. Biomonitoring Methods for the Lower Mekong Basin. MRC, Vientiane (Lao PDR).

Nazarova LB, Semenov VF, Sabirov RM, Efimov IY. 2004. The state of benthic communities and water quality evaluation in the Cheboksary Reservoir. Water Resour, 31:316–322.

Osafo RA, Paintsil DA. 1994. Pollution and water quality studies: Effect of mining activities on River Ankobra and its tributaries. Hase II, WRRI, Accra.

Pearson TH, Rosenberg R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Ocean Mar Biol Ann Rev, 16:229–311.

Peckarsky BL, Horn SC, Statzner B. 1990. Stonefly predation along a hydraulic gradient: a field test of the harsh-benign hypothesis. Freshw Biol, 24:181–191.

Sampaio L, Rodrigues AM, Quintino V. 2010. Carbon and nitrogen stable isotopes in coastal benthic populations under multiple organic enrichment sources. Mar Pollut Bull, 60:1790–1802.

Simpson EH. 1949. Measurement of diversity. Nature, 163:688-688.

Thorne R, Williams P. 1997. The response of benthic macroinvertebrates to pollution in developing countries: A multimetric system of bioassessment. Freshw Biol, 37:671–686.

Thorne RSJ, Williams WP, Gordon C. 2000. The macroinvertebrates of a polluted stream in Ghana. J Freshw Ecol, 15:209–217.

United States Environmental Protection Agency (USEPA). 2004. Water Facts. p. 3–4.

Zimmerman MC. 1993. The use of the biotic index as an indication of water quality. Test Stud Lab Teach, 5:85–98.

PROBABILISTIC RISK ASSESSMENT FOR DIOXIN HEALTH-BASED SOIL CLEANUP GOAL

East Coast Conference, October 2015

Scott C. Simpson[1], Gregory W. Council[1], Talaat Ijaz[2], Douglas J. Covert[3], Christopher M. Teaf[4§]
[1]Tetra Tech, 1165 Sanctuary Parkway, Suite 270, Alpharetta, GA, 30009; [2]Tetra Tech, One Salem Square, 295 Route 22 East, Suite 104E, Whitehouse, NJ, 08889; [3]Hazard Substance & Waste Management Research, Inc., 2976 Wellington Cir. W., Tallahassee, FL, 32309; [4]Center for Biomedical & Toxicological Research, Florida State University, 2035 E. Dirac Dr., Tallahassee, FL, 32310

ABSTRACT

State regulators in Florida recently accepted a first-of-its-kind Probabilistic Risk Assessment (PRA) for determining an alternative residential Soil Cleanup Target Level (SCTL) for dioxin (32 ng/kg TEQ). The previous residential SCTL (7 ng/kg TEQ) had been based on a single, deterministic calculation with numerous conservative assumptions, resulting in a residential SCTL which was overly conservative and protective far beyond the regulatory mandate (i.e., 10^{-6} increase in cancer risk due to exposure). Conversely, this PRA used a Monte Carlo simulation approach to estimate risk for all members of a large population of receptors using a combination of scientific data and professional judgment, with final details developed during negotiations with regulators. The simulation parameters were defined probabilistically and reflect the full ranges of reasonable values for the following exposure variables used to estimate human health risk: body weight, exposure duration, exposure frequency, fraction from contaminated source, soil ingestion rate, and relative bioavailability. Other variable and uncertain parameters which could, and perhaps should, have been treated probabilistically were instead treated deterministically per direction from the state regulators. Regulators also required that a pre-supposed high-risk subpopulation be analyzed separate from the full receptor population. Therefore, the new SCTL is still a conservative estimate given the statutory requirement of 10^{-6} risk. Despite the conservativeness of the newly-approved alternative SCTL, this PRA represents a significant step toward more realistic estimates of human health risks caused by environmental contaminant exposure.

Keywords: probabilistic risk assessment, human exposure, dioxin, soil, screening, cleanup guidelines

[§]Corresponding author: Christopher M. Teaf, Center for Biomedical & Toxicological Research, Florida State University, 2035 E. Dirac Dr., Tallahassee, FL, 32310

1. INTRODUCTION

1.1. General Dioxin Information

Polychlorinated dibenzo-*p*-dioxins and polychlorinated dibenzofurans (collectively referred to as "dioxin") are complex, but related chlorinated compounds with similar chemical structures and biological activity. It is commonly understood that the most potent and best-studied dioxin congener is 2,3,7,8-tetrachlorinated dibenzo-*p*-dioxin (TCDD). Sixteen other congeners with chlorine atoms occupying different combinations of positions on the molecule are commonly believed to have a similar mechanism of toxicity, but for most congeners the toxicity is much less than TCDD. It is common practice to express the total measured concentration of dioxins in a sample in terms of a TCDD "toxic equivalent" (TEQ) concentration. This TEQ concentration is determined by multiplying the measured concentrations of each congener by toxic equivalency factors (TEFs)—estimates of potencies of particular dioxin congeners relative to TCDD—and summing the results for all congeners. Risk due to dioxin exposure is typically assessed using TEQ concentrations rather than on the basis of TCDD or other congener-specific concentrations.

Dioxin molecules are highly hydrophobic and tend to bind strongly with soil particulates in the environment. Due to their hydrophobicity and low solubility in water, dioxins pose a greater threat to human health via exposure to contaminated soil than via ingestion of dioxins dissolved in water.

1.2. Site Background

The Probabilistic Risk Assessment (PRA) described herein was performed for application at a former wood-preserving plant near Baldwin, Florida, that treated railroad crossties, utility poles, and other wood products from the 1950s until 1988. Wood-preserving products used by the facility included chromated copper arsenate (CCA), creosote, and pentachlorophenol. Pentachlorophenol is synthesized by the chlorination of phenol, a process which creates dioxin congeners as chemical by-products. Multiple rounds of on- and off-site soil and sediment sampling were conducted to analyze for several chemical constituents potentially related to past wood-treatment activities. Based on these soil sampling results, dioxin was identified as the key constituent, which defines where corrective action may be appropriate.

1.3. Risk Calculation and Florida Default SCTL

Human health risk assessments often require the calculation of two different types of risk: one for cancer effects and another for non-cancer effects. Florida has statutory requirements (Section 376.30701 of Florida Statutes) that the lifetime incremental cancer risk (LICR) cannot exceed one in one million (10^{-6}) and the non-cancer hazard index cannot exceed one "under actual circumstances of exposure."

The LICR for a given TEQ concentration is calculated by the following formula (modified from FDEP, 2005):

$$LICR = \frac{TEQ \cdot EF \cdot ED \cdot FC \cdot [(IR_o \cdot CSF_o \cdot RBA_o) + (SA \cdot AF \cdot DA \cdot CSF_d \cdot RBA_d) + (IR_i / PEF \cdot CSF_i \cdot RBA_i)]}{BW \cdot AT}$$

where,

AF = Dermal adherence factor
AT = Averaging Time
BW = Body Weight
CSF = Cancer Slope Factor; oral, dermal or inhalation
DA = Dermal Absorption
ED = Exposure Duration
EF = Exposure Frequency
FC = Fraction Contacted from contaminated source
IR_i = Inhalation rate
IR_o = Oral ingestion rate
PEF = Particulate Emission Factor
RBA = Relative Bioavailability; oral, dermal or inhalation
SA = Skin Surface Area
TEQ = Toxicity equivalent concentration

The non-cancer hazard index (HI) is calculated similarly:

$$HI = \frac{TEQ \cdot EF \cdot ED \cdot FC \cdot [(IR_o \cdot RBA_o \div RfD_o) + (SA \cdot AF \cdot DA \cdot RBA_d \div RfD_d) + (IR_i / PEF \div RfD_i)]}{BW \cdot AT}$$

where,

RfD_d = Dermal reference dose
RfD_i = Inhalation reference dose
RfD_o = Oral reference dose

The TEQ concentration is adjusted to the greatest concentration that produces LICR $\leq 10^{-6}$ and HI ≤ 1; this concentration is taken to be the SCTL. Based on the cancer risk formula and the Florida Department of Environmental Protection (FDEP) default parameter values in Table 1, the default residential SCTL for dioxin is 7 ng/kg TEQ. The deterministic approach used to define this SCTL combines multiple conservative assumptions (i.e., default values at the upper-end of likely value ranges) in a way that achieves a cancer risk level that is more restrictive than Florida's statutory 10^{-6} standard.

Table 1. FDEP default values for SCTL variables

Variable	Description	Deterministic Default (FDEP, 2005)
TR	Target cancer risk	10^{-6}
BW	Body weight	51.9 kg
AT	Averaging time	25500 days (70 years)
ED	Exposure duration	30 years
EF	Exposure frequency	350 days/year
FC	Fraction from contaminated source	1
IR_o	Soil ingestion rate (oral)	0.00012 kg/day (120 mg/d)
SA	Surface area of exposed skin	4810 cm^2
AF	Skin adherence factor	0.1 mg/cm^2
DA	Dermal absorption fraction	0.01
IR_i	Inhalation rate	12.2 m^3/day
PEF	Particulate emission factor	1.24 x 10^9 m^3/kg
CSF_o	Cancer slope factor (oral)	150,000 $(mg/kg\text{-}day)^{-1}$
CSF_d	Cancer slope factor (dermal)	166,700 $(mg/kg\text{-}day)^{-1}$
CSF_i	Cancer slope factor (inhalation)	150,000 $(mg/kg\text{-}day)^{-1}$
RBA_o	Relative bioavailability factor (oral)	1
RBA_d	Relative bioavailability factor (dermal)	1
RBA_i	Relative bioavailability factor (inhalation)	1

Florida's default SCTL is similar in magnitude to the EPA Regional Screening Level (RSL) for 10^{-6} cancer risk (4.8 ng/kg), but is considerably lower than the EPA Preliminary Remediation Goal (PRG; 50 ng/kg). The FDEP TEQ-based SCTL is among the most conservative cleanup standards for dioxin in the United States (EPA 2009).

When calculating TEQ concentrations for site soil samples and comparing them to the SCTL, FDEP (2005) recommends using TEFs which were first published by the World Health Organization (WHO) in 1998 (Van den Berg et al. 1998). These TEFs play a critical role in determining TEQ concentrations calculated for soil near the Baldwin, FL site because very little TCDD has been measured in off-site soil samples; 1,2,3,7,8-PeCDD and 1,2,3,4,6,7,8-HpCDD are the dominant congeners at the site for calculation of TEQ.

1.4. Receptor Variability and Monte-Carlo Sampling

The standard deterministic risk assessment approach neglects the fact that different individuals (receptors) within a given population will have different levels of exposure to environmental contaminants. A probabilistic approach captures this variability within the receptor population and allows for the most susceptible subpopulation(s) to be identified. The incidental soil ingestion pathway is the dominant component of the risk estimates for dioxins. Therefore, the

exposure variables that define the ingestion pathway were treated probabilistically in this PRA using a repeated (Monte Carlo) sampling and simulation approach. The result of this approach is a population of simulated receptors, each with a single set of exposure parameter values and an associated LICR result. The risk values assigned to each receptor can then be summarized statistically and used to ensure that even the most exposed portion of the population is protected.

2. METHODS

2.1. Overview

Several of the exposure variables were selected to be simulated probabilistically: body weight (BW), exposure duration (ED), exposure frequency (EF), fraction from contaminated source (FC), soil ingestion rate (IR_o), and relative oral bioavailability (RBA_o). All other exposure variables were defined using the (conservative) FDEP default deterministic values.

The GoldSim software package was used to perform 100,000 Monte Carlo simulations (realizations) in this PRA. GoldSim is a simulation software package with advanced probabilistic (Monte Carlo) simulation capabilities which facilitates the simulation of time-variable processes through use of a time-step loop. The methodology employed in this PRA could also be implemented using other simulation tools.

2.2. Simulation of Receptors and Exposure Periods using Population Data

The first step in estimating risk for an individual receptor was to define three basic receptor attributes relevant to exposure and mobility: gender, exposure duration, and age(s) during the exposure period. Johnson and Capel (1992) provide one of the two key population mobility studies cited in the EPA (2011) Exposure Factors Handbook (EFH). The Johnson and Capel (1992) study describes a Monte Carlo simulation of gender, residential occupancy period (ROP) and ages during exposure for individual receptors using demographic data. Their methodology ("ROPSIM") was used with 2010 U.S. Census data (U.S. Census Bureau 2011) to select a gender for each receptor and, based on the gender, assign the receptor an age during the current year ("current age") using gender-specific 2010 Census age distributions.

The simulated receptor age at the start of exposure and at the end of exposure are found by looking backward and forward from the current age in one year increments. Johnson and Capel (1992) provided gender- and age-specific probabilities that a receptor lived in the same residence during the previous year (PSR). The PSR value corresponding to the receptor's gender and current age is used to determine (in accordance with that likelihood) if the receptor lived in their current residence during the previous year. This procedure is repeated for each successive preceding year until it is determined that either the receptor moved to their current residence, or the receptor's age is zero (i.e., the receptor was born at the residence). A similar procedure is employed when stepping forward in time, which differs in that it also uses gender- and age-specific probabilities that a receptor will die during a particular year (PD). If the receptor lives past his or her current age (based on PD), then the same logic used to define past residency is

used for the future until the receptor either moves from the residence, dies, or reaches 100 years of age[1]. The exposure duration is then calculated based on the ages at the start and end of the exposure period.

Each receptor's body weight is also defined using the age during the exposure period and gender determined by ROPSIM. Gender- and age-specific weight distributions are based on data in the EFH (EPA 2011), which are based on EPA's analysis of the 1999 through 2006 National Health and Nutrition Examination Surveys (NHANES). The time-weighted average body weight during the exposure period is used in calculating each receptor's health risks due to exposure to site soil.

2.3. Input Distributions Defined Based on Data Review, Professional Judgment, and Regulatory Negotiation

A variety of evidence suggests that soil-bound dioxins are less bioavailable than the dioxins in surrogate media typically used in oral toxicity studies (e.g., food, water, oil). Evidence also suggests that higher chlorinated (hexa-, hepta-, octa-) congeners are not absorbed following oral exposure as easily as the lower chlorinated (tetra- and penta-) congeners (e.g., Budinsky et al. 2008; Exponent 2005; Finley et al. 2009; National Academy of Science 2006; SRC 2010). RBA_o estimates for individual congeners were presented by Finley et al. (2009). The prevalence of highly chlorinated congeners in soil samples near the site make the relative bioavailability of these congeners particularly important in assessing the risk to receptors from exposure to site soil. Congener-specific RBA_o distributions based on results from Finley et al. (2009) were originally proposed for this PRA. However, following negotiations with regulators, a single triangular distribution of RBA_o was applied to the dioxin congeners at the site (triangular distribution parameters: minimum = 0.3; mode = 0.5; maximum = 0.7).

The three remaining probabilistic variables—IR_o, EF and FC—were defined using different distributions for children and adults. For the purposes of this analysis, receptors are considered to be children from birth through age 6 years, and adults from age 7 years and older. All three of these exposure parameters were allowed to vary for each receptor from one year to the next using interannual auto-correlation, the rationale being that these variables are related to individuals' behavior patterns, and that receptor-specific behavioral patterns are likely to change over the course of multiple years.

Professional judgment suggests that children typically ingest more soil than adults. Stanek et al. (2001) is the most applicable scientific study for estimating a distribution of long-term (i.e. yearly-average) soil ingestion rates for children. Adult soil ingestion is less well characterized, but it is generally acknowledged that adults ingest significantly less soil than children. The distributions proposed initially for this PRA were based on the Stanek et al. (2001) results. These distributions were modified to be more conservative during negotiations with FDEP, and the resulting distributions used in this PRA are shown in Table 2. The yearly IR_o auto-correlation value of 80% for both adults and children accounts for temporal changes in human behaviors influencing soil ingestion such as the amount of time spent recreating in outdoor residential spaces and eating homegrown fruits or vegetables with residual soil on the surface.

[1] Johnson and Capel (1992) provided the gender- and age-specific PSR and PD for ages 0 to 100 years.

These behaviors vary not only between individual receptors, but also for the same individual over time, with a high degree of correlation (80% assumed) from one year to the next.

Table 2. Triangular distribution parameters for IR_o, EF and FC.

Parameter	Units	Receptor Group	Minimum	Mode	Maximum	Annual Auto-Correlation
IR_O	mg/d	Child	0	100	200	80%
		Adult	0	50	100	80%
EF	d/yr	Child	325	350	365	20%
		Adult	325	340	365	20%
FC	-	Child	0.2	0.7	1.0	20%
		Adult	0.1	0.7	1.0	20%

Little data exists to define scientifically-based estimates of EF and FC. However, professional judgment suggests that children typically spend more time at home and ingest relatively more soil away from home than adults. The negotiated distributions used in this PRA for EF and FC are shown in Table 2. The child and adult distributions for EF both represent receptors who spend 0 – 40 days away from their residence each year. The mode (i.e., most likely) values of 350 d/yr (child) and 340 d/yr (adult) correspond to receptors who spend approximately two to four weeks per year away from home. The FC distributions account for the fact that not all of the soil a person ingests comes from their residence. The low ends of the FC distributions represent receptors such as outdoor workers who get most of their soil intake from a job location distant from his or her residence. The high end of the possible range represents receptors who spend most of their outdoor time at or near their homes. The relatively low annual auto-correlations values for both sets of EF and FC estimates (20%) reflect greater year-to-year variability in EF and FC than IR_o.

3. RESULTS

3.1. Exposure Variable Distributions

The full distribution of exposure durations from all 100,000 simulated receptors is shown in Figure 1, along with the FDEP default deterministic value (30 years) and the ROP estimates calculated by Johnson and Capel using the ROPSIM methodology. Small differences between this PRA's ensemble distribution and those from the Johnson and Capel are attributable to use of different population statistics (i.e., 2010 versus 1990).

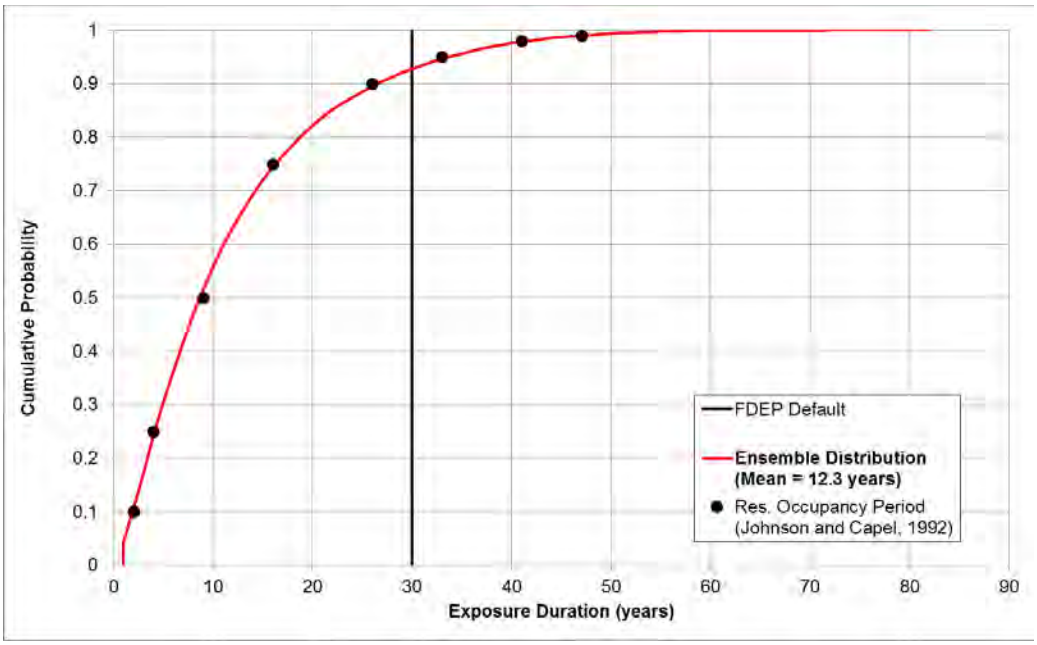


Figure 1. Exposure duration cumulative probability distribution calculated by ROPSIM

Figure 2 shows the time-weighted average body weight of all receptors during exposure. It is noteworthy that (1) the median value of approximately 68.7 kg (151 lb.) is greater than the default FDEP deterministic value of 51.9 kg (114 lb.), and (2) the BW distribution does include a significant fraction of receptors with low average body weights (akin to children) in the risk calculation.

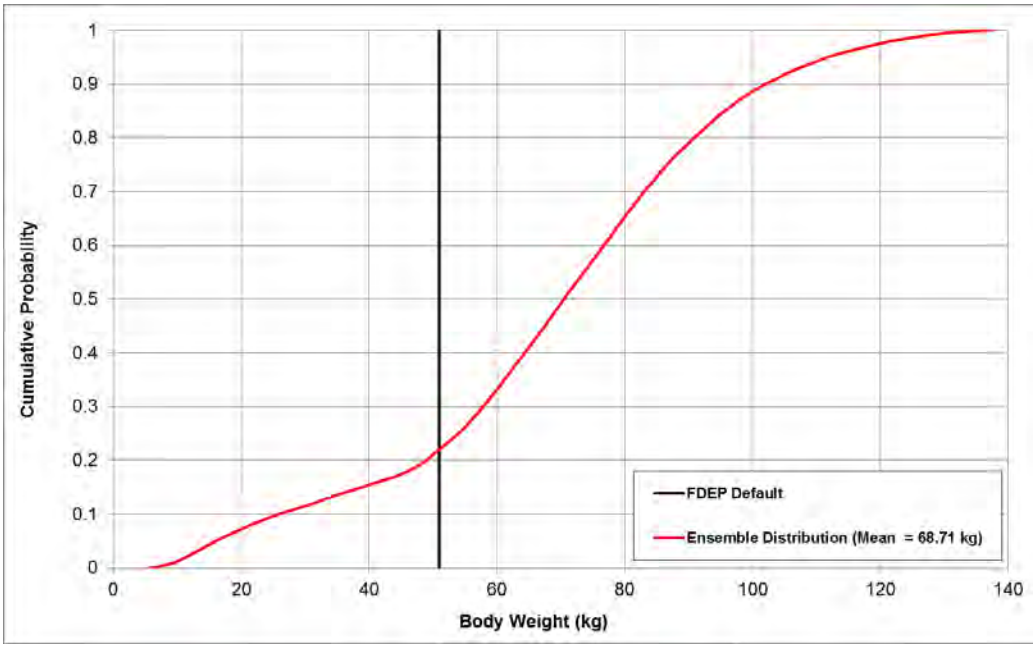


Figure 2. Body weight cumulative probability distribution

Figure 3 shows the distribution of RBA_o for all receptors. The entire distribution notably falls below the FDEP default value (1.0).

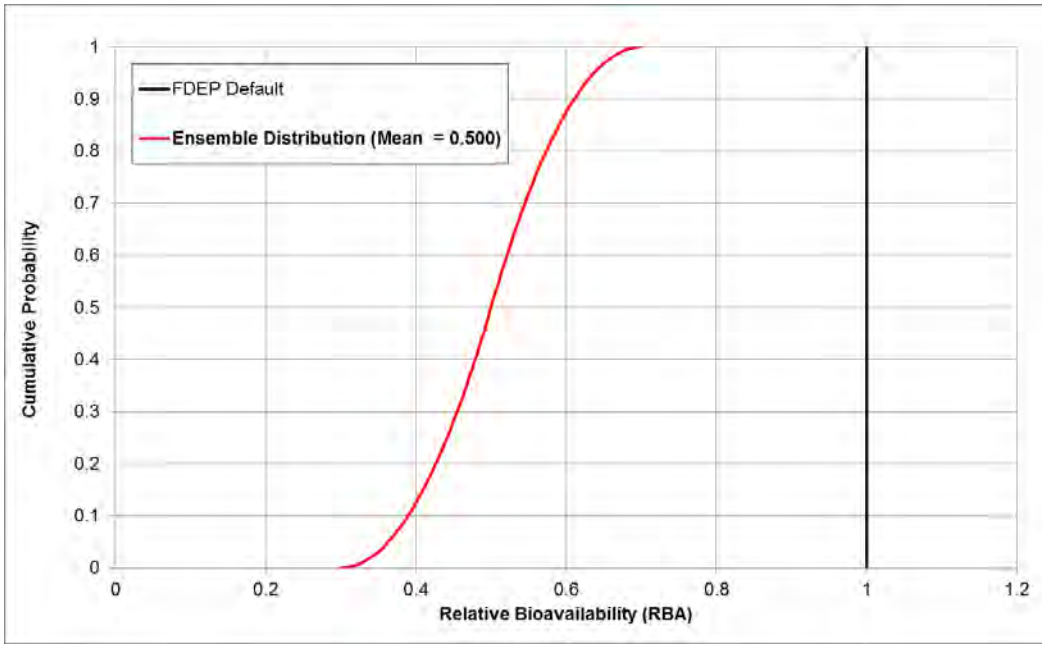


Figure 3. Cumulative probability distribution for relative bioavailability

The specified child, adult, and resulting ensemble distributions for soil ingestion rate are shown in Figure 4. The ensemble distribution generally falls closer to the adult distribution because most exposure periods occur largely or entirely in adulthood.

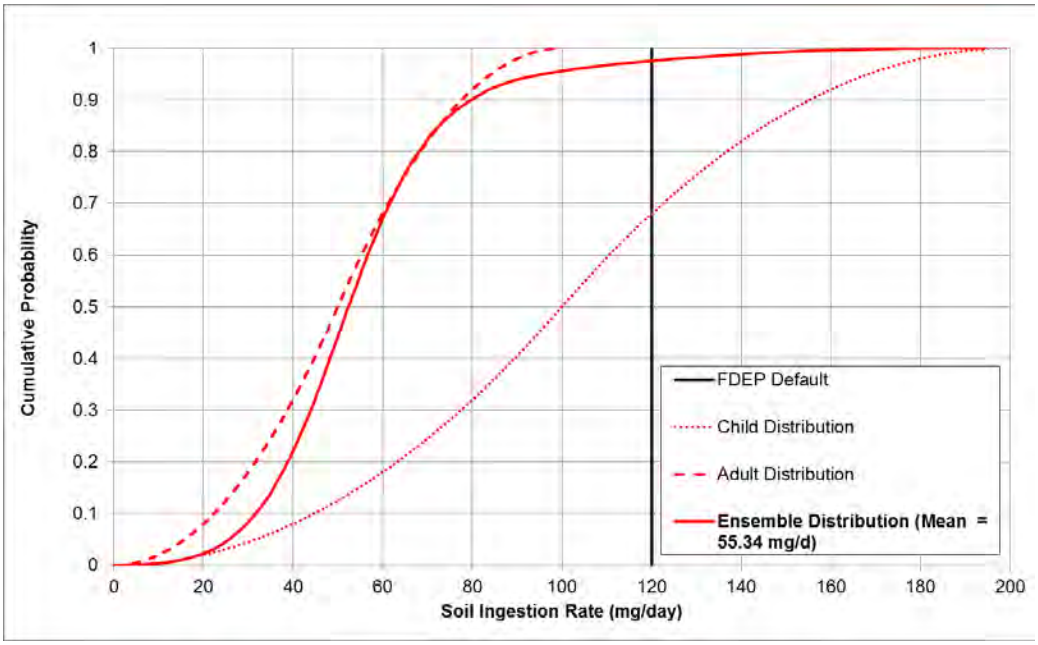


Figure 4. Cumulative probability distributions for soil ingestion rates

Figure 5 illustrates the effects of the annual resampling of the child and adult exposure frequency distributions; the ensemble distribution for all receptors is narrower than either of the parent distributions, but has a central tendency which falls between the child and adult distributions.

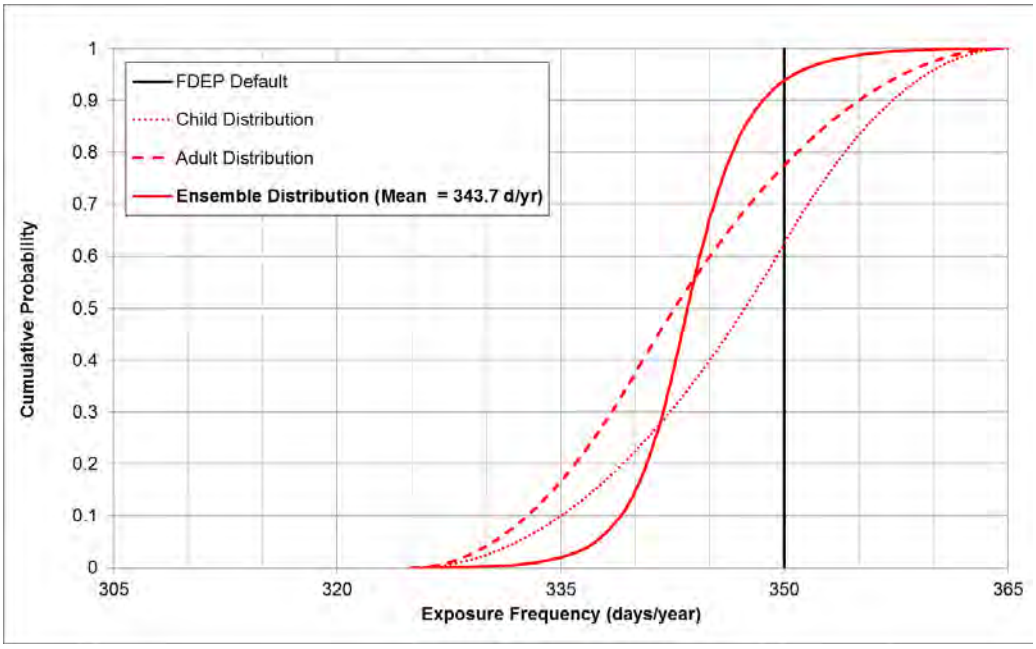


Figure 5. Cumulative probability distributions for exposure frequency

Figure 6 shows a similar pattern for FC, which has an average of approximately 0.6 compared to the FDEP default value of 1.0.

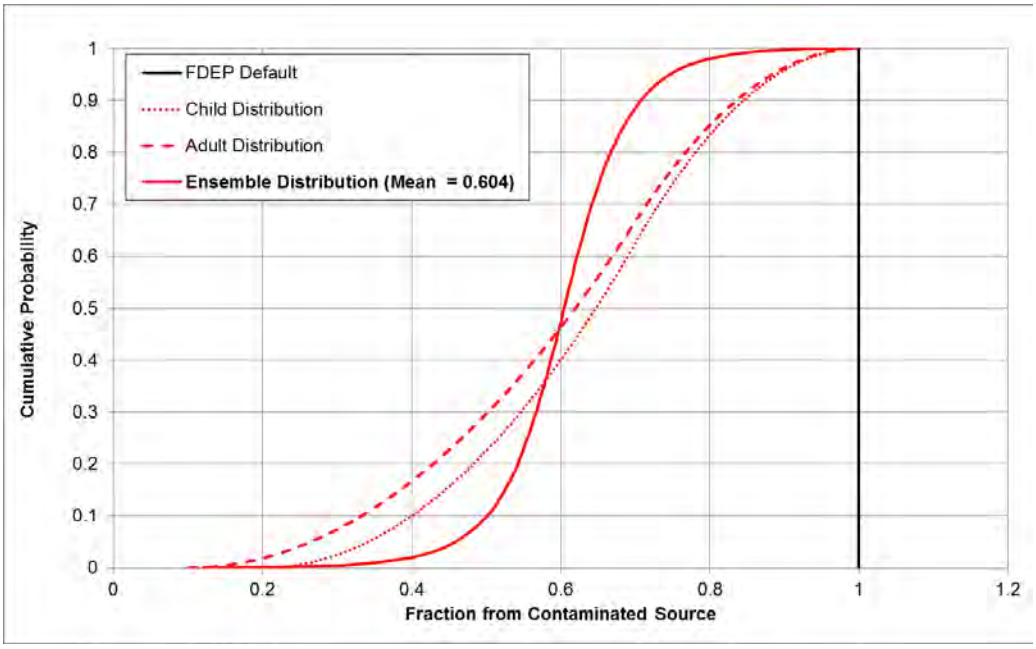


Figure 6. Cumulative probability distributions for fraction from contaminated source

3.2. Calculated Receptor Risk

FDEP required that the exposures beginning during childhood be analyzed separately from the full population. The applicable rules also mandate that the P90 LICR and HI estimates meet the target values for both cancer and non-cancer risk (10^{-6} and 1.0, respectively). The TEQ concentration was modified iteratively until these criteria were met for the subpopulation with exposure beginning prior to age 7. This resulted in a TEQ of 32 ng/kg. The average LICR at this TEQ for this subset of receptors is 5.7×10^{-7} and the 90th percentile (P90) LICR is 1.0×10^{-6} (Figure 7).

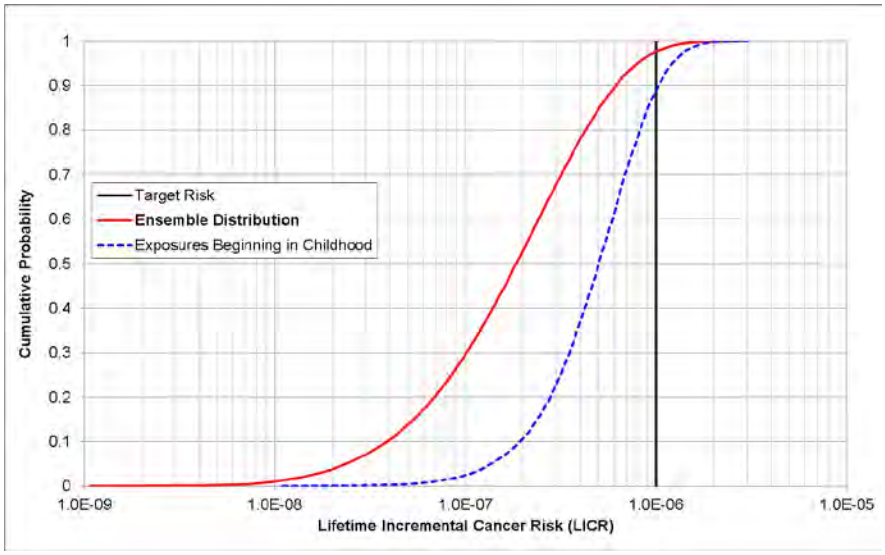


Figure 7. Cumulative LICR probability distributions for all receptors (red) and receptors whose exposure begins prior to age 7 years (i.e. "in childhood"; blue)

The non-cancer ensemble distribution has a mean Hazard Index of 0.07 and the P90 Hazard Index is 0.15 (Figure 8).

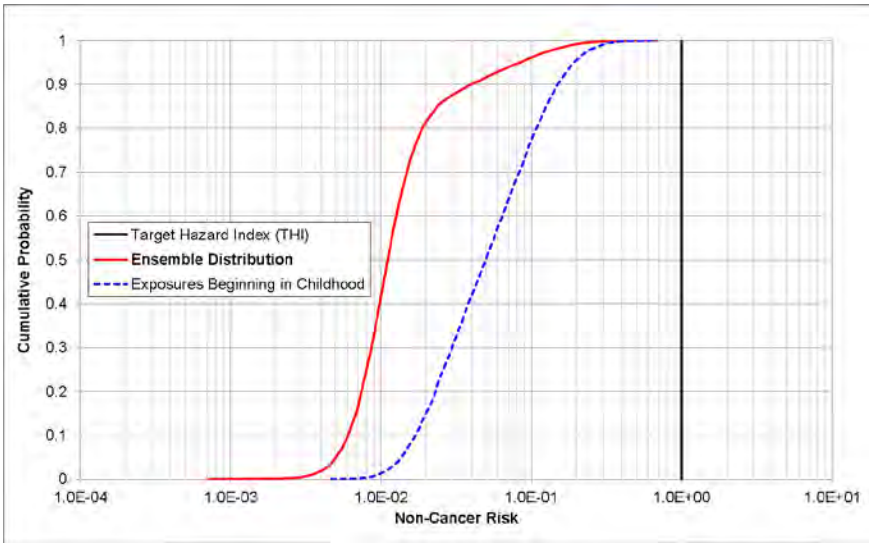


Figure 8. Cumulative non-cancer risk probability distributions for all receptors (red) and receptors whose exposure begins prior to age 7 years (i.e. "in childhood"; blue)

The LICR and non-cancer risk distributions for all 100,000 simulated receptors using the 32 ng/kg TEQ value are indicated by red lines in Figure 7 and 8. The LICR full-population ensemble distribution has an average risk of 2.7×10^{-7} and a P90 risk equal to 6.1×10^{-7}. The average receptor has a non-cancer Hazard Index of 0.021 and a P90 Hazard Index of 0.04.

3.3. Sensitivity Analysis

A series of ten sensitivity simulations were performed using perturbations to four uncertain exposure parameters: IR_o, EF, FC, and RBA_o. The 100,000 realizations were repeated using either a deterministic value or a different assumed autocorrelation value for one of the parameters. The ensemble LICR distributions from each sensitivity analysis revealed that the LICR estimates were relatively insensitive to changes to the auto-correlation values for EF (originally 20%), FC (20%), and IR_o (80%). Substitution of the conservative deterministic parameter values for IR_o (child: 50 mg/d, adult: 100 mg/d), EF (365 d/yr), FC (1.0), and RBA_o (1.0) for the variable parameter distributions (Figures 3 through 6) had greater influence on the entire population's LICR distribution.

4. DISCUSSION

4.1. Variability versus Uncertainty

Through negotiations it was decided not to treat uncertainty probabilistically, thus this PRA only accounts for variability in receptor exposure parameters. However, it is notable that there is considerable uncertainty in many of the factors used to estimate risk.

For example, 2,3,7,8-TCDD carcinogenicity, which is accounted for in the oral cancer slope factor (CSF_o) when calculating LICR, is subject to ongoing disagreement and debate in the scientific community. The data used to establish this CSF have been reviewed extensively and generally found not to reflect the current understanding of dioxin toxicology, which would warrant probabilistic treatment in a PRA such as this. Similarly, dioxin TEFs are defined by the WHO as very approximate, order-of-magnitude estimates due to the high level of uncertainty associated with the toxicity of each congener. Also, as noted previously (see Section 2.3), scientific studies have shown that bioavailability of dioxin from ingested soil varies by congener, receptor, and the composition and type of soil encountered by the receptor (i.e., a source of exposure uncertainty). From a technical (non-regulatory) standpoint, variability and uncertainty in risk factors which affect each congener, including TEFs, RBA_o, and possibly CSF_o, should not be ignored in PRAs.

Uncertainty could be considered in PRAs by performing a "two-dimensional" analysis. Such a two-dimensional PRA could be implemented using a nested simulation approach in which multiple sets of realizations—each considering only variability—are performed within an over-arching set of calculations which introduce uncertainty into parameters that are deemed uncertain.

4.2. Site-Specific Implications

The implications of acceptance of this PRA for the site owners are twofold. First, the remedial costs for cleaning up all areas with measured soil dioxin concentrations greater than 32 ng/kg TEQ will be substantially less than if FDEP had enforced the default standard of 7 ng/kg. Secondly, unnecessary disruption to a surrounding residential neighborhood, which has measured concentrations between 7 and 32 ng/kg, will no longer be necessary.

4.3. Selection of a Pre-Supposed Sensitive Subgroup

FDEP regulations state that the Target Risk (10^{-6}) must be met at the 90^{th} percentile, or that the Target Risk can only be exceeded for 10% of the receptor population. However, during discussions with regulators, FDEP placed an additional constraint on this PRA: that the Target Risk must also be met at the P90 level for the subset of exposures which begin during childhood. That is, only 10% of the approximately 16,800 receptors (out of 100,000) exposed to site soil prior to age 7 were allowed to exceed the 10^{-6} risk standard. Enforcing regulations in this way amounts to regulating based on the most exposed receptors within the (preconceived) most sensitive age group. Imposing this constraint results in a cleanup target (32 ng/kg TEQ) that is protective above the P90 level at a risk equal to 10^{-6}; the P90 risk for all receptors (6.1×10^{-7}) is considerably lower than the 10^{-6} Target Risk, which is achieved at the 98^{th} percentile of the full LICR distribution (Figure 7). Therefore, the accepted alternative SCTL from this PRA is still a conservative cleanup target.

4.4. Other Potential Implications of PRAs

Acceptance of this PRA and the science behind it is very encouraging for the future of risk assessment. Simple deterministic risk assessments provide little information regarding actual risks to receptor populations, and in the name of being protective, frequently make numerous conservative assumptions which compound and result in overly restrictive site decisions. Movement away from such deterministic risk assessments toward more informative and nuanced assessments backed by scientific studies should be the goal for the field of risk assessment. Approval of PRAs such as the one described above are progress in this direction, and need not be limited to dioxin in residential soil applications. For instance, non-residential (workplace) risk assessments could be performed using many of the same inputs and methods used in this PRA. Other environmental media and/or contaminants (e.g., arsenic in groundwater) could also be considered. Furthermore, other variables which are typically treated deterministically, but likely vary between receptors (e.g., inhalation rate) or are uncertain (e.g., cancer slope factors) could also be modeled probabilistically based on scientific results.

5. CONCLUSION

A probabilistic risk assessment for residential exposure to dioxin-contaminated soil was performed for a site in Florida. After considerable debate and deliberation with state regulators, an alternative SCTL of 32 ng/kg TEQ was accepted. The increase from the previous SCTL (7 ng/kg TEQ), which was based on numerous conservative assumptions regarding deterministic exposure parameters, will result in significant short-term and long-term cost savings to the site owners while also protecting the health of nearby residents. The regulators' insistence on selecting the 90^{th} percentile of the most susceptible portion of the receptor population—people whose exposure begins prior to age 7—means that the alternative SCTL is still protective to less than 10^{-6} target risk.

6. REFERENCES

Budinsky R, Rowlands J, Casteel S, Fent G, Cushing C, Newsted J, Giesy J, Ruby M, Aylward L. 2008. A pilot study of oral bioavailability of dioxins and furans from contaminated soils: Impact of differential hepatic enzyme activity and species differences. Chemosphere, 70:1774-1786.

Environmental Protection Agency (EPA). 2009 December. Review of State Soil Cleanup Levels for Dioxin. National Center for Environmental Assessment. Washington (DC).

Environmental Protection Agency (EPA). 2011. Exposure Factors Handbook: 2011 Edition (Final). Office of Research and Development, National Center for Environmental Assessment, US Environmental Protection Agency. [Cited 2011 September.] Available from: EPA/600/R-09/052F.

Exponent. 2005 February 24. Pilot Study Report: Oral Bioavailability of Dioxins/Furans in Midland and Tittabawassee River Flood Plain Soils.

Florida Department of Environmental Protection (FDEP). 2005 February. Technical Report: Development of Cleanup Target Levels (CTLs) for Chapter 62-777. F.A.C. Division of Waste Management, Florida Department of Environmental Protection.

Finley B, Fehling K, Warmerdam J, Morinello E. 2009. Oral bioavailability of polychlorinated dibenzo-p-dioxins/dibenzofurans in industrial soils. Human and Ecological Risk Assessment, 15:146-1167.

Johnson T, Capel J. 1992 August. A Monte Carlo Approach to Simulating Residential Occupancy Periods and Its Application to the General US Population. Research Triangle Park (NC): US Environmental Protection Agency, Office of Air Quality and Standards.

National Academy of Sciences. 2006. Health Risks from Dioxin and Related Compounds: Evaluation of the EPA Reassessment. The National Academies Press Washington DC.

SRC, Inc. 2010 December 20. Final Report: Bioavailability of Dioxins and Dioxin-like Compounds in Soil. Prepared for US Environmental Protection Agency by SRC, Inc.

Stanek E, Calabrese E, Zorn M. 2001. Soil ingestion distributions for monte carlo risk assessment in children. Human and Ecological Risk Assessment, 7(2):357-368.

U.S. Census Bureau. 2011. Current Population Survey, Annual Social and Economic Supplement, 2010. Internet release date: June 2011.

Van den Berg M, Birnbaum L, Bosveld A, Brunström B, Cook P, Feeley M, Giesy J, Hanberg A, Hasegawa R, Kennedy S, Kubiak T, Larsen J, van Leeuwen F, Liem A, Nolt C, Peterson R, Poellinger L, Safe S, Schrenk D, Tillitt D, Tysklind M, Younes M, Waern F, Zacharewski T. 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. Environmental Health Perspectives, 106:775-792.

TOXICOLOGY & ENVIRONMENTAL SIGNIFICANCE OF BENZALDEHYDE

East Coast Conference, October 2015

Christopher M. Teaf [1,2], Kevin W. Teaf [2] and Douglas J. Covert[2§]
[1]Center for Biomedical & Toxicological Research, Florida State University, 2035 E. Dirac Dr., Tallahassee, FL, 32310; [2]Hazardous Substance & Waste Management Research, Inc., 2976 Wellington Circle West, Tallahassee, FL, 32309

ABSTRACT

Benzaldehyde (C_7H_6O) is a colorless, to yellow tinted liquid that gives off an almond odor. It is fairly soluble in water (>1,000 mg/L), and is completely miscible in some organic solvents, including ethanol and diethyl ether. Primarily used in flavoring, fragrances, cosmetics, as well as some pharmaceuticals, benzaldehyde has been implicated in at least one high profile environmental contamination matter. It is classified in the category Generally Recognized as Safe (GRAS) according the USFDA, though it can cause lung, eye and dermal irritation at very high exposure levels, consistent with other members of the aldehyde class. The substance has an environmental half-life ranging from eight to thirty hours depending on the amount of moisture present in the environment. Benzaldehyde is slightly toxic to aquatic life. When evaluated by the National Toxicology Program, no evidence of carcinogenicity in mice was reported, and benzaldehyde actually has been reported to have carcinostatic or antitumor properties in some species. When absorbed either through the lungs or skin, it is distributed to high blood flow organs, metabolized to benzoic acid, and then excreted in the urine. Various state environmental agencies recommend soil exposure guidance values ranging from 5,000 mg/kg to 70,000 mg/kg for unrestricted residential use and non-residential use, respectively, while USEPA reports a Regional Screening Level (RSL) for benzaldehyde in soil of 7,800 mg/kg for residential soils, and 120,000 mg/kg for industrial soil. There is no drinking water standard for the substance, but a protective tap water Regional Screening Level of 1,900 ug/L has been developed by USEPA for unrestricted use. This paper provides scientific and toxicological information for benzaldehyde and discusses potential risks associated with its presence in the environment or in commercial products.

Keywords: benzaldehyde, health risk, exposure assessment, toxicology

1. INTRODUCTION

Benzaldehyde (benzoic aldehyde; benzene carbonal; artificial almond oil) is a naturally occurring chemical compound found primarily in almonds and in various fruits such as cherries, strawberries, apricots, plums, and peaches. The compound also is found in many household products and personal care items (air fresheners, shave gels, bath soaps, moisturizing gels/creams,

§ Corresponding Author: Douglas J. Covert, 2976 Wellington Cir West, Tallahassee, FL, 32309; Tel: 850-681-6894; dcovert@hswmr.com

dyes), pharmaceuticals (drugs), and as an additive for one or more types of tobacco products. It also is reportedly found in candies and drinks (e.g., Kool-Aid®, Life Savers®, Jolly Rancher®, wines) and is used as a solvent for oils, resins, and cellulose fibers.

Concentrations of benzaldehyde have also been found in wood smoke from fireplaces burning pine, cedar, oak, and ash wood. Concentrations have also been discovered in exhausts from engines burning simple hydrocarbons as well as gasoline-powered automobiles. Although these concentrations may be low, they are still detectable.

Benzaldehyde is not a persistent chemical in the environment, exhibiting a half-life in air of about 30 hours. Although it is broken down quickly by air and sunlight, it is possible for benzaldehyde to be carried with dust particles in the air and subsequently returned to the ground by wet and dry deposition. Even though benzaldehyde is readily biodegradable, improper disposal must be avoided. The compound penetrates the soil with ease and can result in impacts to groundwater and surface water.

2. PRODUCTION AND OCCURRENCE

Technical grade benzaldehyde typically is used as an intermediate in the production of other chemicals or products. Recently, about half of the production has been used to aid in the making of various flavor and fragrance chemicals such as cinnamaldehyde, amylcinnamaldehyde, hexylcinnamaldehyde and cinnamyl alcohol. Significant amounts of benzaldehyde are consumed by the pharmaceutical industry to synthesize pharmaceuticals such as ephedrine. Benzaldehyde is also commonly used in the dye and herbicide industries. Purified benzaldehyde is used as a flavoring agent especially for artificial cherry and almond flavors, totaling as much as 0.5 million pounds annually (USEPA 1985).

Historically, benzaldehyde was produced by liquid-phase air oxidation of toluene. The transformation rate of toluene was about 20% and the selectivity of benzaldehyde was 30%-50%. Benzaldehyde is manufactured industrially in the United States by Kalama Chemical, Inc (USEPA 1985). Kalama operates facilities in Kalama, WA, and Garfield, NJ. The production capacities of the two facilities in the mid-1970's were ~8 million and 3.5 million pounds per year, respectively (USEPA 1985). More recent estimates suggest a worldwide production volume for all manufacturers of approximately 30 million pounds per year (WHO 1996).

A number of studies have been conducted over several decades on various different types of water sources and the vast majority of samples have displayed at least the presence of benzaldehyde. Keith (1976) identified benzaldehyde in drinking water from the Carrollton Water Treatment Plant in New Orleans, LA at a concentration of 0.03 ug/L. Benzaldehyde has been detected (no concentrations reported) in various drinking waters in the Philadelphia, PA area (Florida-Spectrum Laboratories, 2014). Kawamura and Kaplan (1983) examined rainwater collected from the campus of UCLA in Los Angeles, CA and reported benzaldehyde concentrations in the of 1-3 ug/L range. Pellizzari et al. (1982) examined mother's milk from four urban areas (Bridgeville, PA; Jersey City, NJ; Bayonne, NJ; and Baton Rouge, LA) for the presence of environmental pollutants, and benzaldehyde was detected (no concentrations reported) in all eight samples that were tested. According to the USEPA (2015) Regional Screening Level (RSL) table regarding unrestricted ingestion of tap water, benzaldehyde in drinking water at a concentration as great as 1,900 ug/L does not represent a health concern.

Researchers from Rutgers University analyzed a variety of ambient air samples for benzaldehyde inside and outside of 36 homes in Central New Jersey. Benzaldehyde was detected inside 14 of 36 homes and outside 22 of 36 homes with mean concentrations of 0.25 ppm and 0.38 ppm, respectively (Andersen 2006).

The environment is especially at risk during transportation, destruction, and experimenting with benzaldehyde. Accidental spills may result in fire, explosion, and possible contamination of surrounding environmental media. Due to its boiling point and flash point, benzaldehyde is classified as a Class IIIA combustible liquid (per OSHA 29 CFR 1910.106). Proper disposal method for benzaldehyde is to burn it in a chemical incinerator equipped with an afterburner and air scrubber. The short-term effects of benzaldehyde can include the death of fish, birds, plants, and other animals depending on the amount released into the environment. After a spill, immediate steps should be taken to limit spread in the environment.

3. TOXICITY SUMMARY AND REGULATORY GUIDELINES

A great deal of the information available for benzaldehyde is from experiments performed on laboratory animals with only limited data involving human exposures. What is known for sure is that when benzaldehyde comes into contact with the eyes, dermis or inhaled into the lungs, it tends to cause severe irritation depending on the dose and time of exposure.

The Joint FAO/WHO Expert Committee on Food Additives found that delayed development and reduced fetal and postnatal pup body weights were observed in developmental toxicity studies with rats, mice, hamsters and rabbits, but only at very high exposure concentrations that were toxic to the mother (WHO 1996).

Benzaldehyde was evaluated by the National Toxicology Program (NTP 1990) and yielded no evidence of carcinogenicity in rats and some evidence of carcinogenicity in mice (forestomach papillomas and forestomach hyperplasia). These data are of uncertain value in that the human does not possess a forestomach. NTP does not list benzaldehyde as either a known or reasonably anticipated human carcinogen. In fact, benzaldehyde has been shown to exhibit carcinostatic properties, the slowing or inhibition of the growth of cancerous tumors, in lab animals and possibly in humans (Pettersen et al. 1983; Morse et al. 1995; MacEwen 1986; Taetle and Howell 1983; Kochi et al. 1980).

The European Food Safety Authority (EFSA) also has evaluated the potential carcinogenicity of benzaldehyde, and reported that carcinogenicity studies were negative and the substance is not genotoxic (EFSA 2010). Another recent study reported potential genotoxic and mutagenic effects on fruit flies (Deepa et al. 2012), although additional studies would be needed to confirm or refute those results.

The Agency for Toxic Substances and Disease Registry (ATSDR) comparison values (CVs) for benzaldehyde in residential soils are 5,000 mg/kg (parts per million or ppm) for a child and 70,000 mg/kg for an adult. These levels represent the amount of exposure the individual can experience without negative health effects. According to USEPA (2015), the Regional Screening Level (RSL) for this compound in industrial soils is 120,000 ppm, compared to the 7,800 ppm that is acceptable in residential soils. The Acceptable Daily Intake (ADI) is defined as the amount of a chemical to which humans may be exposed to on a daily basis over an extended period of time without suffering a harmful effect. For benzaldehyde, the ADI is 15

mg/day. The Reportable Quantity (RQ) value is used to determine the quantity of a hazardous substance for which notification is required in the event of a release. The Reportable Quantity (RQ) value for benzaldehyde is 1,000 pounds.

First aid for benzaldehyde should be applied following direct contact with the eyes, and one should immediately remove contact lenses, accompanied by flushing with large amounts of water for at least 15 minutes, occasionally lifting upper and lower lids. If skin contact occurs, remove contaminated clothing, and immediately wash contaminated skin with large amounts of soap and water. If inhaled, it is recommended to remove the exposed individual from exposure, initiate rescue breathing if breathing has stopped, and initiate CPR if the heart has stopped. The patient should be transferred promptly to a medical facility if symptoms persist.

4. CASE STUDY

Plaintiffs in an environmental exposure matter in Ohio alleged that exposure to benzaldehyde was responsible, at least in part, for an identified local childhood cancer cluster. The compound was indeed found in environmental samples collected by plaintiff experts, primarily in attic dust from several (but not all) of the plaintiffs' homes. The Ohio EPA, the Ohio Department of Health (ODH), the Sandusky County Health Department (SCDH), the USEPA, and the Agency for Toxic Substances and Disease Registry (ATSDR) all have evaluated the circumstances of the case to some extent, and none have concluded a plausible link between the observed cancers and benzaldehyde. A year-long air monitoring study found no concentrations of concern for VOCs nor metals in the area of interest, nor immediately outside of the defendant's manufacturing plant, and the defendant claims that benzaldehyde has never been a component of its core manufacturing process. Specifically related to the benzaldehyde detections in the attic dust, ATSDR commented that they do not evaluate attic dust because the attic is not an area where occupants spend significant periods of time. Further, ATSDR reviewers noted that the Plaintiff expert report suggesting "elevated" levels was incorrect due to the use of wet weight results expressed in units of ug/kg rather than dry weight in mg/kg, an units conversion error that inflates the concentrations by approximately 1,000 fold. As of February 2015, the Plaintiffs had withdrawn their case without prejudice, indicating that it may be refiled with the court within one year.

5. SUMMARY AND CONCLUSIONS

Benzaldehyde is a naturally occurring and synthetic chemical manufactured in large volume for use in flavorings, fragrances, pharmaceuticals, and as an intermediate in the manufacture of other chemicals. It is not especially persistent in the environment, but it does occur commonly in air, soil and water. Benzaldehyde does not express a notable degree of toxicity to animals or humans, and is not a recognized carcinogen. In fact, it is on the USFDA list of Generally Recognized as Safe (GRAS) chemicals.

6. REFERENCES

Andersen A. 2006. Final report on the safety assessment of benzaldehyde. Int J Toxicol, 25(1):11-27.

Deepa PV, Priyanka V, Akshaya S. 2012. Genotoxicity of benzaldehyde in *Drosophila melanogaster* using the Wing Somatic Mutation and Recombination Test (SMART) and protein profiling. Int J Med Clin Res, 3(6):195-198.

European Food Safety Authority (EFSA). 2010. Scientific opinion on Flavouring Group Evaluation 20, Revision 2 (FGE.20Rev2): benzyl alcohols, benzaldehydes, a related acetal, benzoic acids, and related esters from chemical groups 23 and 30. EFSA Journal, 8(7):1405-1540.

Florida-Spectrum Laboratories. Benzaldehyde chemical fact sheet [Internet] [cited 2014 Dec 24]. Available from: http://www.speclab.com/compound/c100527.htm

Kawamura K, Kaplan IR. 1983. Organic compounds in the rainwater of Los Angeles. Environ Sci Technol, 17(8):497-501.

Keith LH. 1976. Identification and analysis of organic pollutants in water. Ann Arbor (MI): Ann Arbor Science Publishers, Inc., p. 329-373.

Kochi M, Takeuchi S, Mizutani T, Mochizuki K, Matsumoto Y, Salto Y. 1980. Antitumor activity of benzaldehyde. Cancer Treat Rep, 64(1):21-23.

MacEwen EG. 1986. Anti-tumor evaluation of benzaldehyde in the dog and cat. Am J Vet Res, 47(2):451-452.

Morse MA, Kresty LA, Toburen AL. 1995. Inhibition of metabolism of 4-(methylnitrosamino)-1-(3-pyridyl)-1-butanone by dietary benzaldehydes. Cancer Lett, 97(2):255-261.

National Toxicology Program (NTP). 1990. Toxicology and carcinogenesis studies of benzaldehyde (CAS No. 100-52-7) in F344/N rats and B6C3F$_1$ mice (gavage studies). U.S. Department of Health and Human Services. Technical Report Series No. 378.

Pellizzari ED, Hartwell TD, Harris BS III, Waddell RD, Whitaker DA, Erickson MD. 1982. Purgeable organic compounds in mother's milk. Bull Environ Contam Toxicol, 28(3):322-328.

Pettersen EO, Nome O, Rønning OW, Oftebro R. 1983. Effects of benzaldehyde on survival and cell-cycle kinetics of human cells cultivated in vitro. Eur J Cancer Clin Oncol, 19(4):507-514.

Taetle R, Howell SB. 1983. Preclinical re-evaluation of benzaldehyde as a chemotherapeutic agent. Cancer Treat Rep, 67(6):561-566.

United States Environmental Protection Agency (USEPA). 1985. Health and environmental effects profile for benzaldehyde. Cincinnati (OH): Office of Research and Development. EPA/600/X-85/395.

United States Environmental Protection Agency (USEPA). 2015. Regional screening level (RSL) summary table - benzaldehyde. Revised 2015 June.

World Health Organization (WHO). 1996. Toxicological evaluation of certain food and additives. WHO Food Additives Series 37: benzyl acetate, benzyl alcohol, benzaldehyde, and benzoic acid and its salts. The 46th Meeting of the Joint FAO/WHO Expert Committee on Food Additives (JECFA). International Programme on Chemical Safety (IPCS).

SUCCESSFUL TCE BIOREMEDIATION IN A LOW PERMEABILITY FORMATION USING SOLAR POWERED RECIRCULATION

East Coast Conference, October 2015

Sami A. Fam, Ph.D., P.E. L.S.P.[1§], David Falatko, P.E.[2]

[1]*Innovative Engineering Solutions, Inc., 25 Spring Street, Walpole, MA, 02081*; [2]*Innovative Engineering Solutions, Inc., 95A Ocean Street, South Portland, ME, 04106.*

ABSTRACT

A 3-acre chlorinated VOC plume (approximately 700 feet long) is currently being remediated by enhanced anaerobic dechlorination (EAD). The remedial objective is treatment of contamination present at depths of approximately 10-100 feet in a low permeability formation. Chlorinated volatile organic compounds (CVOCs) were historically released at the site between 1955 and 1970 at a remote location in an area of a designated drinking water aquifer. The main source area was excavated and disposed off-site by others. A long and narrow VOC plume remained, however, within a silty sand/clay formation at depths varying between 10-100 feet. Attempts by others to remediate the plume by batch amendment injection failed due to poor additive distribution. In 2010, a small solar powered low-flow groundwater recirculation system was installed near the former source area to distribute electron donor (methanol due to the site's low hydraulic permeability) and bioaugmentation culture. Upon demonstrating that groundwater and electron donor can be recirculated in this low permeability formation, the system was incrementally expanded to cover the entire 3-acre plume. In the full scale system, approximately 10,000 gallons of alcohol are batch injected per year as a dilute 20% non-flammable solution. Groundwater is extracted from 6-8 wells and injected into approximately 20-30 wells on a semi-continuous basis based upon available power. The system is equipped with back up batteries that can provide operation for approximately 20 hours without sunlight. The system recirculates approximately 1-2 pore volumes of untreated groundwater per year. Separation of electron donor addition and groundwater recirculation works well for low permeability sites since these sites are the most prone to clogging with biofouling. The relatively concentrated alcohol feeding serves as a well disinfection and cleaning method followed by water flushing. Alkalinity to buffer pH has also been added on a batch basis using diluted potassium hydroxide flushed with recirculated groundwater. Results have been outstanding. In the upgradient portion of the site where the system operated from 2011 to 2013, VOCs (2-5 mg/L range) have been nearly 100% converted to ethene. In the downgradient portion of the site, where operation has only been for 1-2 years, TCE is over 99% converted to daughter products and total organic carbon is present at concentrations in excess of 100 mg/L at almost every monitoring location.

Keywords: groundwater recirculation, bioremediation, anaerobic dechlorination, solar, sustainable remediation.

[§]Corresponding author: Sami A. Fam, Innovative Engineering Solutions, Inc., 25 Spring Street, Walpole, MA, 02081; Tel: 508-623-1221; sami@iesionline.com

1. INTRODUCTION

The design of a subsurface bioremediation system is based upon established methods to enhance anaerobic microbial growth for CVOC biodegradation combined with controlling physical processes of flow in porous media to deliver amendments and create conditions suitable for enhanced microbial growth. These conditions must be maintained for sufficient time to allow for microbial growth (anaerobic growth is slower than aerobic) and for contaminant desorption and diffusion into the dissolved phase. As such, the following are the basic design goals for an EAD project:

1) Select and properly distribute the additives (electron donor and bioaugmentation cultures) in sufficient amounts;
2) Adjust and maintain a neutral pH; and,
3) Maintain appropriate geochemical and amendment distribution conditions for a suitable time until remediation is complete.

The last point is often overlooked in remedial additive approaches; it is essential to maintain elevated total organic carbon (TOC) for sufficient time to allow for desorption of VOCs from low permeability zones (matrix back diffusion) into higher permeability zones where they can be degraded. It is also critical to understand that VOCs are often partitioned into lower permeability zones and that project success rests upon combatting matrix diffusion effects (Fam and Kidd 2005; Falatko et al. 2010; Adamson et al. 2011; Fam et al. 2012). It is this slow back diffusion process that has doomed many remedial approaches, most notable pump and treat systems.

In implementing the remediation program described in this paper, these design principles were strictly followed. The following paragraphs describe the project site and its remediation history. In the subsequent sections, we highlight the remedy design procedures and the remediation results.

Handling of chlorinated solvents (roughly 50 years ago) at the former facility resulted in releases and impacts to groundwater at a recently redeveloped site in New England. The site is underlain by low hydraulic conductivity fine sand/clay and silty soil. TCE contamination appears to have migrated via flushing by precipitation events and diffusion/advection out of the upper fine sand and silt unit into the underlying, slightly higher hydraulic conductivity, fine sand unit. When the TCE reached the fine sand unit, it migrated via advective groundwater flow to the west. The fine sand unit dips to the west becoming the lower fine sand and silt unit in the western portion of the site. Over time, plume length grew to approximately 700 feet and the contamination traveled to the west and was detected at depths in the vicinity of 100 feet at the most downgradient locations. Hydraulic testing indicated that extraction well yields are low and are generally in the range of 0.02 to gpm/ft. of well (4-inch diameter) screen.

Between 2007 and 2010, the main source area contaminated soil was excavated and properly handled off site. A batch sodium lactate injection program (into approximately 25 injection wells) was undertaken by others to remediate the residual 700 foot long VOC plume. The batch injection program did not significantly alter residual VOC concentrations due to inadequate additive distribution.

In 2010, a proof of concept pilot program was initiated to evaluate groundwater recirculation for additive distribution at this low permeability project site.

2. SYSTEM DESCRIPTION

The EAD groundwater recirculation pilot program began in October 2010. The EAD pilot test involved recirculation of groundwater within a test area to distribute the added electron donor. As shown in Figure 1, initially groundwater was extracted from wells REW-1 and REW-2 and was injected into wells RIW-1, RIW-2, and RIW-3. The REW and RIW series wells are 4 inches in diameter and are drilled to an approximate depth of 45 feet. V cut 0.010" slot-screen (PVC) is located at approximate depths of 15 to 45 feet below ground surface. The pilot system operated for approximately 68 days during the Fall of 2010 and recirculated approximately 264,000 gallons of groundwater (approximately 1.4 pore volumes of the area to be treated). During the Fall of 2010, approximately 31 drums of alcohol, pH buffers (potassium hydroxide) and inorganic minerals (di-ammonium phosphate), and NJ-14 bioaugmentation culture were recirculated with the groundwater. The additives were added during two feeding events, but the groundwater was recirculated continuously. The recirculation pilot system consisted of 6 panels at the outset powered by solar panels, and all interconnecting piping was initially above ground hose. The pilot test system was shut down during the winter months. Groundwater recirculation resumed in the Spring of 2011.

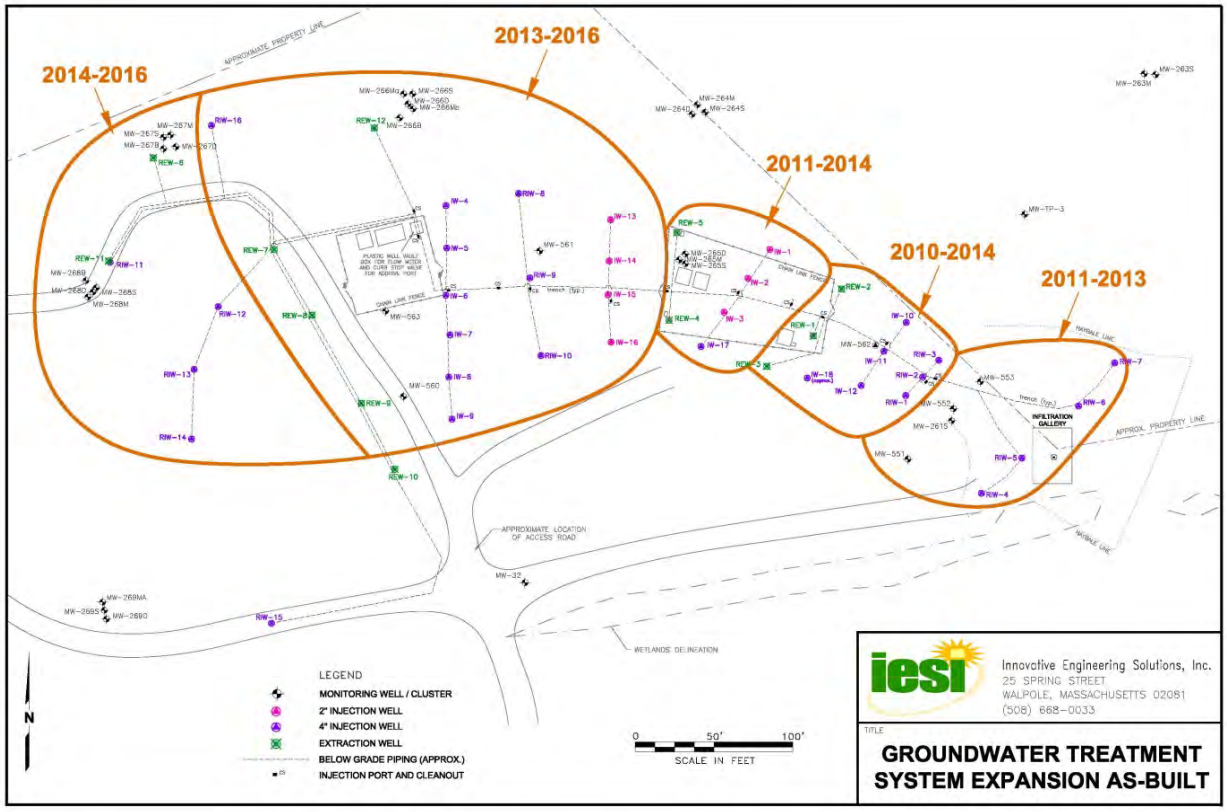


Figure 1. Project site map and area remediation timeline

The initial success (within 4-6 months) of the pilot EAD program lead to its expansion and burial of all interconnecting piping to enable full year operation. In an effort to minimize capital expenditures, the 3-acre parcel was incrementally remediated over time as shown in Figure 1.

The extraction and pumping well configurations were periodically adjusted to enable focused area remediation.

The final, fully expanded (2015) system configuration consisted of 12 extraction wells (only 6-7 operate at any one time due to available solar power) and 20-40 injection locations (Figure 1). Extraction and injection wells were both constructed as fully screened wells, crossing the interbedded fin and coarse-grained zones with a single well and open screen. This promotes the best amendment distribution and allows for the sustained contact of amendments with the fine-grained formation to counteract the effects of matrix back-diffusion of VOCs.

The expanded remediation system (photo in Figure 2) is powered by 36 solar panels for an approximate total generation of 11.5 KW direct current (DC). Over the course of the project, we estimate that approximately 40,000 kWh of alternating current power has been generated. Based upon an average of 1.2 lbs. of carbon dioxide generated per kWh, we estimate that this solar powered system has reduced potential carbon emissions by 48,000 lbs. The solar panels store the generated power in a series of batteries, and the extraction pumps (DC pumps) operate off battery power. The batteries used were standard 12 VDC lead-acid, deep-cycle marine batteries, selected for their low cost, availability, and known maintenance requirements. The entire system used approximately 40 batteries each at 12 VDC, then connected in series and parallel to provide 24 or 48 volts to the well pumps. The well pumps used initially were 48 volt stainless steel well pumps, but these were subject to wear with iron fouling and silt, and were replaced with 24 volt plastic pumps which were less expensive and could be easily rebuilt or replaced as needed.

Figure 2. Photo of the upgradient treatment area

3. OPERATIONAL RESULTS

Through September 2015, approximately 11 million gallons of groundwater have been recirculated by the remediation system. Approximately 43,000 gallons of electron donor (mainly methanol) have been added along with 1,000 lbs. of di-ammonium phosphate (DAP), 600 gallons of 50% potassium hydroxide (KOH), and 90 gallons of bioaugmentation culture (NJ-14 mixed culture). The methanol was injected as a 20% solution followed by continuous recirculation. The high strength alcohol solution acts as a temporary disinfectant for line and well cleaning and after dilution once recirculation restarts, it acts as an electron donor. As such, methanol is considered an almost ideal additive. In general, feedings were conducted monthly. The KOH was injected as a 5% solution on a batch basis and the DAP was injected as a dilute solution during the first two years of the project. Bioaugmentation cultures were manually added to the various injection wells.

Selected, but representative, data is graphed in Figures 3 to 5 (at the end of this manuscript). Each figure is for a single well and is made up of four separate graphs to allow a simultaneous evaluation of biogeochemical and VOC data. In general, the data indicate excellent VOC reduction, electron donor distribution, sulfate reduction, methane generation, and dechlorination of parent compounds to breakdown products (leading to ethene/ethane). The data often indicate initial electron donor induced desorption of VOCs (often with increases of VOCs), followed by dechlorination of the chlorinated VOCs. The degradation of the VOCs in both the fine and coarse-grained portions of the aquifer was completed with the prolonged and sustained electron donor contact with the soil, promoting matrix back diffusion of VOCs into the higher permeability formation and their subsequent degradation. Once the CVOC levels are reduced, there is no rebound effect from back diffusion or desorption of VOCs.

Near complete dechlorination can be seen in the upgradient/mid-plume wells such as MW-562, REW-1, REW-4, REW-5, MW-261S, MW-551, MW-552, MW-553, MW-265M, MW-560, MW-561, and MW-563. The more downgradient wells have not been in contact with the TOC as long, and as such, degradation is incomplete, but following the same trends as the more upgradient wells that are now fully remediated (Figure 6). The expectation is that site remediation will be completed in 2016 and no VOC rebound is expected. It has been our experience at over 40 other project sites that rebound does not occur if groundwater recirculation with sustained electron donor addition is practiced.

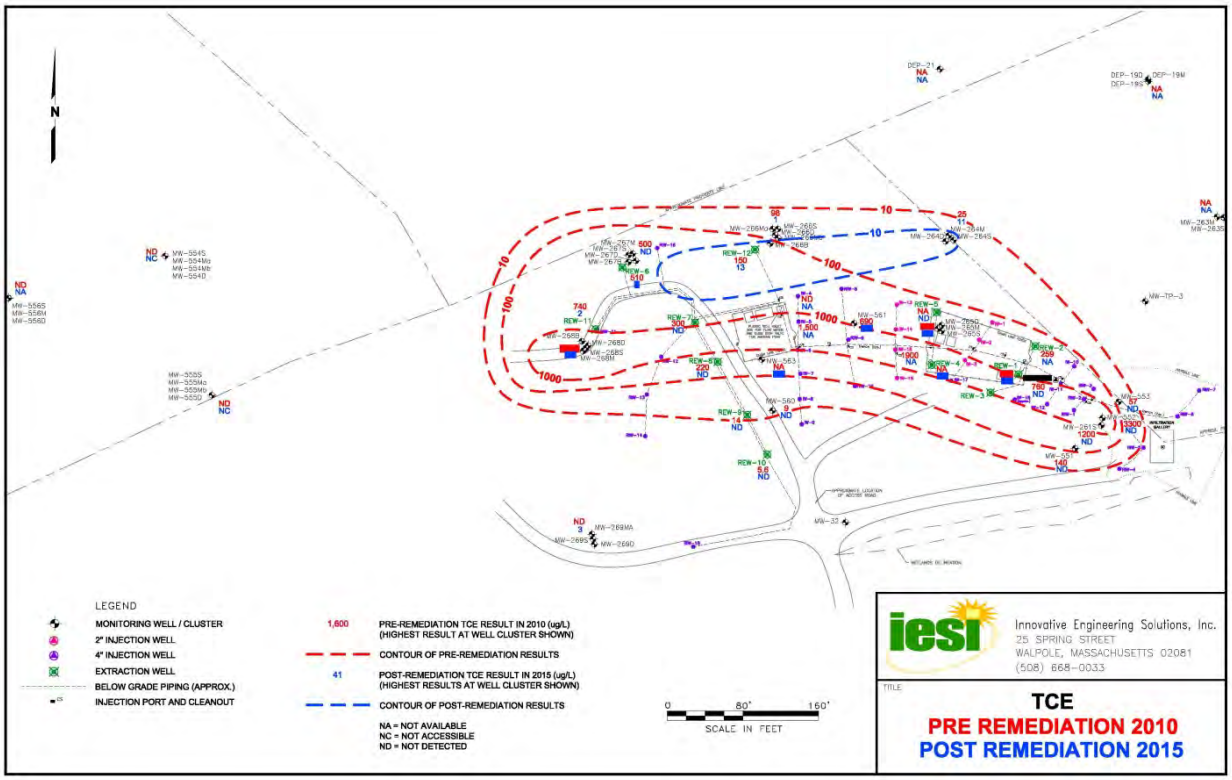


Figure 6. Pre and post remediation TCE iso-contour map

4. CONCLUSIONS

The benefits of groundwater recirculation systems for EAD far outweigh alternate approaches that use batch injection and should be considered for all EAD systems. The basic design approach outlined previously provides the methods to account for the aquifer characteristics and implement EAD remediation in reasonable time frames. The increased capital costs and associated complexity of EAD recirculation systems is justified in that it provides faster and more complete remediation and an overall lower project cost by shortening the duration of remediation, monitoring, and associated project management. The sustained elevated TOC concentrations in the groundwater are essential to affect matrix back diffusion and to enable remediation without VOC concentration rebound. Batch injection approaches can rarely accomplish this feat and are prone to poor treatment and/or VOC concentration rebound.

This project example highlights intelligent remedy selection and implementation procedures in a sustainable manner. Contaminants are nearly 100% biodegraded without use of electrical line power and associated carbon pollution. In consideration that recirculation only needs to turn over groundwater volumes (1.5 to 4 pore volumes per year) in a specific time period, down time due to lack of sunlight is not critical. As such solar powered groundwater bioremediation systems represent an ideal technology and energy source combination.

5. REFERENCES

Adamson DT, McGuire TM, Newell CJ, Stroo H. 2011. Sustained treatment: Implications for treatment timescales associated with source-depletion technologies. Remediation Journal, 21(2):27-50.

Falatko DM, Fam SA, Pon G. 2010. Applications and benefits of groundwater recirculation for electron donor delivery and pH-adjustment during enhanced anaerobic dechlorination. Proceedings of the Annual International Conference on Soils, Sediments, Water and Energy, 16(8).

Fam SA, Kidd DF. 2005. Delivery, delivery, delivery - Slow release vs. injection vs. groundwater recirculation. Battelle Bioremediation Symposium, In-Situ and On-Site Bioremediation. Baltimore (MD).

Fam SA, Falatko DM, Mountain S, Higgins JE, Pirelli AJ, Gaudette M. 2012. A universal design approach for in-situ bioremediation developed from multiple project sites. Remediation Journal, 22(4):49-74.

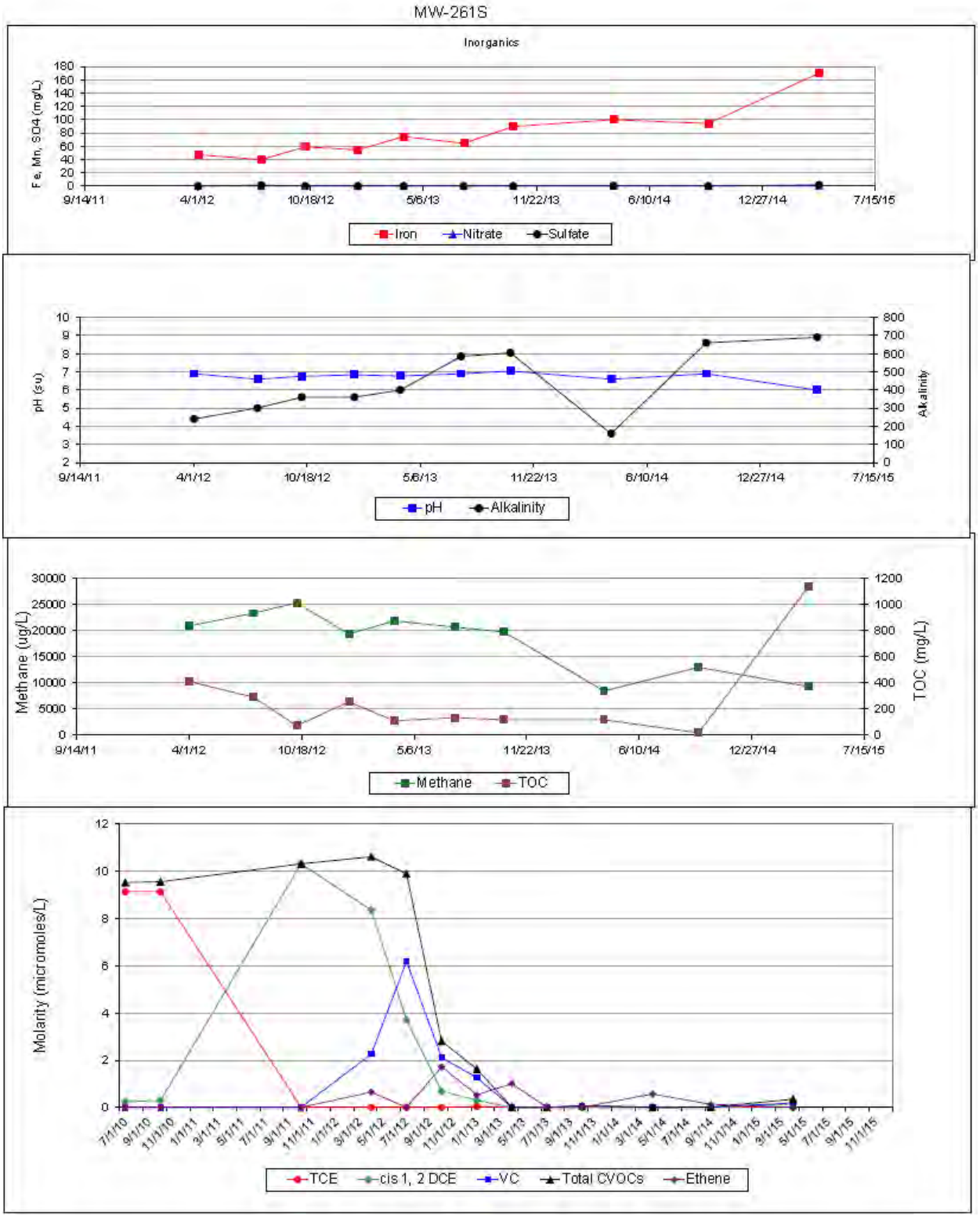


Figure 3. Remediation monitoring data for one of the more upgradient monitoring wells

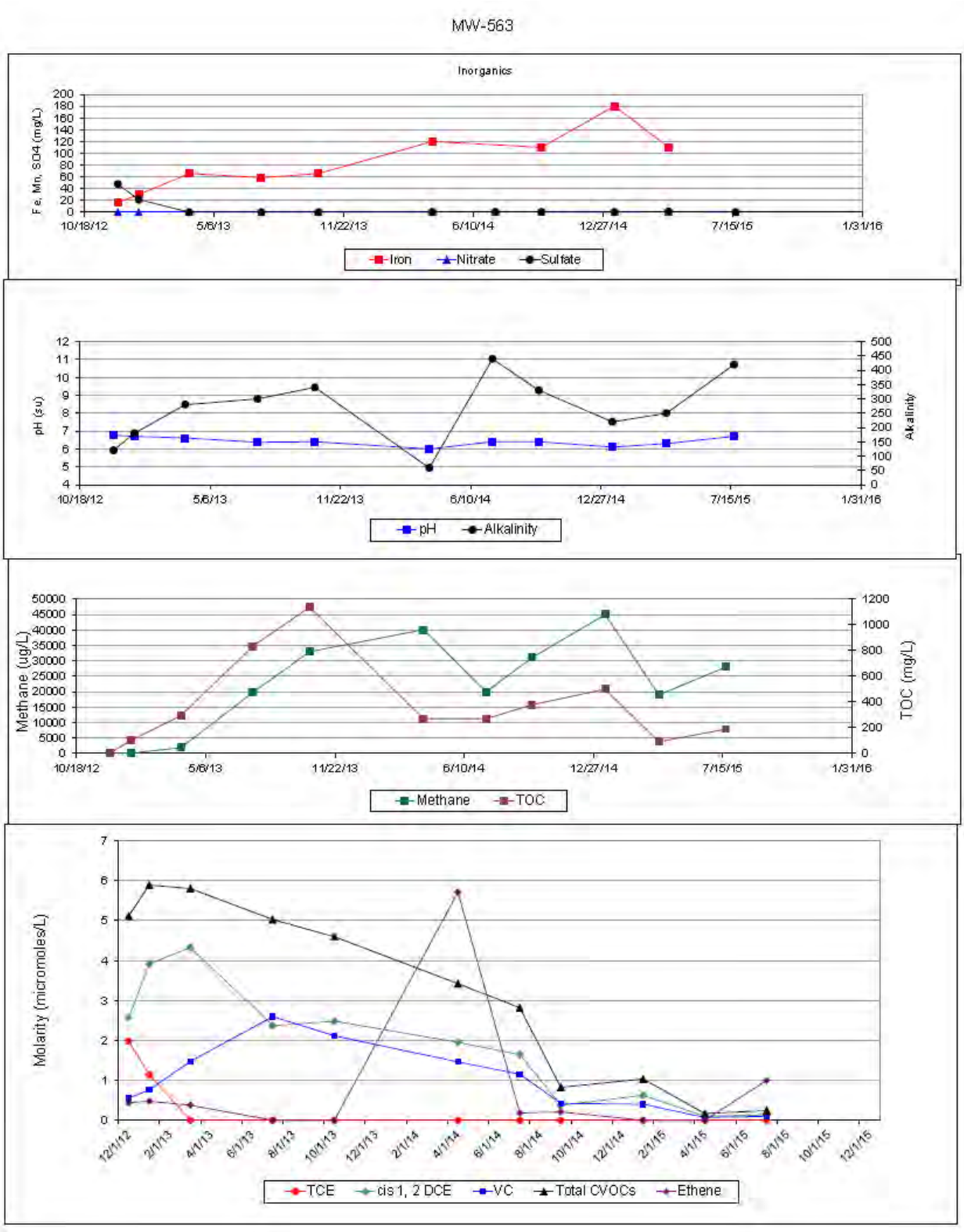


Figure 4. Remediation monitoring data for a mid-plume monitoring well

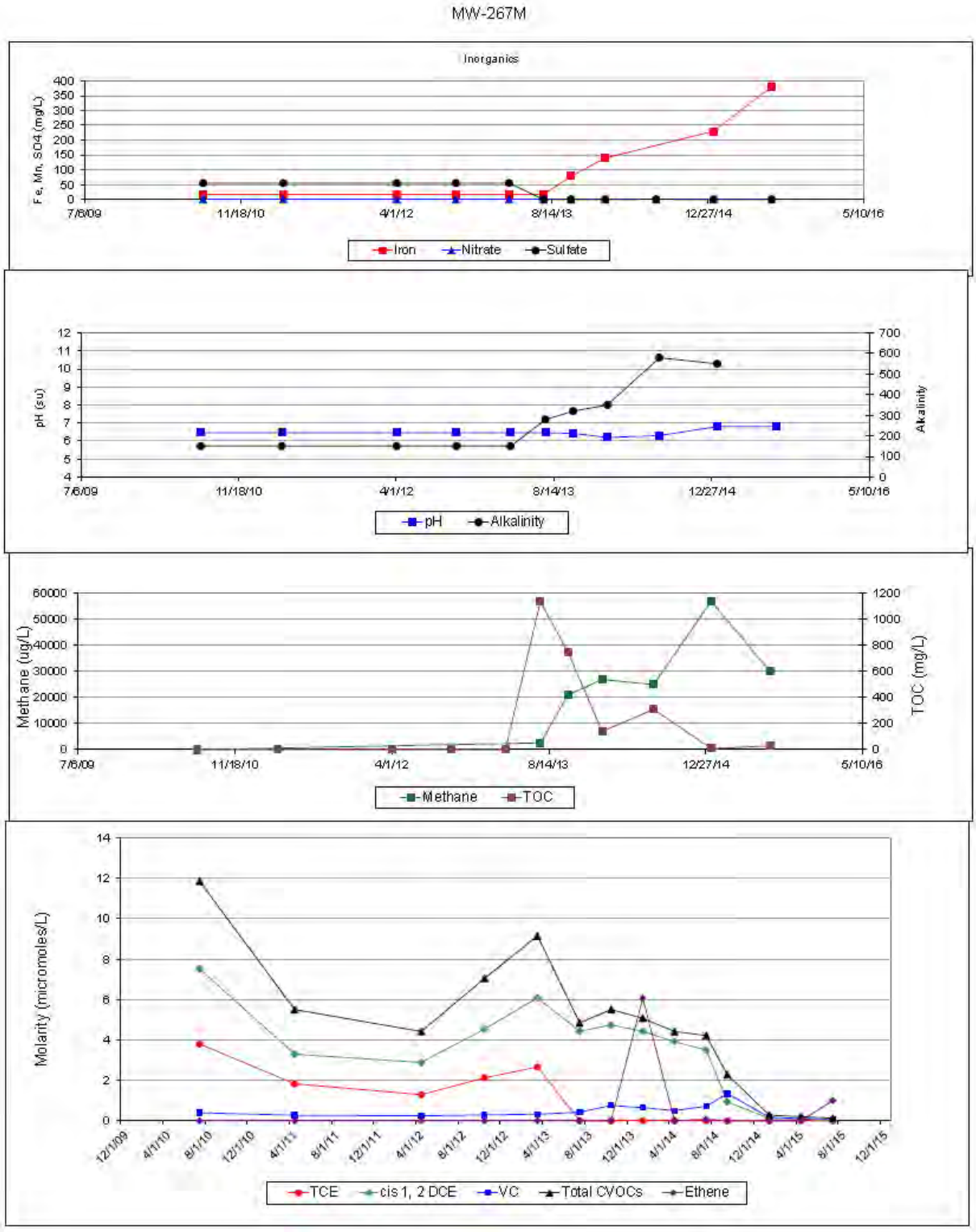


Figure 5. Remediation monitoring data for a downgradient monitoring well

UNSUSTAINABLE WATER PRACTICES IN U.S. HIGHER EDUCATION RESIDENCE HALLS

East Coast Conference, October 2015

Dr. Nakisa Alborz[1§], LEED AP BD+C
[1] *Chair, Interdisciplinary Engineering Department, Wentworth Institute of Technology, Boston, MA.*

ABSTRACT

Worldwide depletion of resources has brought many sustainability issues to the forefront, including the consumption of potable water. Based on various studies, the third largest consumption of water is for indoor use. The uppermost consumption has shown to be for flushing and personal hygiene. This paper compares the water consumption of dormitories to identify whether these buildings show sustainable use of water in practice. Due to different categorization of building typologies in varied water-use studies, the identification of water consumption in dormitories is problematic. Three LEED and six non-LEED dormitories located in the United States, serving a total of over 2,700 students, were selected for this comparative study. Since the International and Uniform Plumbing Codes do not require designers to calculate total water consumption, engineer's metrics have been computed and compared to the actual consumption of the nine dormitories. Design water use assumptions outlined by LEED were also compared to perceived consumption behaviour of occupants through a user survey in one LEED dormitory. Finally, a comparison between the projected design cases and actual water consumption both in LEED and non-LEED dormitories is reported.

Keywords: sustainable buildings, LEED, water consumption, dormitories, higher education

1. INTRODUCTION

This study aims to know and compare the indoor water use of LEED and non-LEED. It addresses several scopes including: identifying indoor water consumption in dormitories, comparing LEED to non-LEED dormitories, assessing LEED modeled case projections with actual water consumption, and comparing actual water consumption to developed engineer's metrics.

In the US, the United States Geological Survey (USGS) is the responsible authority for the collection of data about water use. Reporting of domestic water consumption from self-supplied and public-supplied sources is among the goals of the USGS. From the total water withdrawn for all uses in the US (1.5 trillion litres/day), domestic (residential) water use is the third largest

[§]Corresponding author: Nakisa Alborz, Interdisciplinary Engineering Department, Wentworth Institute of Technology, Boston, MA

category after thermoelectric power generation and irrigation, and has an estimated value of 111.3 billion litres/day (USGS 2005). Domestic applications typically include drinking, food preparation, washing clothes, dishes, flushing toilets and outdoor applications (watering lawns and washing cars). The average consumption for domestic use varies per state from 193 litres per person per day (LPD) in Maine to 715 LPD in Nevada, with the national average at 375 LPD (USGS 2005). A similar value is reported by the Environmental Protection Agency water sense program indicating 379 LPD consumed on average, of which 70% (265 LPD) is for indoor purposes (EPA, 2013).

Categorical disparities (commercial versus domestic) of dormitories make the isolation of dormitory water consumption problematic (USGS 2005; US-DOE 2013 a and b). In fact, USGS does not explicitly categorize building types, hence a lack of clarity whether dormitories fall under commercial or residential buildings exists. Commercial water use data was not collected by USGS in the 2000 and 2005 reports (USGS 2000; 2005), whereas in the 1995 report, it had been categorized for the following typologies of buildings: hotels, motels, restaurants, office buildings, other commercial facilities, and civilian and military institutions (USGS 1995). However, previous building types are particularly different from dormitories and their water consumption values could not be adapted to dormitories. Therefore the residential value suggested by USGS seems more applicable to dormitories although it includes outdoor applications such as watering lawns and washing cars. USDOE categorizes dormitories under lodging, a commercial category. However, the USDOE relies on the USGS report for water use reporting per sector; therefore, no explicit data on the water consumption of dormitories exists. The inconsistency between USGS and USDOE results in a misrepresentation of water consumption data in dormitory applications.

The European Commission (EC) through the DG-ENV Protection of Water Environment compiles data on water use in every member state (Mudgal and Lauranson 2009). A percentage between 60% and 80% of public supply water is used for domestic applications, with personal hygiene and flushing accounting for 60%. Case studies from different member states were analyzed based on metering information to generate domestic water consumption metrics resulting in an average residential domestic consumption of 168 LPD. In the water use report, dormitories are not explicitly categorized (Mudgal and Lauranson 2009). However, this report identifies educational buildings in the non-residential public sector without further specifying educational building typologies. In reviewing the water use in public buildings, where dormitories may be included, a lack of water consumption benchmarking data exists.

As it is evident, large differences between US and EU data collection and categorization exist. This compounds the problem of isolating dormitory water consumption. This study will assess and compare the water consumption in some US dormitories. In doing this, the different uses of water, such as washing dishes and clothing, flushing toilets, showering, drinking, and food preparation will be considered (Vickers 2001; Schleich and Hillenbrand 2009). Readers should be aware that the water consumption can be influenced by various factors including geographical location, climate, culture, gender, and occupant behaviour (Vickers 2001; SIU 2002; Hurlimann 2006; Balling et al. 2007; Randolph and Troy 2008; Alshuwaikhat and Abubakar 2008; Schleich and Hillenbrand 2009; Vinz 2009; Kats 2010; Elliot 2013), and for these reasons, this study will consider different dormitories.

This paper is composed of four sections: the next section will show the methodology of data collection, then section 3 will discuss the results, and finally some conclusions will be reported.

2. METHODOLOGY

Three LEED and six non-LEED dormitories, varying from 3 to 62 years of age, have been included in this study. The research methodology involved the collection of various specifications including: number and gender split of students served, flow fixture consumption values, actual water meter readings, and LEED documentation pertaining to water efficiency (WE) credits in LEED certified dormitories. Data was gathered from the designers, facilities departments, and residential life offices of the various higher education (HE) institutions. All dormitories are located in the US with eight in the Northeast and one on the West coast. For the purposes of anonymity, the dormitories have been designated through acronyms. Table 1 provides an overview of the nine dormitories studied.

Monthly actual water meter readings were collected for EH, PS, WT, MH1, MH2, MH3, HH, and KH, and quarterly actual water meter readings for CSC. The average number of students served per year was used to develop the litres per person per day (LPD) metric used in the study to compare dormitory water performance. Dormitories EH, CSC, WT, MH1, MH2, MH3, HH, and KH are located in the Northeast, experiencing cold to mix-humid climates, whereas dormitory PS is located on the West coast, experiencing a hot-dry climate. Typically, the peak water consumption occurs in summer, whereas the off-peak period is in winter (AWWA 1999). The weather in the US followed typical patterns in the years from 2002 to 2009, and from 2011 to 2013; reversely, the coldest winter was experienced in 2010, and in 2012 there was record heat in the summer and also mildest winter (Hansen et al. 2013; NWS, 2013).

Table 1. Overview of dormitories (F=female; M=male; *based on USDOE, 2013a)

Dormitory name	LEED certification	Dormitory age (yrs.)	No. of occupants	Gender split (%) (F/M)	Location	Building America zone*
EH	LEED-Gold	5	232	F=31% M=69%	Northeast	Cold
CSC	LEED-Gold	3	450	F=53% M=47%	Northeast	Mixed-Humid
PS	LEED-Silver	3	622	F=44% M=56%	West Coast	Hot-Dry
WT	Non-LEED	11	475	F=18% M=82%	Northeast	Cold
MH1	Non-LEED	62	284	F=60% M=40%	Northeast	Cold
MH2	Non-LEED	52	190	F=49% M=51%	Northeast	Cold
MH3	Non-LEED	47	190	F=60% M=40%	Northeast	Cold
HH	Non-LEED	54	163	F=50% M=50%	Northeast	Cold
KH	Non-LEED	52	191	F=53% M=47%	Northeast	Cold

Flow fixture consumption values were also collected to highlight differences in technologies used between dormitories. Non-LEED flow fixture data was collected from the HE facilities departments and walkthroughs, while for LEED dormitories documentation was collected from designers. Dormitory age was also explored as newer dormitories are less likely to experience plumbing leakage related issues and may have implemented higher efficiency fixtures.

2.1 Engineer's Metrics

The International and Uniform Plumbing Codes do not require designers to calculate total water consumption of buildings (ICC, 2009; IAMPO, 2009), therefore engineer's metrics were developed based on two separate reports: the EC report providing European metrics (Mudgal and Lauranson 2009), and the AWWA report, providing guidance for US metrics (AWWA, 1999).

The AWWA report values are based on data from over 1000 households in 12 study sites around the US. The data includes historic billing records and detailed mail surveys broken into two sets to capture winter (off-peak) and summer (peak) indoor water consumption. The AWWA water end use findings are as follows: 70 LPD toilet use, 57 LPD clothes washer, 44 LPD shower use, 41 LPD faucet use, 36 LPD leaks, 5 LPD baths, 4 LPD dishwasher, and 6 LPD other domestic use (AWWA 1999). In calculating the comparative AWWA metric, the value applicable to dormitories was assessed to be 212 LPD (including toilet use, clothes washer, shower use, and faucet).The EC report values are based on information collected from local case studies in different European member states through the involvement of stakeholders and a literature search. Findings in water using products of residential buildings in European countries indicate the following per use: 41 LPD toilet, 26 LPD clothes washer, 37 LPD showers, 29 LPD faucet, 10 LPD dishwasher, and 11 LPD outdoor use (Mudgal and Lauranson 2009). In calculating the comparative EC metric, the value applicable to dormitories was assessed to be 143 LPD.

3. RESULTS AND DISCUSSION

3.1 Average Overall (LEED and non-LEED) Actual Water Consumption

As indicated in Table 2, the overall range of actual LEED and non-LEED dormitory water consumption fell between 85-175 LPD, with an average of 144 LPD and a *stand. dev.* of 34 LPD. Comparing the average consumption (144 LPD) to the EC and AWWA engineer's metrics, the consumption was higher by almost 1% and 32%, respectively. Compared to US metrics, all the dormitories performed well, including non-LEED ones. On the other hand, as compared to European standards, the average overall savings were minimal at almost 1%.

Table 2. Average overall water consumption results in LPD (litres per person per day)

Dormitory	Data Range Dates	observations 'n'	Actual Avg. Consumption (LPD)	Standard Deviation of Dormitory Data Set (LPD)	Comparison Actual to EC Engineer's Metric (143 LPD)	Comparison Actual to US Engineer's Metric (212 LPD)
EH	Sept. '08-June '12	46	85	52	- 41%	- 60%
WT	Jan '02-June '13	138	107	37	-25%	-50%
HH	July '07-May '12	59	110	74	-23%	-48%
MH2	July '07-June '12	60	160	104	+12%	-25%
KH	July '07-June '12	60	162	114	+13%	-24%
CSC	May '11-April '13	24	163	82	+ 14%	-23%

MH3	July '07-June '12	60	164	98	+15%	-23%
PS	July '11-May '13	23	172	107	+20%	-19%
MH1	July '07-June '12	60	175	101	+22%	-18%

Figure 1 represents the dormitories in order of highest performance (lowest consumption). LEED dormitory EH is the top performer followed by non-LEED dormitories WT and HH, while LEED dormitory PS performed slightly better than the poorest performer non-LEED dormitory MH1.

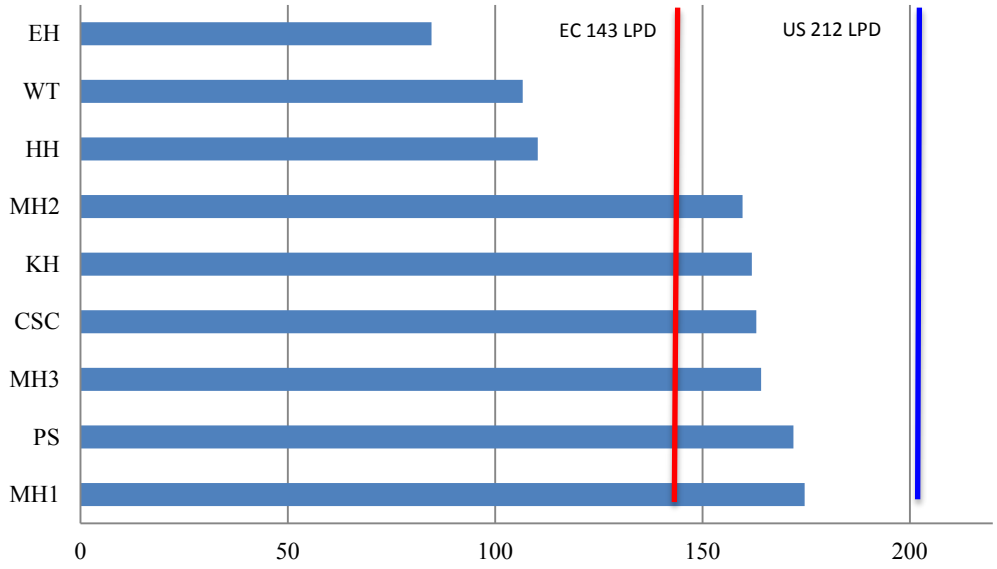


Figure 1. Actual water consumption of the nine dormitories in LPD (compared to engineer's metrics)

3.2 Non-LEED Dormitories

The average water consumption of all non-LEED dormitories was 146 LPD with a *stand. dev.* of 30 LPD. Figure 2 provides a profile of the water consumption of the six non-LEED dormitories for the years of data collected.

Examining building WT with over a decade of water consumption data, it averaged 107 LPD with a *stand. dev.* of 37 LPD with an increase in consumption over the twelve years of 3%. Comparing average consumption in WT (107 LPD) to the engineer's metrics (EC equals to 143 LPD, and AWWA to 212 LPD), the consumption was lower by 25% and 45%, respectively.

Excluding WT from the non-LEED dataset and examining MH1, MH2, MH3, HH and KH with five years of data, the average consumption increases to 154 LPD and a *stand. dev.* of 25 LPD. Comparing this average consumption (154 LPD) to the engineer's metrics, the consumption was higher by 8% and lower by 38%, respectively. In dormitories MH1, MH2, MH3, KH, and HH, the percent net change over the five years of analysis was 3% indicating an uptick.

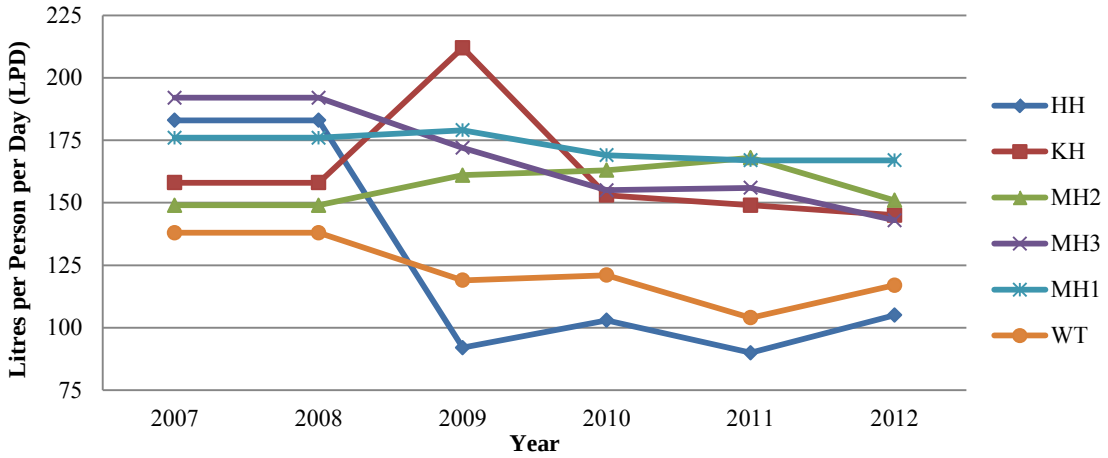


Figure 2. Actual average yearly water consumption of non-LEED dormitories

As shown in Figure 2, dormitories HH and KH showed the highest variation over the years versus steadier consumption in MH1, MH2, MH3, and WT. Several factors can vary consumption, including academic schedules of institutions, weather, water technologies, gender split, and occupant behaviour (Vickers 2001; Schleich and Hillenbrand 2009; Vinz 2009; Kats 2010; Elliot 2013), however, no single variable could be identified as the sole source of these variances. In an effort to dissect the purpose behind the variations, an exploration of the monthly consumption values is provided in Figure 3 showing the average monthly LPD of the six non-LEED dormitories in the six years for which data was collected. As can be seen in Figure 3, the months with the highest average consumption were during the Fall and Spring semesters for dormitories MH1, MH2, MH3, KH, and HH. The water consumption for the summer months (June, July, and August) were the lowest, followed by January winter recess. The highest consumption periods were attributed to periods of high occupancy (returning students) and warmer weather conditions for months within those semesters.

Dormitory WT also experienced consumption during the summer months (June-August) as it is operating year round due to academic requirements in the summer. Reversely, dormitories MH1, MH2, MH3, KH, and HH do not have summer sessions and showed minimal summer consumption.

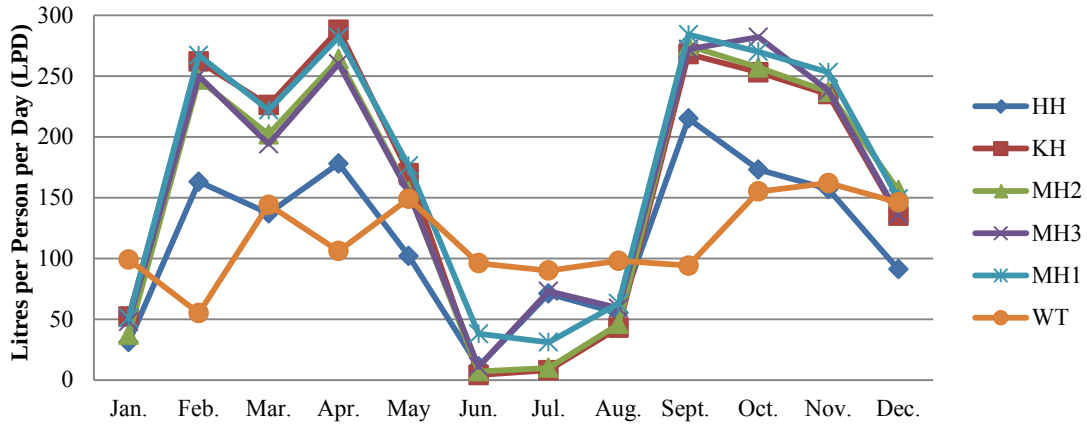


Figure 3. Actual average (from 2007 to 2012) monthly water consumption of non-LEED dormitories

3.3 LEED Dormitories

3.3.1 Dormitory EH

In calculating the LEED 'green' case, designers assume a specific number of days the dormitory shall be in operation. The assumed operational days plays an important part over the water performance calculation. Often designers assume different days, depending on the information provided by owner's facilities departments, which are typically tied to academic and use schedules. Designers of dormitory EH assumed 305 days with a LEED 'green' case consumption of 89 LPD. Using the 305-day assumption, dormitory EH resulted in the lowest average water consumption when compared to all the dormitories (LEED and non-LEED) of 85 LPD. Even though dormitory EH outperformed its counterparts in further dissecting the water consumption savings over the years, the savings were reduced by 38% per year on average. The average yearly consumption values from 2008 to 2012 were 133, 62, 68, 78, and 82 LPD, respectively. Potential reasons behind increased consumption in 2008 and 2012 may be attributed to first year commissioning and record heat in 2012, respectively. If consumption of the first and last years is excluded, the resultant average consumption is 69 LPD. This value is 29% lower than the 'green' case (89 LPD), however, the yearly savings compared to the LEED 'green' case diminish yearly in both scenarios (85 and 69 LPD). The water consumption trends upwards, reducing the percent savings as compared to the LEED 'green' case, making the building less sustainable every year. EH actual consumption was less than modeled consumption by an average of 22% over the three-year period (2009-2011), but only 4% over the five-year period (2008-2012).

To further explore the discrepancy between actual and LEED case consumption values, a user survey was distributed to EH occupants via the online tool SurveyMonkey. Since 44% of indoor residential water end use is related to shower and toilet use (AWWA 1999), questions were developed on the shared assumptions used in LEED (USGBC 2009) and AWWA (AWWA 1999) of shower duration (8 minutes), shower frequency (1/day/occupant), and toilet flushes (5 flushes/occupant/day). Figure 4 provides the percent breakdown of responses to the LEED and AWWA assumptions posed in the user survey. Sixty occupants answered the questionnaire in the two weeks following the survey distribution (November 2010), a value corresponding to 26% of students living in the dormitory at the time.

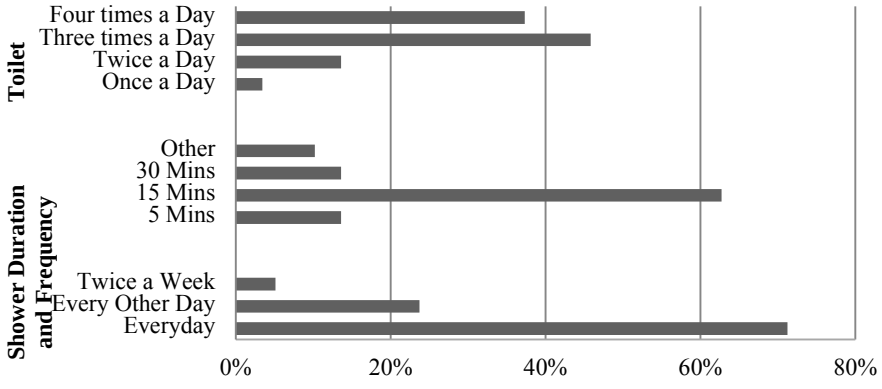


Figure 4. Occupant responses on toilet use, shower duration and frequency in dormitory EH

The responses indicated shower frequency and daily toilet flushes fall within the thresholds of the shared AWWA and LEED design assumptions. However, the shower duration assumptions of 8 minutes dramatically fell short. Over 87% of respondents indicated taking longer than 15-minute showers. Such variations in actual practice versus modeled assumptions can result in large differences in water estimations and performance evaluations. In another survey about occupant water conservation attitudes of a non-LEED residential building, it was found that 31% of occupants believed their power to minimize water consumption was minimal (Randolph and Troy 2008).

Such results highlight differences in occupants' attitudes and behaviours having substantial impacts on sustainable practices to lower water consumption (Barr 2003; Bamberg 2003; Hand et al. 2003; Hurlimann 2006; Alshuwaikhat and Abubakar 2008; Randolph and Troy 2008).

3.3.2 Dormitory CSC

CSC designers assumed 360 operational days with a LEED 'green' case of 88 LPD. CSC exceeded modeled consumption by an average of 85% over the three-year period (2011-2013). The yearly consumption values for 2011, 2012, and 2013 were 147, 170, and 172 LPD, respectively. This results in drastic percent differences in consumption as compared to the modeled case with 67% higher, 93% higher, and 95% higher consumption in 2011, 2012, and 2103, respectively. As previously mentioned, part of the increase may be due to record heat in 2012 throughout the US. However, the drastic percent increases in consumption over the years echoes the findings in other dormitories to behave less sustainably over time.

3.3.3 Dormitory PS

PS designers assumed 250 operational days with a LEED 'green' case of 87 LPD. The yearly consumption values for 2011, 2012, and 2013 were 198, 146, 171 LPD, respectively, resulting in differences in consumption as compared to the modeled case with 128% higher, 68% higher, and 97% higher consumption in 2011, 2012, and 2013, respectively. Dormitory PS actual consumption exceeded modeled consumption by an average of 98% over the three-year period. It must be noted given the dormitories location its occupants may have been better equipped to handle the heat of 2012, as consumption of PS in that year was lower than any other year.

3.4 Comparison of LEED and non-LEED Dormitories

Exploring the age and technologies employed among the dormitories, the average age of non-LEED dormitories is 46 years with a *stand. dev*. of 18 years, while the average age of LEED dormitories is 4 years with a *stand. dev*. of 1 year. Dormitories EH, WT, CSC, and PS were built in 2008, 2002, 2011, and 2011, respectively, when the 1992 Federal Energy Policy Act (FEPA) was already valid. This act includes maximum consumption for fixtures of 9.5 litres per minute (LPM) and 6.0 litres per flush (LPF). MH1, HH, MH2, KH, and MH3 were built in 1951, 1959, 1961, 1961, and 1966, respectively, and do not comply with the 1992 Federal Energy Policy Act.

All non-LEED dormitories and CSC used full flush toilets, while EH and PS used dual-flush toilets (low/full). Figure 5 represents the average and *stand. dev.* of flow fixture rates in LEED

and non-LEED dormitories in LPM for lavatory, kitchen sink, and shower fixtures, and in LPF for toilets.

On average, non-LEED dormitories used flow fixtures with 6.4, 7.9, and 7.9 LPM for shower, lavatory, and kitchen sink, respectively, with toilets using 10.9 LPF. On average, LEED dormitories used flow fixtures with 5.9, 1.9, and 8.1 LPM for shower, lavatory, and kitchen sink, respectively, with toilets using 3.6 and 5.7 LPF for low and full flush, respectively. Even though non-LEED flow fixtures were higher on average, the dormitories outperformed LEED ones in terms of total LPD, indicating that reliance on technology may not be the answer to lowering overall consumption. Attention must also be given to occupant expectations and behaviours. For example, some respondents in the EH survey commented about their frustrations with low flow fixtures and declared they had replaced low flow showerheads with higher flow fixtures, whilst others commented taking longer showers. Similar comments were provided for low flow toilets, where respondents indicated often double and triple flushing as the toilet low flush was simply not sufficient.

Examining the average water consumption of LEED dormitories between years, building EH, CSC, and PS consumed 10% more, 9% more, and 5% less, respectively, between yearly readings. However, compared to their LEED 'green' cases, the average yearly consumption of EH, CSC, and PS were 4% lower, 85% higher, and 98% higher, respectively. These values result in an overall percent increase in consumption of 60% as compared to their LEED 'green' cases. Dormitory EH and CSC are LEED-Gold while PS is LEED-Silver. Even though the LEED-Gold dormitory outperformed the LEED-Silver one, both did not provide the expected savings (Kats 2010). Moreover, LEED dormitory data indicates diminished consumption savings over time, rendering them less sustainable every year.

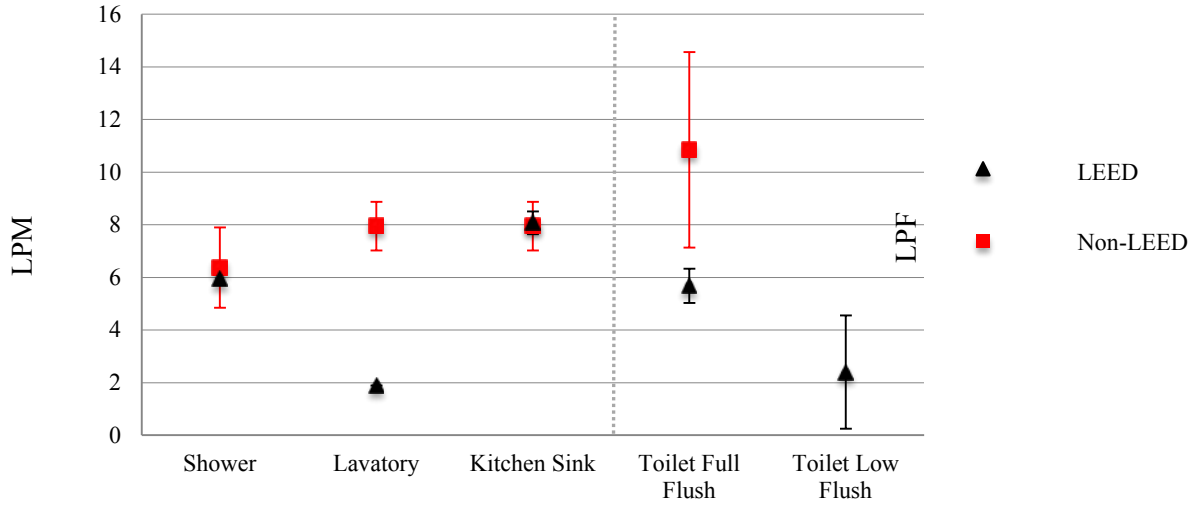


Figure 5. Average flow fixture rates in LPM and LPF for LEED and non-LEED dormitories

Non-LEED dormitories WT, MH1, MH2, MH3, KH, and HH resulted in a percent increase of 3% in water consumption over the years. Based on the findings, on average non-LEED dormitories outperformed LEED ones depicting steadier consumption profiles over the years. It is interesting to note as the gender split equalized in dormitories the consumption increased. Dormitories EH

and WT, the highest performing, had the highest male populations at 75% on average while dormitories MH1, MH2, MH3, KH, HH, PS, and CSC had average male populations of 47%.

4. CONCLUSIONS

Isolating water consumption of dormitories using USDOE, USGS, AWWA, and EC data is problematic due to the differences in the categorization of dormitories between water studies and a lack of available data. Varied classifications of residential customers by utility companies (Vickers 2001; Randolph and Troy 2008) also compound the problems in collecting published data on water consumption in dormitories. As noted previously, many factors impact water use including: geography, climate/weather conditions, socioeconomic factors, gender, occupant behaviors, and modeling assumptions. Based on actual consumption data collected from nine dormitories, indoor water use falls in the range from 85 to 175 LPD. Overall average actual dormitory consumption was lower than values found in USDOE (375 LPD), EPA (265 LPD), EC (168 LPD), AWWA (212), and EC (143 LPD) engineer's metrics.

On average, non-LEED dormitories consumed 4% more than LEED ones, however, the LEED *stand. dev.* was 40% higher than non-LEED dormitories.

On a yearly and monthly basis, non-LEED dormitories depicted steadier consumption values with an overall 3% uptick for the entire time for which data was collected. On the other hand, LEED dormitories showed an increase of 5% over the years, and on average had higher variations in consumption patterns. The average water consumption of EH, CSC, and PS was 60% higher when compared to LEED 'green' cases. The data showed yearly decreases in savings, rendering LEED dormitories less sustainable every year. These results highlight the possibility that the LEED label does not fully capture actual user behaviour and it may result in unrealistic savings expectations.

Examining assumptions of LEED and AWWA, over 87% of respondents indicated longer than 15-minute showers. Such vast differences in assumptions (eight minutes) and actual practice (over 15 minutes) must be ameliorated to ensure performance gaps are minimized.

It is interesting to note as the gender differential equalized the consumption in the dormitories increased, tying to arguments made by researchers on the inequality of gender consumption (Vinz 2009; Elliot 2013). The best performing dormitories had on average 75% males, while the poorer performing dormitories held an average of 47% males.

Finally, it is important to highlight that technology alone is not the answer to conservation. Larger reductions in water consumption can be gained through improved users' attitudes and changes in occupants' behaviours.

5. REFERENCES

Alshuwaikhat HM, Abubakar I. 2008. An integrated approach to achieving campus sustainability: Assessment of the current campus environmental management practices. Journal of Cleaner Production, 16(16):1777-1785.

American Water Works Association (AWWA). 1999. Residential End Uses of Water. Denver (CO): AWWA Research Foundation.

Balling RC, Gober P, Jones N. 2008. Sensitivity of residential water consumption to variations in climate: An intraurban analysis of Phoenix, Arizona. Water Resources Research, 44(10):1-11.

Bamberg S. 2003. How does environmental concern influence specific environmentally related behaviours? A new answer to an old question. Journal of Environmental Psychology, 23(1):21-32.

Barr S. 2003. Strategies for sustainability: Citizens and responsible environmental behaviour. Area, 35(3):227-240.

Elliott R. 2013. The taste for green: The possibilities and dynamics of status differentiation through "green" consumption. Poetics, 41(3):294-322.

Environmental Protection Agency (EPA). 2013. WaterSense. Washington (DC): Office of Wastewater Management.

Hansen J, Sato M, and Reudy R. 2013. Reply to Rhines and Huybers: Changes in the frequency of extreme summer heat. Proc. Natl. Acad. Sci., 110:E547-E548

Hand M, Southerton D, Shove E. 2003. Explaining daily showering: A discussion of policy and practice. Economic and Social Science Research Council, Sustainable Technologies Programme.

International Association of Plumbing and Mechanical Officials (IAPMO). 2009. Uniform Plumbing Code.

Hurlimann A. 2006. Water, water, everywhere - Which drop should be drunk? Urban Policy and Research, 24(3):303-305.

International Code Council (ICC). 2009. International Plumbing Code (IPC). Country Clubs Hills (IL).

Kats G. 2010. Greening our built world: Costs, benefits, and strategies. Washington (DC): Island Press.

Mudgal S, Lauranson R. 2009. Study on Water Performance of Buildings. European Commission, DG Environment. Reference Number: 070307/2008/520703/ETU/D2.

National Weather Service (NWS). 2013. Indices and Forecasts Daily Arctic Oscillation Index. National Oceanic and Atmospheric Administration (NOAA).

Randolph B, Troy P. 2008. Attitudes to conservation and water consumption. Environmental Science and Policy, 11(5):441-455.

Schleich J, Hillenbrand T. 2009. Determinants of residential water demand in Germany. Ecological Economics, 68(6):1756-1769.

Southern Illinois University (SIU). 2002. Predictive Models of Water Use: An Analytical Bibliography. Carbondale (IL).

Vickers A. 2001. Handbook of Water Use and Conservation: Homes, Landscapes, Businesses, Industries, Farms. Amherst (MA): WaterPlow Press.

Vinz D. 2009. Gender and sustainable consumption: A German environmental perspective. European Journal of Women's Studies, 16:59-179.

United States Department of Energy (US-DOE). 2013a. Chapter 8: Water. In: Buildings Energy Data Book (BEDB).

United States Department of Energy (US-DOE). 2013b. Building Technologies Office, Residential Buildings Integration.

United States Geological Survey (USGS). 1995. Estimated Climate Zone Designations in: Water Use in the United States 1995.

United States Geological Survey (USGS). 2000. Estimated Climate Zone Designations in: Water Use in the United States 2000.

United States Geological Survey (USGS). 2005. Estimated Climate Zone Designations in: Water Use in the United States 2005.

United States Green Building Council (USGBC). 2009. LEED New Construction Design Guide, Volume 3.

PHOTOVOLTAIC SOLAR COOLING IN REGIONS WITH HIGH SOLAR IRRADIATION: A CASE STUDY

West Coast Conference, March 2015

Muhammed Emin Tolu[1§], Mehmet Numan Kaya[2] , Seyit Alperen Çeltek[3]
[1]Karamanoglu Mehmetbey University, Department of Mechanical Engineering, 70100, Karaman, Turkey, metolu@kmu.edu.tr; [2]Karamanoglu Mehmetbey University, Department of Mechanical Engineering, 70100, Karaman, Turkey, mnumankaya@kmu.edu.tr; [3]Karamanoglu Mehmetbey University, Department of Energy Systems Engineering, 70100, Karaman, Turkey, salperenceltek@kmu.edu.tr

ABSTRACT

Turkey is one of the luckiest countries in the world, receiving high levels of solar radiation. This encourages individuals to use solar power for various applications, including space cooling, since the demand for cooling of indoor air is growing due to increasing comfort expectations. Today, cooling systems dominate the energy consumption in most office buildings and solar power can be used to cover this demand. In this study, a photovoltaic solar powered cooling system is designed to meet the cooling requirement of the office rooms of Karamanoglu Mehmetbey University located in Karaman, Turkey. By using the calculated cooling load, the system components are selected and the economic feasibility of the system is evaluated under the climatic conditions of Karaman. Long-term solar irradiation measurements of the Turkish State Meteorological Service are used for calculations in this study.

Keywords: solar cooling, photovoltaic, solar irradiation, cooling load

1. INTRODUCTION

There is no doubt that solar energy is an essential requirement for continuity of the life cycle. It is an energy source that is anticipated to be a solution for the endless energy need of humanity. By means of the various methods that have been developed by humanity since humankind has learned how to use equipment and instruments, solar energy has been used in the areas such as heating, cooking, food drying, and supplying hot water to ease human life. After the beginning of electricity production by using solar radiation, there has been a rapid increase in the number of areas that benefit from solar energy. Nowadays, ensuring the comfort conditions is becoming more important and for this reason, heating, cooling, and air conditioning systems lead to excessive electricity consumption. Due to the difficulty and ineffectiveness of the cooling processes, cooling systems consume more electricity compared to heating and air conditioning systems.

Solar powered cooling systems have been analyzed for several years by researchers. Balghouthi et al. (2008) designed a specific solar-powered absorption cooling technology for Tunisian conditions using simulator program. Their cooling system for a typical building of 150 m2 is comprised of a water lithium bromide absorption chiller with a capacity of 11 kW, a 30 m2 flat

[§]Corresponding author: Muhammed Emin Tolu, Karamanoglu Mehmetbey University, Department of Mechanical Engineering, 70100, Karaman, Turkey; Tel:+90 338 226 20 00-50 67, Fax: +90 338 226 20 23; metolu@kmu.edu.tr

plate solar collector area tilted 35° from the horizontal, and a 0.8 m3 hot water storage tank. Their results show that absorption solar air-conditioning systems are suitable under Tunisian conditions. Mateus and Oliveira (2009) have analyzed the integrated solar cooling system in different building types and climates. The authors considered three different locations and climates: Berlin (Germany), Lisbon (Portugal), and Rome (Italy). Their results point out that in three cities, it is possible to save in total energy costs and CO_2 emissions by using the solar system. Despite the fact that the exploitation cost of a solar air-conditioning system is considerably lower when compared to a conventional system, the total cost (including investment, service charge, and maintenance expenses) is high, even when extending the operation period as much as possible over the course of many years. They also emphasized that it is difficult to compete with present energy sources like electricity and gas for solar cooling. Thus, it is necessary that initial costs for solar collectors are further reduced. Guo and Shen (2009) proposed a lumped method combined with a dynamic model for use in investigating the performance and solar fraction of a solar-driven ejector refrigeration system for office air conditioning applications for buildings in Shanghai, China. They concluded that compared with a traditional compressor based air conditioner, the system can save up to 80% electric energy while providing the same cooling capacity for office buildings. Hence, the system offers a good energy conservation method for office buildings. Jaunzems and Veidenbergs (2010) studied a small scale solar cooling system in the climate conditions of Latvia and presented the results of the environmental and economic analyses. It has been concluded that despite the fact that solar cooling systems have significant potential to reduce CO2 emissions due to a reduction of electricity consumption, the economic feasibility and attractiveness of solar cooling systems is still low. Eicker et al. (2014) studied the energy analysis and economic evaluation of solar thermal and photovoltaic cooling systems used for the air conditioning in office buildings in Palermo, Madrid, and Stuttgart. The results show that the photovoltaic cooling system is favorable in comparison with the thermal solar cooling system. An exhaustive sensitivity analysis shows a strong influence of initial investment costs on the payback and cost of saved primary energy. Fong et al. (2015) investigated the technical effectiveness of solar cooling and heating for the typical low-rise residential building in the subtropical climate of Hong Kong. The results indicate that the solar cooling and heating is technically feasible for the typical low-rise residential building in Hong Kong. They have also highlighted that the solar energy is necessary, but not enough, so the approach of solar cooling and heating can be extended to renewable cooling and heating like bioenergy. Wider application of renewable energy is needed, and within this framework renewable cooling and heating is an effective strategy in response to the growing energy needs. Allouhi et al. (2015) presented a review of the suitable systems to provide cooling from solar energy for both thermal and photovoltaic ways. The results show that all solar cooling systems have a great potential for environmental and energy advantages such as energy saving and reduction of CO_2 emissions. In this study, a solar powered cooling system that will be used for cooling the office rooms of the Karamanoğlu Mehmetbey University was designed using a calculated cooling load and was also financially analyzed.

2. MATERIAL AND METHODS

Air conditioning systems are often used for heating and cooling of living spaces, especially in arid regions with a hot climate. Compared to other systems that are used for heating and cooling,

air conditioning systems result in more costs based on electricity consumption. Therefore, reducing the consumption of electricity is an essential field of study. Various methods such as changing technical properties of air conditioners, or using auxiliary systems that aid air conditioners to improve efficiency or decrease consumption of electricity are used to achieve the desired conditions. In this study, electricity demand of the designed system is supplied by a PV solar system. An appropriate PV solar system is selected by using the climate data of Karaman, Turkey.

Properties of the application area are as follows:

- Location: City of Karaman (33 E, 37 N)
- Area: 15 m^2 for each office room
- Height: 3 m from floor to ceiling
- Purpose: Office room for one person
- Insulation quality: Well-insulated
- Window size: 2 m^2
- Person: 1 per office

Karaman is located in a region with high solar irradiation, which makes individuals and companies want to invest in Karaman. In addition, the Ministry of Energy and Natural Resources has identified Karaman as one of the most efficient regions in Turkey for investments on solar energy. A solar irradiation map and total solar irradiation values were obtained from the General Directorate of Renewable Energy of Turkey and are shown in Figure 1.

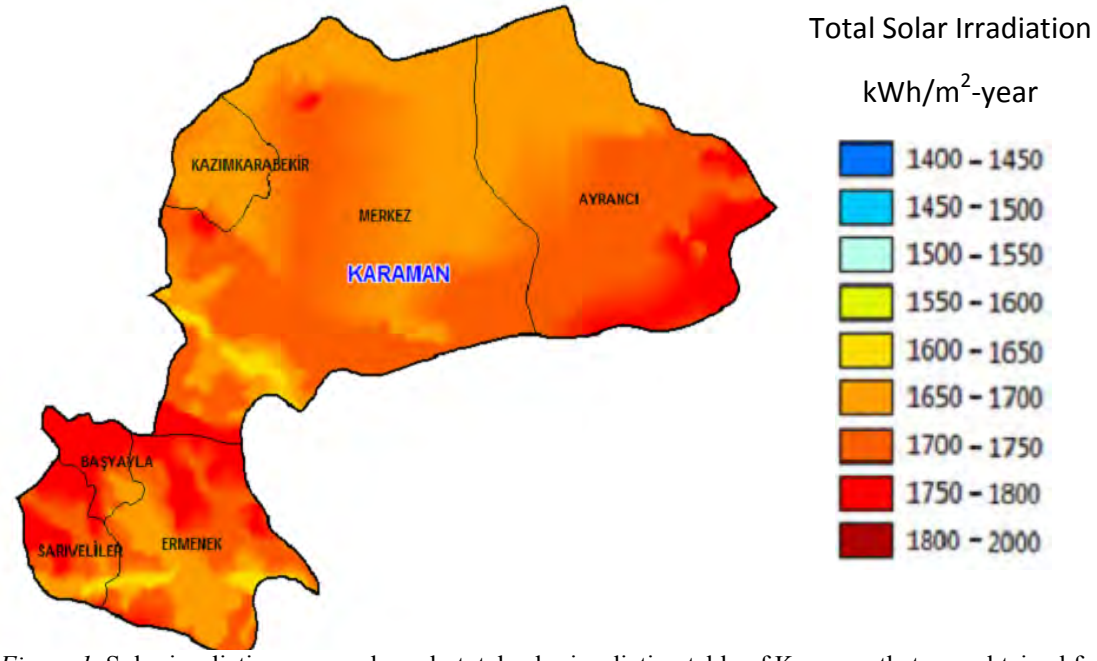


Figure 1. Solar irradiation map and yearly total solar irradiation table of Karaman that was obtained from the General Directorate of Renewable Energy.

It is important to calculate cooling load accurately, otherwise system selection for cooling or air conditioning will be improper and desired comfort conditions will not be achieved. There are

various factors that increase cooling load. These are generally referred to as “heat gain” and each reason for heat gain must be calculated properly. Heat gain from the window is calculated using Eq. 1 and Eq. 2.

$$Q_{1r} = A^* (A_R/A)\, q_G \quad (W) \qquad (1)$$

$$Q_{1k} = K^*A\, (T_o - T_i) \quad (W) \qquad (2)$$

Where Q_{1r} is the heat gain from solar radiation through the window, Q_{1k} is the heat gain from conduction and convection through the window, A is the area of the window, A_R / A is the radiation transmission rate of the window, q_G is the correction and shading factor, K is the total heat transfer coefficient of the window and T_o and T_i are outdoor and indoor temperatures respectively. Heat gain from the outer wall is calculated using Eq.3.

$$Q_2 = K^*A^*\Delta T_{eq} \quad (W) \qquad (3)$$

Where Q_2 is the heat gain from the outer wall, K is the total heat transfer coefficient of the wall, A is the area of the wall perpendicular to the heat flow, and ΔT_{eq} is the equivalent temperature difference. Heat gain from the floor and ceiling is calculated by Eq. 4.

$$Q_3 = K^*A\, (T_n - T_i) \quad (W) \qquad (4)$$

Where Q_3 is the heat gain from the floor and ceiling, K is the total heat transfer coefficient of the floor or ceiling, A is the area of floor or ceiling and T_n and T_i are the temperature of the neighboring space and the temperature of the indoors. Heat gain from people is calculated by Eq. 5.

$$Q_4 = Q_{4s} + Q_{4l} \quad (W) \qquad (5)$$

$$Q_{4s} = n^*\, Q_{4s,p} \quad (W) \qquad (6)$$

$$Q_{4l} = n^*\, Q_{4l,p} \quad (W) \qquad (7)$$

Where Q_4 is the heat gain from people, Q_{4s} and Q_{4l} are the sensible heat gain from people and latent heat gain from people respectively, n is the number of people, and $Q_{4s,p}$ and $Q_{4l,p}$ are the sensible heat gain per person and latent heat gain per person. Heat gain from illumination is calculated by Eq. 8.

$$Q_5 = k_1 * k_2 * Q_i \quad (W) \qquad (8)$$

Where Q_5 is the heat gain from illumination, k_1 and k_2 are the armature factor (1 for incandescent bulb; 1,2 for fluorescent bulb) and the utilization factor, respectively, and Q_i is the luminosity (power of illumination).

After the calculation of cooling load, an appropriate and adequate photovoltaic solar system must be designed by using the cooling load in calculations. The PVGIS (Photovoltaic Geographical Information System) database is used for the calculation of electricity production of the designed photovoltaic solar system. Daily and monthly electricity production of the system is obtained from the PVGIS database and by these data, the most efficient and optimal PV solar system is designed to generate the electricity that the air conditioning system needs.

3. RESULTS

As a result of the cooling load calculations using Equations 1 through 8, the cooling load of an office room is found to be 2,46 kWh, and it is found to be appropriate to select an air conditioner for each room that has a cooling capacity of 9000 BTU and a power consumption of 750 is given in Table 1. The entire system is designed for the cooling of 4 office rooms and will be used for 8 months (from March to October) per year, 4 hours in March and October, 6 hours in April and September, and 8 hours in May, June, July, and August per day. Within this context, daily, monthly, and yearly electricity consumption of the system can be calculated.

Table 1. Technical data of selected air conditioner

Cooling Capacity (Rated)	9000 Btu
Cooling Capacity (Min-Max)	4400-9500 Btu
Power Consumption (Cooling)	750 W

Daily electricity consumption of the system is calculated with Eq. 9.

$$C_d = P_a * t \qquad (9)$$

Where C_d is the daily electricity consumption of the system for one office room, P_a is the hourly power consumption of the system, and t is the daily usage.

By using daily electricity consumption data, yearly cost of electricity of the entire air conditioning system is found to be $702. This means designed system profits of $702 a year as presented in Table 2. In addition, in the months that the air conditioning system is idle (January, February, November, and December), generated electricity can be transferred to the grid circuit of Karaman. In Turkey, $0.133 per kW of electricity generated is paid by the Ministry of Energy and Natural Resources. Therefore, an additional profit of nearly $335.60 a year can be gained and in total, the PV solar system will generate a profit of $1037.60 a year as presented in Table 2.

Table 2. Yearly income and outcome

	5 kW PV SOLAR SYSTEM	ELECTRICITY TO OPERATE THE AC	ELECTRICITY TRANSFERRED TO GRID	PROFIT FROM ELECTRICITY TRANSFERRED TO GRID		COST OF ELECTRICITY USED BY AIR CONDITIONING (COVERED BY THE PV SOLAR SYSTEM)	
Month	*Generated Electricity (kW)*	*Daily Consumption Cd (kW)*	*Unused Electricity (kW)*	*Daily Profit ($)*	*Monthly Profit ($)*	*Daily Cost ($)*	*Monthly Cost ($)*
January	11,6	0	11,6	1,5428	46,284	0	0
February	14,8	0	14,8	1,9684	59,052	0	0
March	19,9	12	7,9	1,0507	31,521	1,8	54
April	21,4	18	3,4	0,4522	13,566	2,7	81
May	23,4	24	-0,6	-0,0798	-2,394	3,6	108
June	25,2	24	1,2	0,1596	4,788	3,6	108
July	26,1	24	2,1	0,2793	8,379	3,6	108
August	26,2	24	2,2	0,2926	8,778	3,6	108
September	24,4	18	6,4	0,8512	25,536	2,7	81
October	19,6	12	7,6	1,0108	30,324	1,8	54
November	16	0	16	2,128	63,84	0	0
December	11,5	0	11,5	1,5295	45,885	0	0
				YEARLY PROFIT ($)	335,559	YEARLY COST ($)	702
				TOTAL YEARLY PROFIT ($)		1037,559	

According to a small market survey in Turkey, installing a PV solar system that has 1kW of capacity costs $2500. To operate the air conditioning system as desired, a 5 kW capacity PV solar system must be installed. The cost of a system of this capacity is $12500 in Turkey. After the calculations, the basic payback period of the system is found to be nearly 12 years as presented in Table 3.

Table 3. Table containing investment cost of the PV solar system and basic payback period of the investment

Investment Cost ($)	12500
Total Yearly Profit ($)	1037.559
Basic Payback Period (Years)	12.05

In addition, monthly profit gained by transferring electricity to grid, and monthly profit generated by the PV solar system, are shown in Figure 2.

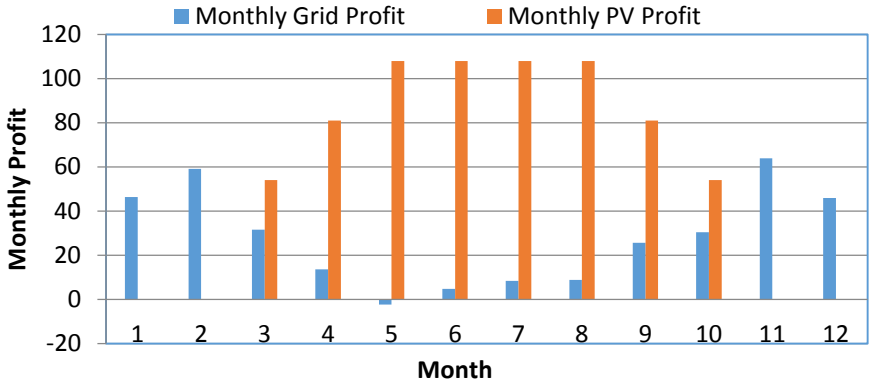


Figure 2. Graphical representation of monthly profits

4. CONCLUSION

Karaman has a hot and arid climate especially in the summer months. An air conditioning system was designed for cooling 4 office rooms at Karamanoğlu Mehmetbey University. Investment cost of the designed air conditioning system is not included in the calculations since the aim of the study is to compare a PV powered cooling system with air conditioners and a conventional cooling system with air conditioners. Selected air conditioners are completely the same. Daily, monthly, and yearly costs and profits are calculated and the entire system is analyzed economically and feasibly. The basic payback period of the designed PV solar powered cooling system is calculated as nearly 12 years. Although the region has great solar power potential, the basic payback period is found to be very high and such a basic payback period is not very economically feasible. The reason for this is that the investment costs of small PV systems are still too high.

5. REFERENCES

Allouhi A, Kousksou T, Jamil A, Bruel P, Mourad Y, Zeraouli Y. 2015. Solar driven cooling systems: An updated review. Renewable and Sustainable Energy Reviews, 44:159-181.

Balghouthi M, Chahbani MH, Guizani A. 2008. Feasibility of solar absorption air conditioning in Tunisia. Building and Environment, 43:1459-1470.

Eicker U, Colmenar-Santos A, Teran L, Cotrado M, Borge-Diez D. 2014. Economic evaluation of solar thermal and photovoltaic cooling systems through simulation in different climatic conditions: An analysis in three different cities in Europe. Energy and Buildings, 70:207-223.

Fong KF, Lee CK. 2015. Investigation of separate or integrated provision of solar cooling and heating for use in typical low-rise residential building in subtropical Hong Kong. Renewable Energy, 75:847-855

Guo J, Shen HG. 2009. Modeling solar-driven ejector refrigeration system offering air conditioning for office buildings. Energy Build, 41(2):175-181.

Jaunzems D, Veidenbergs I. 2010. Small scale solar cooling unit in climate conditions of Latvia: Environmental and economical aspects. Environ Clim Technol, 4:47-52.

Mateus T, Oliveira AC. 2009. Energy and economic analysis of an integrated solar absorption cooling and heating system in different building types and climates. Applied Energy, 86:949-957.

Photovoltaic Geographical Information System (PVGIS). 2012 February 10. [Internet]. European Commission Joint Research Centre, Ispra, Italy. Available from: http://re.jrc.ec.europa.eu/pvgis/

RENEWABLE ENERGY POWERED RURAL IRRIGATION: FEASIBILITY COMPARISON OF SOLAR AND WIND BASED WATER PUMPING SYSTEMS IN TURKEY

West Coast Conference, March 2015

Mehmet Numan Kaya[1§], Faruk Köse[2]
[1]Karamanoğlu Mehmetbey University, Department of Mechanical Engineering, Yunus Emre Campus, Karaman, Turkey, mnumankaya@kmu.edu.tr; [2]Selcuk University, Department of Mechanical Engineering, Aladdin Keykubat Campus, Konya, Turkey, fkose@selcuk.edu.tr

ABSTRACT

Renewable energy sources are being used in a wide range of applications including water pumping for rural irrigation all over the world. Turkey is one of the developing countries supporting investments in renewable energy technologies with various incentive programs. In this study, the feasibilities of wind and solar power systems to cover the energy need of irrigation pumps for rural irrigation are compared for a small town, Alibeyhuyugu, which is located in Konya, Turkey. According to real energy consumption data, investment costs of both systems are calculated and an economic analysis was performed. In addition, reduction in CO_2 emissions afforded by the use of the renewable energy systems are calculated and environmental effects are discussed.

Keywords: Solar, wind, water pumping, irrigation, feasibility

1. INTRODUCTION

Our planet is getting dirtier and dirtier each passing day and people are now anxious about future generations. One of the main reasons of continuous damage is power production from fossil fuels. In the last decade, people have finally realized how horrible it can be for the humanity if some precautions are not taken. Since then, renewable energy sources that are clean, unlimited, and environmentally friendly have become more popular, especially for researchers. Today, renewable power is being used in a wide range of applications including power supply for irrigation. Using renewables for irrigation fits one of the futuristic goals of Turkey - sourcing one-third of its power needs from renewable energy by 2023 (EPDK 2012).

Plentiful studies have been conducted about suitability, feasibility, and optimization of wind and solar energy for water pumping. Cloutier and Rowley (2011) performed a case study on the feasibility of renewable energy sources for pumping clean water for Central Nigeria. They concluded that if the water demand at a particular site exceeds the capabilities of a hand pump, a renewable energy-powered pumping system is an attractive option, both economically and logistically in comparison to fossil-fuel-powered alternatives. Purohit (2007) evaluated the use of renewables for irrigation water pumping in India. It has been concluded that the unit cost is

[§]Corresponding author: Mehmet Numan Kaya, Karamanoğlu Mehmetbey University, Department of Mechanical Engineering, Yunus Emre Campus, Karaman, Turkey; Tel: +90 338 226 20 00-50 67, Fax: +90 338 226 20 23; E-mail: mnumankaya@kmu.edu.tr

substantially higher for PV pumps as compared to the corresponding values for diesel or electric pumps. Purohit has also stated that it is very important to match the design characteristics of the rotor of the windmill pump with the wind resource available at the end use location, as the unit cost of water delivered for a windmill pump would critically depend upon the prevailing wind resource at the site as well as the design characteristics. The feasibility of wind-powered water pumping systems for irrigation applications in India was studied by Parikh and Bhattacharya (1984), where the authors reported that wind energy based water pumping systems are best suited for irrigation applications for Indian meteorological conditions. Rehman and Sahin (2012) made an attempt to utilize the power of wind for pumping the water for remotely located inhabitants not connected with the national power grid. It has been reported that an annual total water pumping capacity of 30,000 m^3 is possible from a depth of a total dynamic head of 50 m by using wind turbines 2.5 kW of capacity. Meah et al. (2008) stated in his study regarding the opportunities and challenges of solar water pumping that solar water pumping systems could be viable by using local resources such as skills, materials, and finances that are economically viable in developing countries and competitive with the conventional diesel generator water pumping systems. Rehman and Sahin (2014) compared solar PV and diesel power systems for water pumping in Saudi Arabia and stated that solar PV power generating system is comparable in the unit cost of energy with the diesel system, even though the unit price of the diesel fuel is very low. Kelley et al. (2010) examined the feasibility of solar powered irrigation and found out that PVP irrigation is technically and economically feasible. In a similar study Mekhilef et al. (2012) stated that photovoltaic systems and/or solar thermal systems would be the suitable options for agricultural application, and especially for the distant rural area. This study investigates the feasibilities of wind and solar power systems to cover the energy need of irrigation pumps for rural irrigation of a small town, Alibeyhuyugu, Cumra, considering the conditions for Turkey.

2. DESCRIPTION OF THE REGION AND POWER DEMAND FOR IRRIGATION

Alibeyhuyugu is a small town located in Cumra, Konya in the Central Anatolian Region of Turkey. Location of the region is presented in Fig. 1. The region's economy relies mostly on agriculture and stockbreeding. Irrigation is supplied by a private company through submerged irrigation pumps that have input powers changing between 45 and 110 kW. Annual mean energy consumption of irrigation pumps is about 6000 MWh. This value will be used to select the capacities of wind and solar power plants. Pumps are in operation only seven months of the year. This period is called the irrigation period that begins in April and ends in October. Monthly mean energy consumption of irrigation pumps is shown in Fig. 2.

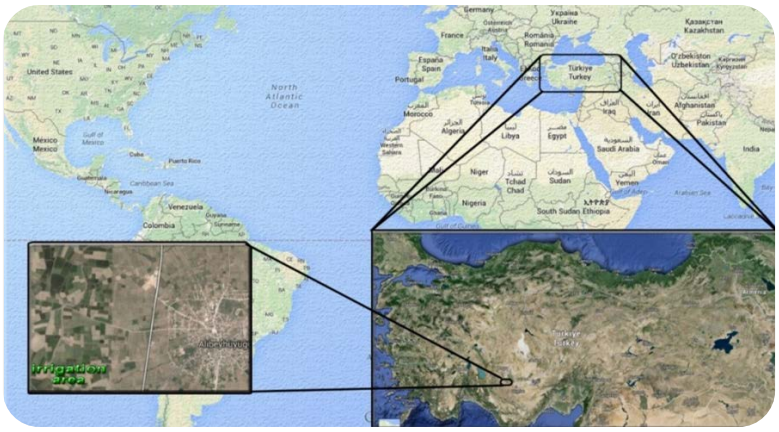


Figure 1. Location of the region (Google Maps, 2015)

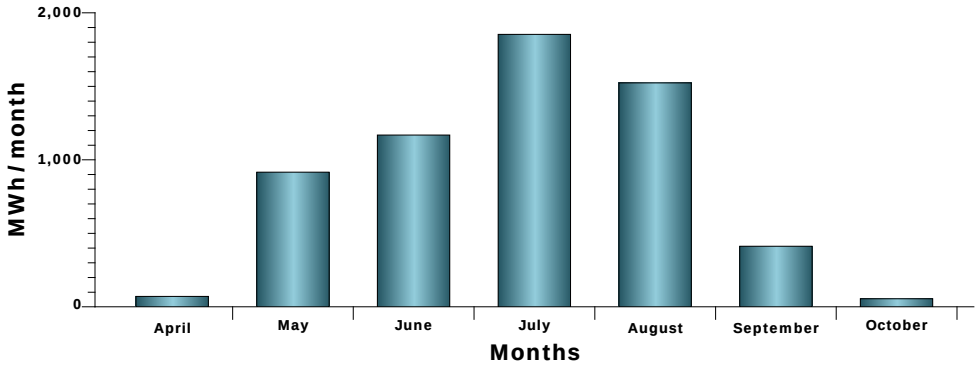


Figure 2. Monthly mean energy consumption of irrigation pumps

3. SOLAR AND WIND CHARACTERISTICS

Solar radiation data for Cumra, Turkey are obtained from the Turkish Meteorological Center and presented in Fig. 3 (EIE 2015). Daily mean solar radiation and sunshine durations are 4.5 kWh/m^2 and 8.17 hours, respectively.

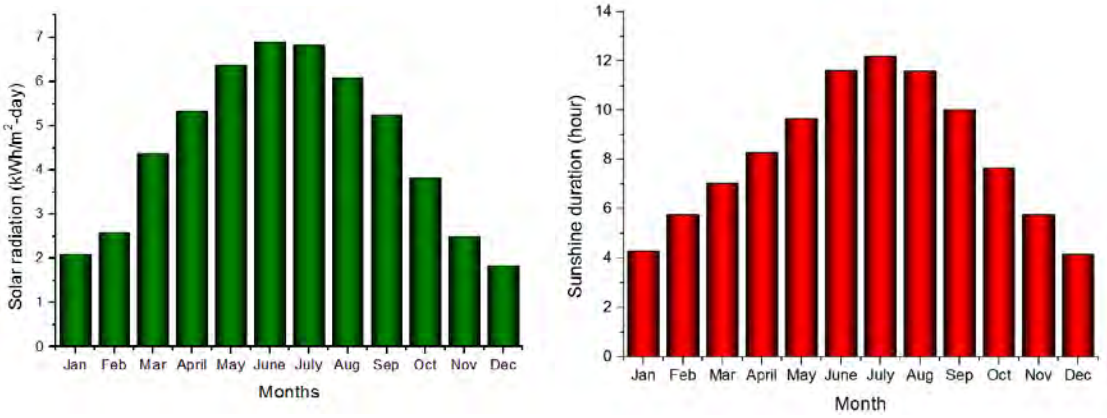


Figure 3. Solar radiation and sunshine duration in the region

Wind climate data has been measured for several years with the help of a wind pole in the region. Monthly mean wind speed values are shown in Fig. 4. The mean wind speed is 5.12 m/s at 35 m height.

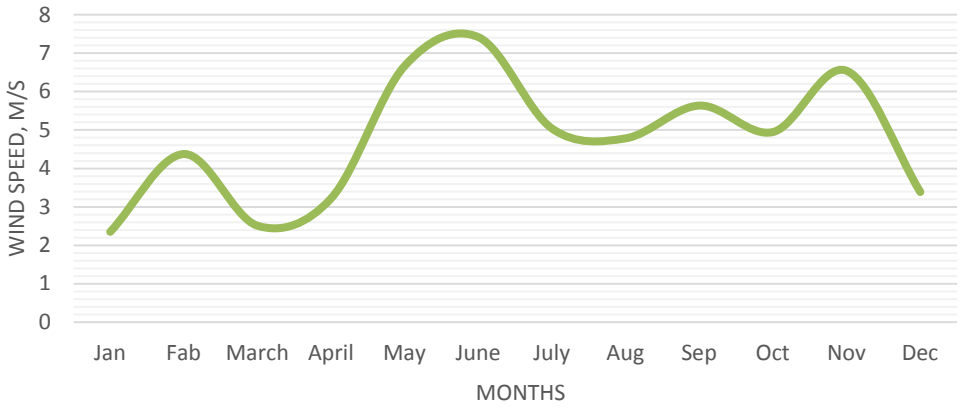


Figure 4. Mean wind speeds in the region

4. SOLAR AND WIND POWERED IRRIGATION SYSTEMS

4.1. Solar based system

The solar powered irrigation system consists of solar photovoltaic panels, power regulators and controllers, and deep well pumps, and all these are connected to the national grid in case the generated electricity is not enough to cover the demand. Schematics of the solar powered irrigation system is presented in Fig. 5. To calculate the power output of the solar power plant, Photovoltaic Geographical Information System (PVGIS 2015) is used.

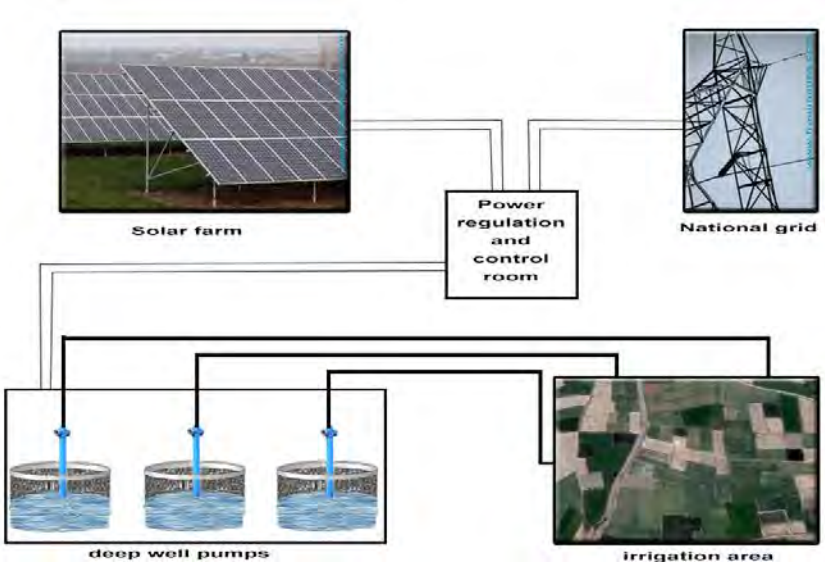


Figure 5. Schematics of solar powered irrigations system

The investment cost of a photovoltaic solar power plant is given as 3400 USD/kW by Batman et al. (2012) in 2012 for Turkey. This value includes the PV module price, grid tie inverter price, and all other costs including wiring and settings. According to the report of Sunshot (2014), commercial PV system prices declined 6%–7% per year, on average, from 1998– 2013, and by 12%–15% from 2012–2013. It is assumed that the PV system prices declined 10% each year from 2012 to 2015, therefore unit price of the PV system is considered to be 2480 USD/kW for Turkey in 2015. In addition, operation and maintenance cost of the PV system is assumed to be annually 0.12% of the system cost as reported by Moore and Post (2008).

4.2. Wind based system

Wind based systems have the same components as the solar powered system, except photovoltaic panels are replaced with wind turbines. In the present work, four commercially available wind turbines are selected to be studied. The specifications of the selected wind turbines are given in Table 1.

Table 1. Characteristics of the selected turbines

Wind turbines	Cut-in wind speed, V_c (m/s)	Cut-out wind speed, V_F (m/s)	Rated output Power, P_{eR} (kW)	Selected Hub Height (m)	Rotor diameter(m)
WT - 1	3	25	1500	100	82
WT - 2	3	22	1500	100	87
WT - 3	3	25	1800	100	100
WT - 4	3	25	3000	100	101

The initial investment cost of the WPS includes the wind turbine cost and all other initial costs, e.g., the cost of transportation, installation, civil work and connections and it can be calculated using Eq. 1.

$$C_{wt} = C_{spe} \cdot P_r \quad (1)$$

Where C_{spe} is the specific cost and P_r is the rated power of the wind turbine. The specific cost of wind turbines varies according to the rated power and the manufacturer of the wind turbine, and it is chosen using a band interval as given in Table 2. In the present study, it is assumed to be 1400 USD/kW. Other initial costs are assumed to be 30% of the wind turbine cost for the WPS. Operation and maintenance costs for the wind power station is assumed to be a fraction of the facility cost. In this paper, it is assumed to be 20% of the annual cost of the WPS (facility cost/lifetime).

Table 2. Cost of wind turbines based on the rated power (Gokcek and Genc 2009; Sathyajith 2006)

Wind turbine size (kW)	Specific cost ($/kW)
10-20	2200-2900
20-200	1500-2300
> 200	700-1600

5. METHODOLOGY FOR ECONOMIC ANALYSIS

In the present study, three evaluation methods, which are Basic Payback Period (BPB), Net Present Value (NPV), and Internal Rate of Return (IRR), are used for the economic feasibility analysis of the project. BPB is the value in years that shows the amount of minimum time to recover the total investment and it is calculated with Eq. (2) (Ozerdem et al. 2006).

$$BPB=(C/AS) \quad (2)$$

Where C is the total capital cost and *AS* is the net annual saving. In this paper, annual saving is assumed to be the annual amount paid each year for water pumping by the ABH irrigation cooperation, assuming that the installed PV or wind power system will cover all of the electricity demand by using a yearly auto-producer license.

NPV is calculated by discounting all future income and expenditure flows to the present with Eq. (3) (Ozerdem et al. 2006).

$$NPV=\sum[(B-C)/(1+r)^n] \quad (3)$$

Where *B* is the benefit, *C* is the cost, *r* is the discount rate, and *n* is lifecycle year of the project. In this study, the project lifespan was taken as 25 years for the analysis as suggested by many turbine manufacturer companies and the overall annual interest rate (*r*) is assumed to be 5%. Salvage cost was not taken into account which was estimated to be equal to the disassembly cost of the wind power system components at the end of the project lifespan.

IRR is the rate which would make the NPV value zero and it can be calculated with Eq. (4), where the parameters are same as the ones of NPV (Ozerdem et al. 2006).

$$\sum[B/(1+r)^n]=\sum[C/(1+r)^n] \quad (4)$$

6. RESULTS

Solar power generation values according to the installment capacity are calculated using the Photovoltaic Geographical Information System (PVGIS), which is provided by the Institute of Energy and Transport of European Commission and results are given in Table 3.

Table 3. Electricity generation from various solar farm capacities in the region

System capacity (kWh)	Daily average(kWh)	Monthly average(kWh)	Total for year(kWh)
600	2540	77000	925000
1200	5070	154000	1850000
1800	7610	231000	2780000
2400	10100	308000	3700000
3000	12700	386000	4630000
3600	15200	463000	5550000
3800	16100	488000	5860000
4000	16900	514000	6170000
4200	17700	540000	6480000

As mentioned before, the deep well pumps used in the region have a consumption amount of 6000 MW/year. This value will be used to select the capacity of the solar and wind power plants. It will be assumed that the facility will have a yearly auto-producer license, which means that excess and deficient productions will be calculated at the end of the year and either the government will pay the user if there is excess production or the user will pay the government if there is deficient production. PV solar plant capacity is selected to be 4 MW since it covers the annual demand.

To calculate annual power outputs of the wind turbines, WaSP wind flow modelling software is used. The output power of each wind turbine is presented in Table 4. Two of WT – 2s are selected to be used since WT – 2 is the most efficient option in the region.

Table 4. Annual power outputs of each wind turbine

Sector no	angle	Wind speed	WT – 1	WT – 2	WT – 3	WT – 4
1	0	3,61	0,097	0,138	0,147	0,178
2	30	3,81	0,106	0,134	0,158	0,193
3	60	4,49	0,134	0,136	0,191	0,246
4	90	7,63	1,307	1,282	1,768	2,500
5	120	5,74	0,504	0,521	0,721	0,933
6	150	3,99	0,081	0,109	0,131	0,147
7	180	3,00	0,021	0,032	0,033	0,039
8	210	3,65	0,052	0,068	0,080	0,098
9	240	6,35	0,263	0,274	0,359	0,506
10	270	6,50	0,372	0,347	0,508	0,705
11	300	5,61	0,311	0,322	0,433	0,583
12	330	4,48	0,159	0,192	0,227	0,303
Mean wind sp./Annual prod.		5.48	3406	3,555	4,755	6429

Economical values for solar and wind based systems are presented in Table 5. Costs for pumps and other connections are excluded since they are already installed in the region.

Table 5. Results of economic analysis

Method	PV Solar system	Wind power system
Investment cost ($)	9,920,000	5,460,000
Annual O&M cost ($)	11904	43,680
Annual Savings ($)	807,000	807,000
Basic payback period (year)	12.47	7.18
Net present value ($)	1,286,039	5,298,190
Internal rate of return (%)	6.26	13.4

Nearly 70% of electricity is produced from fossil fuels, mainly from natural gas (70%) and coal (30%) in Turkey (TEIAS 2015). CO_2 emissions from burning coal and natural gas are around 820 and 490 $gCO2_{eq}$/kWh, respectively (IPC 2014). According to these values, annual CO_2 prevention is calculated as 2850 tons.

7. CONCLUSION

In this paper, the feasibilities of solar and wind power plants to cover the electrical energy demand of irrigation pumps for water pumping are compared for a small town located in Turkey. Although the solar radiation values are very high and mean wind speed is low in the region, wind power plants were found to be much more feasible compared to the PV solar power system to cover the energy consumption of water pumps. In addition, necessary solar power plant capacity was found to be about 30% more than wind power plant capacity to generate a similar amount of electricity. It was also determined that a significant amount of CO_2 emission will be prevented using either PV solar or wind power systems. In conclusion, wind power seems to be better option for water pumping for rural irrigation, and both options have a very positive effect on reduction of CO_2 emission.

8. ACKNOWLEDGEMENT

Authors would like to thank to ABH irrigation cooperation and Turkish Meteorological Center for providing data.

9. REFERENCES

Batman A, Bagriyanik FG, Aygen ZE, Gül O, Bagriyanik M. 2012. A feasibility study of grid-connected photovoltaic systems in Istanbul, Turkey. Renewable and Sustainable Energy Reviews, 16(8):5678-5686.

Cloutier M, Rowley P. 2011.The feasibility of renewable energy sources for pumping clean water in sub-Saharan Africa: A case study for Central Nigeria. Renewable Energy, 36:2220-2226.

Electric Power Resources Survey and Development Administration of Turkey (EIE). 2015. Solar Energy Atlas (GEPA). Ankara (Turkey).

Energy Market Regulatory Authority of Turkey (EPDK). 2012. Turkish Energy Market: An Investor's Guide. Ankara, Turkey.

Google Maps. 2015. Alibeyhuyuga, Cumra, Konya, Turkey [Internet]. Available at: maps.google.com.

Gokcek M, Genc MS. 2009. Evaluation of electricity generation and energy cost of wind energy conversion systems (WECSs) in Central Turkey. Applied Energy, 86:2731-2739.

Intergovernmental Panel on Climate Change (IPCC). Working Group III. 2014. Mitigation of Climate Change, Annex III: Technology - specific cost and performance parameters. Intergovernmental Panel on Climate Change. New York (NY): Cambridge University Press. p. 10.

Kelley LC, Gilbertson E, Sheikh A, Eppinger SD, Dubowsky S. 2010. On the feasibility of solar-powered irrigation. Renewable & Sustainable Energy Reviews, 14(9):2669-2682.

Mekhilef S, Faramarzi SZ, Saidur R, Salam Z. 2013. The application of solar technologies for sustainable development of agricultural sector. Renewable and Sustainable Energy Reviews, 18:583-594.

Meah K, Ula S, Barrett S. 2008. Solar photovoltaic water pumping—opportunities and challenges. Renewable and Sustainable Energy Reviews, 12:1162-75

Moore LM, Post HN. 2008. Five years of operating experience at a large, utility-scale photovoltaic generating plant. Prog Photovolt, 16:249-259.

Ozerdem B, Ozer S, Tosun M. 2006. Feasibility study of wind farms: A case study for Izmir, Turkey. Journal of Wind Engineering and Industrial Aerodynamics, 94(10):725-743.

Parikh MM, Bhattacharya AK. 1984. Wind data analysis for studying the feasibility of using windmills for irrigation. Energy in Agriculture, 3:129-136.

Photovoltaic Geographical Information System (PVGIS). 2015. [Internet] Available from: http://re.jrc.ec.europa.eu/pvgis/

Purohit P. 2007. Financial evaluation of renewable energy technologies for irrigation water pumping in India. Energy Policy, 35:3134-3144.

Rehman S, Sahin AZ. 2015. Performance comparison of diesel and solar photovoltaic power systems for water pumping in Saudi Arabia. International Journal of Green Energy, 12(7):702-713.

Rehman S, Sahin AZ. 2012. Wind power utilization for water pumping using small wind turbines in Saudi Arabia: a techno-economical review. Renewable and Sustainable Energy Reviews, 16:4470-4478.

Sathyajith M. 2006. Wind Energy Fundamentals, Resource Analysis and Economics. Berlin (Germany): Springer-Verlag.

Sunshot. 2014. Photovoltaic System Pricing Trends. Historical, Recent, and Near-Term Projections. U.S. Department of Energy [Internet]. Available from: http://www.nrel.gov/docs/fy14osti/62558.pdf.

Turkish Electricity Transmission Corporation (TEIAS). 2015. Electricity Production in Turkey. Ankara (Turkey).