

**BEFORE THE ILLINOIS POLLUTION CONTROL BOARD**

IN THE MATTER OF: )  
)  
PROPOSED AMENDMENTS TO CLEAN ) R-12-009  
CONSTRUCTION OR DEMOLITION ) (Rulemaking – Land)  
DEBRIS (CCDD) FILL OPERATIONS: )  
PROPOSED AMENDMENTS TO 35 ILL. )  
Adm. Code 1100 )

**NOTICE OF FILING**

TO: SEE ATTACHED SERVICE LIST

PLEASE TAKE NOTICE that I have electronically filed today with the Illinois Pollution Control Board, the Post-Hearing Comments of the Public Building Commission of Chicago, copies of which is herewith served upon you.

Dated: April 18, 2012

Respectfully submitted,

**PUBLIC BUILDING  
COMMISSION OF CHICAGO**

By: WD Ingersoll  
One of Its Attorneys

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**PROOF OF SERVICE**

I, William D. Ingersoll, certify that I have served the attached Post-Hearing Comments of the Public Building Commission of Chicago, by U.S. Mail, first class postage prepaid, on April 18, 2012 to the following:

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**POST-HEARING COMMENTS**

**OF THE PUBLIC BUILDING COMMISSION OF CHICAGO**

NOW COMES the PUBLIC BUILDING COMMISSION OF CHICAGO (“PBC”), by and through its counsel Brown, Hay & Stephens, LLP, and presents the following Post-Hearing Brief for consideration by the Illinois Pollution Control Board (“Board”).

**I. LEGISLATIVE CONSIDERATIONS**

As the record establishes, PBC was thoroughly engaged in the three-year legislative effort that led to the legislation which underlies this rulemaking. PBC’s key interest in that legislation (and this rulemaking) is a workable, sensible definition of “uncontaminated soil” (i.e., clean dirt) so that public contractors and public contracting entities would be able to readily ascertain what soil is appropriate for transfer to a permitted clean construction and demolition debris (“CCDD”) facility.

As the record also establishes, the cost of transporting and disposing excavated soil to a Subtitle D Landfill is approximately double for transporting, and 627 times the cost of using that same soil as fill at a CCDD facility. For large public construction projects in an urban area, such as those conducted by PBC, those are the two viable options. The cost differential for these

choices is in the millions of dollars. See *Attachment A*, List of Pending and Recent PBC Projects and Associated Costs of Dirt Removal.

Prior to the underlying legislation that drives this rulemaking, the CCDD provisions of the Illinois Environmental Protection Act (“Act”), in particular 415 ILCS 5/3.160, authorized uncontaminated soil to be considered CCDD and used as fill at permitted CCDD fill operations. However, the Act (and Board regulations) did not define uncontaminated soil. In 2006 the IEPA and Board were legislatively directed to develop a comprehensive regulatory and permitting program for CCDD fill operations. See *In the Matter of Clean Construction and Demolition Debris Fill Operations Under P.A. 94-272*, R2006-09.

During that rulemaking, which promulgated Part 1100, various participants expressed concern about the IEPA’s “we know it when we see it” approach to uncontaminated soil. In its Second Notice Opinion and Order in that rulemaking, the Board summarized the issue of uncontaminated soil (and participants’ comments regarding such) as follows:

At first notice, the Board discussed the Agency’s explanation for not defining the term “uncontaminated,” which is that the law that allows an exemption from the term “waste” using the term “uncontaminated” has been in effect for 15 years. Tr.1 at 21. The Agency stated “we knew contaminated when we saw it,” and noted that when materials do not meet the definition of CCDD, the matter can be handled as an enforcement issue. Tr.1 at 22. In its public comment, VMC requests that the Agency define “uncontaminated.” VMC states that if otherwise left undefined, the proposed Part 1100 is overbroad, impermissibly vague, and creates an incentive to avoid testing proposed clean fill material. PC 6 at 2. VMC offers examples of four different approaches used by other states to define “uncontaminated.” PC 6 at 4-5. Ultimately, VMC proposes that Illinois adopt a two-pronged test that would require the fill operator to: (1) perform due diligence on the proposed fill material; and (2) test the material only if due diligence turns up evidence that the fill has been impacted by a spill or release. PC 6 at 5-6.

L&LC supports VMC’s recommendation, but adds that as part of the definition, the Agency should set a “maximum contamination level for soils.” PC 11 at 7. LRS is also concerned about leaving the term “uncontaminated” undefined. For example, LRS questions what circumstances will trigger the Agency to do additional testing of CCDD and the standards the Agency will apply in determining whether a load of CCDD

is contaminated. PC 8 at 3. LRS contends the proposed regulations do not specifically dictate whether CCDD containing background level of contaminants can be accepted at a facility. LRS is also worried that the Agency may be able to reject a load “based upon more stringent standards nowhere stated in the proposed regulations.” *Id.*

See *In the Matter of Clean Construction and Demolition Debris Fill Operations Under P.A. 94-272*, R2006-19, July 6, 2006, at pages 3 - 4.

Despite these public comments, however, the Board found that:

“...no new language is required to clarify the definition of “uncontaminated.” In response to VMC’s concerns surrounding leaving the term “uncontaminated” undefined, the Board finds the proposed Part 1100 is not overbroad or vague, and does not create an incentive to avoid testing CCDD. The Board notes that the load-checking requirements of the proposed Section 1100.205 allow either a facility-designated inspector or an Agency inspector to reject any load resulting in a contaminant reading above background levels. Proposed 35 Ill. Adm. Code 1100.205(a), (b). Proposed Section 1100.205(a) provides that every load must be checked before acceptance at the facility, and if material other than CCDD is found at the facility, it is the owner or operator’s responsibility to remove and properly dispose of the material. See Proposed 35 Ill. Adm. Code 1100.205(f). The Agency has also stated that it will provide training if a permit holder is not meeting the Agency’s goals or standards. Tr.1 at 50.

Accordingly, because each load must be checked and may be rejected if produces readings above background levels of contamination, the Board finds that the proposed Section 1100.205 is adequately instructive and also alleviates VMC’s concern that the proposed rule would create an incentive to avoid testing CCDD fill. Also for these reasons, the Board declines to require certification from CCDD generators. The Agency has drafted the rules in a way that places the liability for complying with the Part 1100 rules on owners or operators of a CCDD fill site. The Board agrees.

As discussed above, loads of CCDD containing up to but not exceeding background levels of contaminants (interpreted based on the instrument manufacturer’s margin of error) are acceptable at a site permitted for the use of CCDD as fill material.”

See *In the Matter of Clean Construction and Demolition Debris Fill Operations Under P.A. 94-272*, R2006-19, July 6, 2006, at pages 7 -8.

However, as many in this rulemaking testified, the IEPA did indeed apply a very strict standard to determine whether it would enforce against an entity for receiving (or transferring) soil to a CCDD facility. That standard has been variously referred to as the “from God and the Glaciers” standard or the “pristine soil” standard or the “TACO Tier I strictest application” standard. And it was not applied solely to enforce against the receiving facility, as the Board envisioned, but also against haulers and site owners and soil excavators and project managers. Further, such standard was not based upon background conditions at the fill or excavation site (also as the Board envisioned). Thus, as the participants in the original CCDD rulemaking feared, history has borne out the fact that those who responsibly tested soil were at the greatest risk for enforcement, as documentary evidence existed to establish that the soils did not meet the IEPA’s un-promulgated standard.

As recently promulgated by the legislature in the public acts that drive this rulemaking, any standard promulgated by the Board in this rulemaking should be risk-based. Indeed, key to PBC’s support of this legislation is the following definition: “For purposes of this Section, the term "uncontaminated soil" means soil that does not contain contaminants in concentrations that pose a threat to human health and safety and the environment.” See 415 ILCS 5/1.1650(c).

Also key is the fact that the legislature tasked the Board, on the basis of the record in this public rulemaking, to establish a standard consistent with the above definition and a workable regulatory methodology so that responsible contractors and site owners understand their obligations under the Act. Both are subject to the Board’s legislatively-charged responsibility as set forth in Section 27 of the Act:

The Board may adopt substantive regulations as described in this Act. Any such regulations may make different provisions *as required by circumstances for different*

*contaminant sources and for different geographical areas...and may include regulations specific to individual persons or sites.* In promulgating regulations under this Act, the Board shall take into account the existing physical conditions, the character of the area involved, including the character of surrounding land uses, *zoning classifications*, the nature of the existing air quality, or receiving body of water, as the case may be, and *the technical feasibility and economic reasonableness of measuring or reducing the particular type of pollution.* The generality of this grant of authority shall only be limited by the specifications of particular classes of regulations elsewhere in this Act. 415 ILCS 5/27, emphasis added.

The general specifications set forth in the underlying legislation to guide the Board in this rulemaking is found at Section 22.51(f)(1):

No later than one year after the Board's receipt of the Agency's proposal, the Board shall adopt, rules for the use of clean construction or demolition debris and uncontaminated soil as fill material at clean construction or demolition debris fill operations. The rules must include standards and procedures necessary to protect groundwater, *which may include, but shall not be limited to*, the following: requirements regarding testing and certification of soil used as fill material, surface water runoff, liners or other protective barriers, monitoring (including, but not limited to, groundwater monitoring), corrective action, recordkeeping, reporting, closure and post-closure care, financial assurance, post-closure land use controls, location standards, and *the modification of existing permits to conform to the requirements of this Act and Board rules.*" 415 ILCS 5/22/51(f)(1), emphasis added.

Thus, notwithstanding the IEPA's proposal, it is the Board's responsibility to ensure the promulgated regulations are consistent with the charge given it by the legislature. Further discussion of that charge is found in the legislative history, which PBC placed in this record for the Board's consideration. See PC 20, Attachment A.

**II. IEPA'S PROPOSED MAXIMUM ALLOWABLE CONTAMINANT ("MAC") DOES NOT REPRESENT A SCIENCE-BASED OR RISK-BASED APPROACH TO PROTECTION OF GROUNDWATER, IS ECONOMICALLY UNREASONABLE, AND SHOULD BE ADJUSTED BY THE BOARD PRIOR TO PROMULGATION**

Testimony presented by PBC and other participants in this rulemaking establishes that IEPA's proposed rule represents an arbitrary and unreasonable approach to regulation of clean dirt for use as fill at a permitted CCDD fill operation.



**A. Economic Reasonableness**

First, the IEPA's proposed rule is economically unreasonable. As Attachment A establishes and as PBC has testified, the cost differential between allowing for the use of excavated dirt from its existing pending construction projects as fill in a permitted CCDD facility and requiring disposal in a Subtitle D facility is approximately \$15,000,000. Yet, given the size of these public projects, disposal at a Subtitle D landfill is the only reasonable alternative to allowing for the use of such dirt as fill. Waste Management's intense participation in this rulemaking, in support of the overly strict standard proposed by IEPA, provides anecdotal support for this reality.

Virtually all of PBC's projects are subject to due diligence investigation and, where warranted, testing to evaluate specific levels of targeted contaminants – those contaminants that might reasonably be found at the subject site, on the basis of an evaluation conducted by environmental professionals and based upon standard and well recognized engineering protocols, such as, but not limited to, the ASTM standards referred to in the Board's First Notice Opinion and Order. The Board need only compare Chicago soil background levels (set forth in the Board's TACO tables at 35 Ill. Adm. Code Part 742, Appendix A, Tables G and H) against the IEPA's proposed MAC to ascertain that, if the Board adopts the proposed MAC, analytical data would no doubt require that all excavated urban soil be landfilled, as it would not be acceptable for use as fill at a CCDD facility pursuant to the overly strict MAC proposed by IEPA. Instead, adoption of the proposed MAC would divert public dollars otherwise available for schools, libraries, parks, fire houses and police stations, to pay landfill disposal fees. This is not the scenario envisioned by the legislature when it charged the Board to engage its scientific and

technical expertise to make an informed determination as to the health risks associated with use of soil as fill in a CCDD fill operation.

### **B. Concentrations of Contaminants and Protection of Groundwater**

The IEPA has put forth virtually no scientific evidence in support of its proposed MAC in relation to actual risk to groundwater. Rather, its filings and testimony simply confirm, as we all know and appreciate, that groundwater is a source of drinking water in many Illinois regions and must be protected from contamination. However, as to the specific issues involved in this rulemaking (what level of contaminants can safely be contained in soils so as to be placed in permitted CCDD fill operations without impact to area groundwater) the IEPA has not connected the dots in any scientific way. Instead, the IEPA urges the Board to adopt the strictest of the TACO standards (including pathways that are not appropriately transferrable in this context) on the basis that large amounts of soil might be placed in permitted CCDD fill operations (quarries) and, on that basis alone, a risk is posed.

Such position loses sight of the fact that one of the major purposes of this rulemaking is to ascertain a MAC, and a methodology, that adequately and accurately assesses actual risk to groundwater. Simply, the IEPA position loses sight of the fact that the legislative charge here is to define clean soil in a manner that is reasonable yet protective. This is important given the very real costs for alternatives to the allowance of clean soil for use as fill at CCDD fill operations: landfilling. Clear from record testimony is the fact that the proposed MAC does not achieve that legislatively-charged objective.

The testimony of PBC's expert, Dr. William Roy, confirms the above statements. Dr. Roy's extensive Vita was provided as Exhibit I in his Pre-Filed Testimony (Exhibit 50 in this

proceeding) and in his hearing testimony (March 14, 2012 Hearing Transcript, pages 12-50).

Dr. Roy is a Senior Geochemist with the Illinois State Geological Survey (ISGS), with a Ph.D. in Soil Chemistry (Dissertation: On the Competitive Adsorption of Oxyanions by Soils). He is an Adjunct Professor in the Department of Nuclear, Plasma, and Radiological Engineering, College of Engineering, University of Illinois.

Dr. Roy is a member of many professional and technical associations, including the Soil Science Society of America. He has worked on many scientific projects important to the State of Illinois, including an ongoing research project related to potential soil impacts related to the sequestration of CO<sub>2</sub>, as well as a former research and consulting project with the Illinois Department of Agriculture (which resulted in soil remediation standards relevant to pesticide spills). See Illinois Administrative Code, Title 8, Subchapter I, Part 259. He has worked for ISGS for 32 years and, during that time, has had occasions to interact with IEPA.

As Dr. Roy testified, he reviewed the entire record in this proceeding (as it then existed, of course) prior to the preparation of his testimony. As he noted on the basis of such review: “among the first things I noticed was a lack of science.” March 14, 2012 Transcript, at p. 13. He also reviewed relevant scientific studies and literature, including the study of Chicago soils published by the United States Geological Survey, which was utilized in the development of the urban background soil tables found in TACO. See 35 Ill. Adm. Code Part 742, Appendix A, Tables G and H.

Dr. Roy testified competently on the nature and science of soils, and the actual scientific interplay between soil and groundwater. He is well versed on the nature and composition of urban soils and is both scientifically and practically knowledgeable concerning the nature and

location of quarries (“I grew up in southern Indiana in a little town called Bedford, which is nicknamed the limestone capital of the world. I grew up collecting fossils in quarries. So this is a big part of my background.” See March 14, 2012 Transcript, at p. 32-33).

The following points are clear from Dr. Roy’s testimony:

- The pH numbers used by the IEPA in its proposed MAC formula are not representative of Illinois soils, particularly Illinois soils found in northern Illinois, where the greatest economic concerns in this rulemaking are at play. Utilization of a lower pH than is actually representative of soils that enter Illinois quarries results in an overly conservative MAC, as it is based upon acidic conditions not representative of such soils, and certainly not representative of background conditions in Illinois quarries. Dr. Roy opined that the proper values to be utilized to evaluate soils relevant to this rulemaking are 6.25 through 8.74. See 35 Ill. Adm. Code Part 742, Appendix B, Tables C and D.
- Although TACO may be an appropriate methodology to utilize in this rulemaking, the specific TACO application urged by the IEPA in its proposed MAC is inappropriate in context and overly conservative: “these numbers generated from TACO are going to be conservative...we’re assuming desorption is completely reversible when experimentally (sic) we know it’s not...I don’t think TACO takes into account when lead precipitates as a solid phase. Then its dissolution behavior is not based on sorption/desorption, but a whole bag of chemistry, which I don’t think TACO takes into account.” See March 14, 2012 Transcript, at p. 19.
- The MAC proposed by IEPA does not take into account the scientific fact that, in a quarry environment, some constituents in soil (e.g., benzopyrene) will be

strongly sorbed by soil organic matter as it is “not very water soluble...it won’t come out into solution”. See March 14, 2012 Transcript, at p. 20 - 21.

- The MAC proposed by IEPA does not adequately consider the alkaline nature of Illinois quarries: “Another thing that seemed to be missing was...the influence of the quarry rocks, the limestone....[in the quarry environment] you have the sources of things that aren’t acidic that would help immobilize things like lead and zinc and copper..” See March 14, 2012 Transcript, at p. 25.
- The MAC proposed by IEPA does not adequately address the anthropogenic concentrations of urban soil, such as lead, since lead isn’t soluble in a more alkaline pH environment, endemic to Midwest quarries. See March 14, 2012 Transcript, at p. 26.
- The MAC proposed by IEPA does not allow for a more flexible (and appropriate in context) use of TACO: “TACO...talks a lot about flexibility and different options and I really admire TACO. It’s obvious a lot of thought went into it, but when I started reading how it’s applied to CCDD, a lot of those options seemed to be missing and I can’t understand why we didn’t take advantage of all the options and all the things that are there.” See March 14, 2012 Transcript, at p. 30.
- The MAC proposed by IEPA is not justified by distinctions in quarry type (rock, limestone, sand) as actual risk is inherent in an appropriate TACO analysis. See March 14, 2012 Transcript, at p. 31.
- The MAC proposed by IEPA places unwarranted emphasis on the TACO groundwater ingestion pathway, which is not suitable in the context of this rulemaking: “it didn’t make a lot of sense to me that you would consider that [the

ingestion] pathway. I was focusing on the pathway to groundwater.” See March 14, 2012 Transcript, at p. 32.

Consistent with Dr. Roy’s testimony, PBC suggests that the Board revisit the proposed MAC, and utilization of TACO methodology, as it relates to the four types of contaminants of concern: inorganics, ionizable organics, other organics in the TACO tables, and other constituents that might be eligible for chemical-specific numbers. Specifically, Dr. Roy’s testimony establishes that the final rule should (a) use realistic pH values; (b) allow Class II groundwater numbers when site appropriate; (c) use construction worker numbers for PAHs since that is the only appropriate risk-based exposure issue and (d) allow for the consideration of site specific conditions (as envisioned by the legislature). See 415 ILCS 5/3.160(c)(1) (“Any background concentrations set forth in 35 Ill. Adm. Code 742 that is adopted as a maximum concentration must be based upon the location of the quarry, mine, or other excavation where the soil is used as fill material.”). Accordingly, PBC proposes the adoption of alternative language in proposed Section 1100.605, similar to that set forth in *Attachment B*. PBC also proposes that the Board adopt rules which allow for the use of TACO background tables for all PNAs.

PBC would also support language deemed appropriate by the Board to allow for consideration of site specific conditions and operational infrastructure and protocol (groundwater monitoring, placement of fill, etc.) in the determination of any site specific MAC. PBC has consistently opposed IEPA’s one-size-fits all approach to the issue of uncontaminated soil, since such approach fails to consider actual and relevant risk.

### III. DUE DILIGENCE METHODOLOGY AND NECESSITY OF TESTING

As stated previously, PBC supports and consistently requires its professionals, consultants and contractors to utilize a due diligence investigation in the evaluation of soils prior to determination of appropriateness of transfer to permitted CCDD facility for use as fill, especially where large projects are involved. (PBC requires the transfer of excavated soil to a *permitted* facility, either Subtitle D or CCDD, as the context and investigation determine appropriate. PBC does not allow for the transfer or excavated soil to a soil only fill site allowed for in the underlying legislation.)

PBC believes that several points need to be clarified, in any rule promulgated by the Board in this rulemaking.

#### **A. Testing: Sites Which Have Never Been Developed as Industrial or Commercial and IEPA's Proposed "Potentially Impacted Property" Language**

The underlying legislation drew a line of demarcation based upon zoning categories, appropriate for the regulatory adoption by the Board pursuant to its specific authority under Section 27. Specifically, soil taken from sites that were in areas that have never before been developed as industrial or commercial were to be presumed uncontaminated, without further investigation, since the site owner or operator has the following option: certify that the soil excavation removal site "has never been used for commercial or industrial purposes and is presumed to be uncontaminated soil." See 415 ILCS 5/22.51 (D)(2)(b). This line of demarcation is appropriate, as it adequately balances risk against cost.

However, because the IEPA proposed, in its stakeholder discussions, to exclude all right-of-way excavation sites from the above-referenced option, the IEPA determined to propose to the

Board, in lieu of the above-referenced statutory language, a line of demarcation based upon a determination as to whether a particular site is a “potentially impacted property.” Such language then invoked the Board to require a Phase I –type evaluation on all properties where soil is to be considered for use as fill, regardless of whether such properties had ever before been developed as industrial or commercial. If the Board moves forward with the IEPA’s proposed language, several clarifications are crucial.

First, how does this change apply in the context of a small to medium sized project where soil is removed from an site, such as those owned by the Chicago Park District (CPD), which is in an urban area but has never before been developed as industrial or commercial? One could read the rule proposal to require a due diligence review of all such projects, at a cost which far exceeds any beneficial cost/risk analysis. For example, during the course of routine capital improvements, CPD completes many small to medium sized projects ranging from utility work to playground installation that require subsurface soil excavation. In many instances onsite reuse of that material is not feasible and soil must be hauled away. Under the currently proposed rules, for these projects where less than 150 cubic yards of excess soils are produced, CPD would likely elect not to go through the uncontaminated soil certification process for transfer to a clean construction and demolition debris (CCDD) facility for use as fill, due to the certification expense compared to disposing of the soil directly at a Subtitle D landfill.

Following is a cost analysis of the certification process as compared to disposal costs in the above-referenced hypothetical. Step one of the certification process would be having a Property Transaction Screen per ASTM E1528-06 completed at an estimated cost of \$1,000. If the property could be considered “potentially impacted” simply from having an unknown related to historical site use or even an unknown related to historical site use of an adjacent property,



then a Phase I Environmental Site Assessment would need to be completed per ASTM E1527-05 at an estimated cost of \$2,500 with soil samples collected at an estimated cost of another \$3,000. Assuming soil transportation and disposal to a CCDD facility at approximately \$10 per ton and soil disposal at a Subtitle D at \$35 per ton, disposal of 200 tons of soil at a CCDD facility would cost approximately \$2,000 and disposal of this at a Subtitle D landfill would cost approximately \$7,000. In other words, costs of assessment and disposal totals \$7,500, already exceeding the landfill cost. As the difference in cost for disposal at a Subtitle D landfill is less than the cost to for CCDD certification as uncontaminated, the site owner would likely opt for Subtitle D landfill disposal – a scenario not envisioned in the underlying legislation directing this rulemaking.

Further, unless the Board develops more realistic contaminant parameters, based upon actual risk to groundwater (see above discussion of Dr. William Roy's testimony), a public contracting entity might still opt for Subtitle D landfill disposal clean soil from even larger projects, as opposed to going through the DDCC certification process that includes investigation and testing. While the savings in disposal cost would more than cover the certification cost, the MAC as currently proposed in Subpart F make it unlikely that much, if any, urban soil would qualify.

Further, it remains unclear as to what a "recognized environmental condition" or REC is in the context of urban soil, even soil from undeveloped areas or developed residential and commercial areas, such as existing school sites. Given the IEPA's past perspective on urban soil, PBC is concerned that if the Board does not provide clarification to the concept of "potentially impacted property", the legislative concerns that were the impetus for this rulemaking will not adequately be addressed.

To address this issue, PBC proposes an amendment to proposed Section 1100.205, similar to the following:

New (a)(1)(C)

(C) for a publicly-owned source site, a certification from the source site owner or source site operator that the site is not a potentially impacted property, as can be determined through adequate knowledge of the site history, and is presumed to be uncontaminated soil. If soil is consolidated from more than one such source site owned by the same public entity, a certification must be obtained for each source site.

Then, change current (C) to (D), then add a new certification type to the subsection now marked as “D.”

(iii) for publicly-owned source site, for the person making the certification under subsection (a)(1)(C), the following language: In accordance with the Environmental Protection Act [415 ILCS 5/22.51 or 22.51a] and 35 Ill. Adm. Code 1100.205(a), I \_\_\_\_\_, [name of person authorized by public owner entity] certify that this site is not a potentially impacted property, as determined by an adequate review of the site history showing no reasonable likelihood of such impact, and the soil is presumed to be uncontaminated soil. I also certify that I am a duly authorized representative of the site owner or site operator and am authorized to sign this form. Furthermore, I certify that all information submitted, including but not limited to all attachments and other information is, to the best of my knowledge and belief, true, accurate and complete.

**B. Discretion of Environmental Professional: Testing for Targeted Compounds, based upon Investigation -- or Required Testing for Entire TACO List?**

Throughout the hearing, the IEPA suggested it intended to defer to an environmental professional's judgment on his or her determination of uncontaminated soil, and appeared to suggest that it would only go behind such judgment if there is reason to suspect malfeasance (in which case it would report to the appropriate professional licensing entity). Yet, the IEPA's proposed language (and the hearing record) leave the regulated community unclear as to how this reliance will play out in practice. This is especially crucial as to whether due diligence and the

proposed MAC requires a testing for all TACO parameters, or only those targeted as a result of site investigation. (See discussion between IEPA and Dr. Deanna Glosser, Board Member, in the March 13, 2012 Transcript, at pages 32 – 35).

In this regard, PBC proposes that (1) the language clearly allow for targeted testing and (2) language be developed to absolve the excavation site owner of any responsibility for contamination at a CCDD fill operation if it properly followed the Board's rules and relied upon the judgment of a licensed environmental professional.

#### **IV. SITE SPECIFIC CONSIDERATIONS**

As stated above, PBC has never supported the IEPA's one-size-fits all approach to the question of appropriate determinations of placement of soil in quarries so as not to pose a risk to groundwater. The IEPA's proposed rule unfortunately lumps all soil (and all CCDD placement) into one category and, in doing so, proposes an overly conservative approach to allowance of soil for use as fill in a permitted CCDD facility. This is not what the legislature envisioned. As stated by Rep. Mike Tryon in debates on the underlying legislation: "Not all soil is created equal." See PC 20, Attachment A.

#### **V. INFORMATION REQUESTED AT HEARING**

During the April 14, 2012 hearing, PBC was asked to provide additional information to the Board. These included copies of articles referenced in Dr. Roy's testimony, soil pH data from southern Illinois counties, and information about the cost to perform soil sample analysis.

A. The following articles<sup>1</sup> are attached:

- Attachment C  
Jang, Y.C. and T.G. Townsend. 2001. Occurrence of organic pollutants in recovered soil fines from construction and demolition waste. *Waste Management*, **21**, 703-715.
- Attachment D  
Townsend, T., T. Tolaymat, K. Leo, and J. Jambeck. 2004. Heavy metals in recovered fines from construction and demolition debris recycling facilities in Florida. *Science of Total Environment*, **332**, 1 – 11.
- Attachment E  
Wang, Y., S. Sikora, H. Kim, B. Dubey, and T. Townsend. 2011. Mobilization of iron and arsenic from soil by soil and demolition debris landfill leachate. *Waste Management* (in press).

B. Soil Data from Five Southern Illinois Counties

The Board requested the complete Cahill ISGS report, with specific reference to Sangamon and Hardin Counties. The report has not yet been finalized, but Dr. Roy has extracted the data in the table below from the Cahill Report. There is no data from Hardin County, so data from nearby Saline County is provided.

<sup>1</sup> These articles discuss construction and demolition debris and the fines from that debris. This material is not the same as CCDD in that it could include residues of general construction and demolition debris such as wood and treated wood. Thus, there could be sources of arsenic that would not be present in Illinois CCDD. This does not affect the conclusions Dr. Roy draws from the articles regarding the effects of pH.

**Soil pH values and corresponding depths for five southern Illinois Counties (from Cahill, in preparation).**

Greene County		St. Clair County		Sangamon County		Saline County	
Depth (feet)	pH	Depth (feet)	pH	Depth (feet)	pH	Depth (feet)	pH
0 to 0.7	6.2	0 to 0.5	6.1	0 to 0.7	5.6	0 to 0.8	6.85
0.7 to 1.4	6.3	0.5 to 1.0	6.4	1.1 to 1.6	6.2	0.8 to 1.5	4.46
2.0 to 2.4	6.1	1.0 to 1.7	6.4	3.1 to 3.7	6.4	1.5 to 2.2	4.03
2.4 to 2.9	6.2	1.7 to 2.6	5.6	4.3 to 5.0	6.4	2.2 to 3.2	4.74
4.0 to 5.0	6.3	6.1 to 6.9	7.6	6.8 to 7.4	6.6	3.5 to 4.1	6.34
18.6 to 19.2	6.2	11.8 to 12.8	7.6	17.0 to 18.2	7.7	12.4 to 13.1	8.06
Jersey County							
0 to 1.0	6.3			0 to 0.8	5.6		
1.0 to 1.5	5.4			0.8 to 1.5	6.1		
1.5 to 2.0	4.6			1.5 to 2.0	6.3		
2.0 to 2.8	4.6			2.7 to 3.4	6.8		
8.0 to 8.9	5.7			3.9 to 4.9	7.5		
17.8 to 18.2	7.9			8.4 to 8.8	7.7		

C. Soil Sampling Analytical Costs

During the March 14 hearing, Hearing Officer Tipsord requested information regarding the costs of soil testing for the parameters at issue in this proceeding. For PBC, the typical analytical cost for soils analyzing for all TACO Appendix B parameters is \$1100 per sample.

PBC is concerned about this cost for the small jobs where there is no recognized environmental concern. The sidewalk in the park is the simple example of this issue. The analytical cost for each such tiny project becomes quite unreasonable when there could be many projects, but none have any more expected environmental concern than the long time park. Sampling at every project makes some sense for the large projects, but seems absurd for the sidewalk in the park example. Rather, we contend only the acceptance criteria for the permitted CCDD site would apply, which would not likely mandate the sampling of every scoop of dirt that may come from even un-impacted properties

**VI. CONCLUSION**

PBC appreciates the opportunity to have presented argument and testimony in this rulemaking and thanks the Board for its consideration of these comments. PBC looks forward to providing additional comments in its April 27 filing.

Dated: April 18, 2012

Respectfully submitted,

**PUBLIC BUILDING COMMISSION  
OF CHICAGO**

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ATTACHMENT A

**Public Building Commission of Chicago Subtitle D Disposal Costs compared to CCDD Costs” and today’s date, Input provided by Concord Group and PBC Job Order Contract estimators**

Project	Site Preparation Tons	Construction	UM	Subtitle D			CCDD			Total Difference
				Hauling per Ton	Disposal per Ton	Total	Hauling per Ton	Disposal per Ton	Total	
Back of the Yards HS		77,590	TONS	13.00	22.00	\$2,715,650	5.25	3.25	\$659,515	\$2,056,135
Boone Clinton Elementary School	7,792		TONS	13.00	22.00	\$272,720	5.25	3.25	\$66,232	\$206,488
Brighton Park II	29,969	1,750	TONS	13.50	22.00	\$1,126,025	5.25	3.25	\$269,612	\$856,413
Brooks College Prep	23,748	15,920	TONS	11.50	22.00	\$1,328,878	7.00	3.25	\$406,597	\$922,281
CPD - District 12 - Blue Island	7,394	43,300	TONS	13.50	22.00	\$1,901,025	5.50	3.25	\$443,573	\$1,457,453
Durkin Park Annex	5,552	2,930	TONS	11.50	22.00	\$284,147	7.00	3.25	\$86,941	\$197,207
Edgebrook Addition	31,507	3,030	TONS	11.50	20.00	\$1,087,916	8.00	3.25	\$388,541	\$699,374
Engine 16	14,885	2,270	TONS	13.00	22.00	\$600,425	6.00	3.25	\$158,684	\$441,741
Holmes		369	TONS	13.00	22.00	\$12,915	6.00	3.25	\$3,413	\$9,502
Humboldt Park Library	4,951	3,669	TONS	13.00	20.00	\$284,460	7.50	3.25	\$92,665	\$191,795
Jones HS		45,490	TONS	13.00	22.00	\$1,592,150	6.00	3.25	\$420,783	\$1,171,368
Lloyd ES Artificial Turf	2,200		TONS	13.00	22.00	\$77,000	3.00	3.25	\$20,350	\$56,650
Mt. Greenwood School	4,333	2,980	TONS	11.50	20.00	\$230,360	7.00	3.25	\$74,958	\$155,401
Ogden School	5,445	34,030	TONS	13.50	20.00	\$1,322,413	6.00	3.25	\$365,144	\$957,269
Onahan Annex	6,056	8,000	TONS	12.50	20.00	\$456,820	8.50	3.25	\$165,158	\$291,662
Osterman Beach Comfort Station		540	TONS	13.00	20.00	\$17,820	6.50	3.25	\$5,265	\$12,555
Powell School		14,270	TONS	13.00	22.00	\$499,450	6.50	3.25	\$139,133	\$360,318
Southshore HS (new)	74,837		TONS	13.00	22.00	\$2,619,295	6.50	3.25	\$729,661	\$1,889,634
Stevenson Annex	5,545	10,370	TONS	11.50	22.00	\$533,153	6.50	3.25	\$155,171	\$377,981
Southwest Area High School		102,667	TONS	13.00	22.00	\$3,593,345.00	6.50	3.25	\$1,001,003.25	\$2,592,341.75
						<b>\$20,555,964.50</b>			<b>\$14,903,567.50</b>	

ATTACHMENT B

Section 1100.605 Maximum Allowable Concentrations for Chemical Constituents in Uncontaminated Soils

- a) Except as provided for background concentrations in subsection (b) of this Section, the maximum allowable concentrations for chemical constituents in uncontaminated soil must be determined pursuant to subsections (a)(1) through (a)(5) of this Section.
- 1) The maximum allowable concentration for a chemical constituent in uncontaminated soil will be the lowest Tier 1 chemical-specific soil value of the exposure routes for residential and construction worker receptors set forth in 35 Ill. Adm. Code 742. Appendix B, Tables A and B (e.g., soil ingestion exposure route, outdoor inhalation exposure route, soil component of the groundwater ingestion exposure route, construction worker exposure route), unless the CCDD fill site owner can demonstrate, pursuant to supplemental permit application, that one or more of those exposure routes are not reasonably required to protect groundwater, based upon site conditions and established operational infrastructure or protocols. ~~Class I~~Site groundwater classification values must be used when determining the lowest Tier 1 chemical-specific value for the soil component of the groundwater ingestion exposure route. Before making the comparison among exposure routes to determine the lowest value for ionizing organic chemical constituents and inorganic chemical constituents, the requirements of subsections (a)(2) and (a)(3) of this Section must be satisfied, as applicable.
  - 2) For ionizing organic constituents, the lowest pH-dependent value, using pH values from 6.25 through 8.74 for the soil component of the ~~Class I~~site groundwater classification for the groundwater ingestion exposure route in 35 Ill. Adm. Code 742. Appendix B, Table C or D must be substituted for the pH- neutral value provided for the soil component of the ~~Class I~~site groundwater classification groundwater ingestion exposure route in Appendix B, Table A or B before determining the lowest Tier 1 chemical-specific soil value pursuant to subsection (a)(1) of this Section.
  - 3) For inorganic constituents, the remediation objectives for the soil component of the ~~Class I~~ groundwater ingestion exposure route in Appendix B, Tables A and B are based on the contaminant concentration resulting from an extraction test and are not directly comparable to the remediation objectives provided for the ingestion and inhalation exposure routes, which are based on total concentrations. The following values, based on total concentrations, must be substituted for the extraction test values in Table A before determining the lowest Tier 1 chemical-specific soil value pursuant to subsection (a)(1) of this Section:



- A) The lowest chemical-specific, pH-dependent values, using pH values from 6.25 through 8.74, in Appendix B, Table C or D; or
- B) For inorganic constituents that are listed in Appendix B, Tables A or B but not in Appendix B, Tables C or D, the extraction test values for the soil component of the groundwater ingestion exposure route in Appendix B, Table A may be multiplied by twenty (i.e., 20 liters/kilogram, the liquid to solid ratio in the extraction test assuming complete constituent leaching) to enable direct comparison with the ingestion and inhalation exposure route values. The resulting value must be substituted for the extraction test value before determining the lowest Tier 1 chemical-specific soil value pursuant to subsection (a)(1) of this Section.
- 4) If the lowest Tier 1 soil value for a chemical is less than the Acceptable Detection Limit (ADL), the ADL will serve as the lowest soil value.
- 5) The total concentration of organic contaminants may not exceed the attenuation capacity of the soil as determined in accordance with subsections (b)(1) and (b)(1)(A) of 35 Ill. Adm. Code 742.215 using a default value of 2000 mg/kg for the natural organic carbon fraction ( $f_{oc}$ )
- b) Background concentrations from 35 Ill. Adm. Code 742. Appendix A, Tables G and H may be used as the maximum allowable concentrations at locations specified by the tables if the most stringent exposure route value for the chemical constituent as determined pursuant to subsection (a) of this Section is lower than the chemical's applicable background value listed in Tables G or H. The chemical's applicable background value in Table G or H must be established based on the location of the fill operation where the soil is placed.
- c) For chemicals not listed in 35 Ill. Adm. Code 742. Appendix B, Tables A, B, or C, the values may be obtained from the Agency by making a request for chemical-specific values.
  - 1) The Agency will develop these objectives based upon the United States Environmental Protection Agency's (USEPA) toxicity value hierarchy as specified in OSWER Directive 9285.7-53, incorporated by reference at Section 1105.115 of this Part. USEPA's Integrated Risk Management System (IRIS), incorporated by reference at Section 1100.104 of this Part, is the first tier of this hierarchy.
  - 2) Calculation of the maximum allowable concentrations must use the applicable risk-based soil screening level equations from 35 Ill. Adm. Code 742. Appendix C, Table A. Default exposure durations and contact rates from 35 Ill. Adm. Code 742. Appendix C, Table B must be used in making these calculations.
  - 3) If the person making the request of the Agency disagrees with the Agency's decision, the person who made the request may file an appeal of the

Agency's decision with the Board pursuant to Section 40(a) of the Act (415 ILCS 5/40(a)) and 35 Ill. Adm. Code 105.

- d) Other provisions of 35 Ill. Adm. Code 742 (e.g., institutional controls, engineered barriers, exposure route exclusions, site-specific evaluations, local area background calculations) may not be used to exclude or otherwise alter exposure routes or exposure route values for the purpose of determining the maximum allowable concentrations under this Part.
- e) For purposes of this Part, the Agency shall publish at its website a list of chemical-specific values for maximum allowable concentrations of chemical constituents in uncontaminated soils based on the methodology for determining those values set forth in this Section. In addition, the Agency shall publish at its website a list of chemical-specific values for chemicals not listed in 35 Ill. Adm. Code 742.Appendix B, Tables A, B or C when values are calculated by the Agency in accordance with subsection (c) of this Section or subsection (c ) of 35 Ill. Adm. Code 742.510.

(Source: Added at 36 Ill. Reg. , effective )



PERGAMON

## ATTACHMENT C

Waste Management 21 (2001) 703–715

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Occurrence of organic pollutants in recovered soil fines  
from construction and demolition wasteYong-Chul Jang<sup>a</sup>, Timothy G. Townsend<sup>b,\*</sup><sup>a</sup>Post-Doctoral Associate, Environmental Engineering Sciences, University of Florida, PO Box 116450, Gainesville, FL 32611-6450, USA<sup>b</sup>Assistant Professor, Environmental Engineering Sciences, University of Florida PO Box 116450, Gainesville, FL 32611-6450, USA

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**Abstract**

The objective of this study was to characterize recovered soil fines from construction and demolition (C&D) waste recycling facilities for trace organic pollutants. Over a period of 18 months, five sampling trips were made to 14 C&D waste recycling facilities in Florida. Screened soil fines were collected from older stockpiles and newly generated piles at the sites. The samples were analyzed for the total concentration (mg/kg) of a series of volatile organic compound (VOCs) and semi-volatile organic compounds (semi-VOCs). The synthetic precipitation leaching procedure (SPLP) test was also performed to evaluate the leachability of the trace organic chemicals. During the total analysis only a few volatile organic compounds were commonly found in the samples (trichlorofluoromethane, toluene, 4-isopropyltoluene, trimethylbenzene, xylenes, and methylene chloride). A total of nine VOCs were detected in the leaching test. Toluene showed the highest leachability among the compounds (61.3–92.0%), while trichlorofluoromethane, the most commonly detected compound from both the total and leaching tests, resulted in the lowest leachability (1.4–39.9%). For the semi-VOC analysis, three base-neutral semi-VOC compounds (bis(2-ethylhexyl)phthalate, butyl benzyl phthalate, and di-n-butyl phthalate) and several PAHs (acenaphthene, pyrene, fluoranthene, and phenanthrene) were commonly detected in C&D fines samples. These compounds also leached during the SPLP leaching test (0.1–25%). No acid extractable compounds, pesticides, or PCBs were detected. The results of this study were further investigated to assess risk from land applied recovered soil fines by comparing total and leaching concentrations of recovered soil fines samples to risk-based standards. The results of this indicate that the organic chemicals in recovered soil fines from C&D debris recycling facilities were not of a major concern in terms of human risk and leaching risk to groundwater under reuse and contact scenarios. © 2001 Elsevier Science Ltd. All rights reserved.

**Keywords:** Recovered soil fines; Organic chemical leaching; C&D waste recycling; Synthetic precipitation leaching procedure; Risk assessment

**1. Introduction**

Construction and demolition (C&D) waste may be defined as waste generated from the construction, renovation and demolition of buildings and other such structures. C&D waste results from residential, commercial, industrial, and governmental activities. The major components of C&D waste include wood, concrete and brick, asphalt pavement and roofing, metal, dirt, rock and vegetation from land-clearing operations [1]. The recycling of C&D waste has evolved into a large and growing industry in the United States over recent

years. Issues such as increasing disposal costs and the recycling potential of many of the components in the C&D waste stream have resulted in the start-up of a large number of C&D recycling operations [1–4]. These recycling operations typically consist of up-front removal of large recoverable items (wood, concrete) followed by a separation process for the remaining smaller materials. One processing step is the screening of the waste, which results in a fine, soil-like fraction, defined here as recovered soil fines. Recovered soil fines have been observed to make up approximately 25% or more of the mass of C&D waste at these recycling facilities [5]. The availability of viable reuse markets for C&D fines is therefore a necessary step for the success of most C&D waste recycling facilities. Replacement of soil in construction activities (embankments, soil fill)

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and use as landfill cover have been the primary reuse markets proposed for recovered soil fines from C&D waste processing.

The reuse of a recovered waste such as C&D waste fines must be balanced with the need to protect human health and the environment. Such protection is provided by following applicable regulations and policies. It is no longer sufficient to simply determine whether a solid waste is a regulatory hazardous waste or not. The risks to human health and the environment must be evaluated for different reuse scenarios, even for wastes classified as non-hazardous by regulation. Appropriate regulatory requirements must be met before other waste-derived materials (e.g. compost, biosolids) are reused or land-applied. Concerns must be addressed in regard to the potential for human exposure and the impact on environmental resources, particularly groundwater.

In response to the rapid growth of the C&D recycling industry in Florida and the generation of large quantities of recovered soil fines, a study was initiated to characterize this material with regard to trace chemical components. This paper reports the results for the occurrence of trace organic pollutants. The results of this study were further examined to assess risk from land-applied recovered soil fines by comparing to risk-based standards. The impact of heavy metals has been reported elsewhere [5].

## 2. Background

### 2.1. C&D waste and material recycling facility

The recycling of C&D waste may be performed by separation at the source or by means of a material recovery facility (MRF) [6,7]. While separation at the source may provide the best chance for successful separation and removal of hazardous chemicals (e.g. solvent containers), factors such as economics and resistance to changing existing operations have limited the application of this activity. More commonly, C&D MRFs have been constructed as an alternative to traditional C&D landfills. In areas where land is scarce, or where regulations limit the use of landfills, C&D debris recycling is often the most economical solution for generators and transporters of C&D waste.

A C&D MRF typically consists of a tipping area for the unloading of waste from trucks or containers, a waste picking line, a hopper, a conveyor belt, a magnet, a screen shaker, and a grinder. Identifiable bulky waste materials (i.e. carpet, corrugated cardboard, trees from land clearing, steel studs), which are easily separated and can be readily recycled, are removed by manual labor or a mechanical arm. The remaining waste is processed to separate or recover additional reusable

materials. An optional step prior to separating the C&D debris components is size reduction. Methods of size reduction vary from crushing using steel-wheeled compactors or other heavy equipment to employing devices such as hammer mills, crushers, or impactors that use mechanical force to disintegrate the material. The material (size-reduced or not) is sent through a series of unit operations to remove the various materials. Examples of these operations include manual sorting, screening, magnetic separation, and density separation (float tanks or air classifiers). The MRFs that perform size-reduction tend to employ the greatest amount of mechanical separation.

The screening step is typically performed early in the process. The types of screens used include bar screens, trommel screens, and vibratory screens. The material which passes the screen at C&D waste recycling facilities has been commonly referred to as recovered screen material (RSM) or recovered soil fines. The physical characteristics of the screened fines vary as a function of the screen size used. Screen sizes typically range from 0.6 to 5 cm (0.25–2 in). The composition of the materials that pass the screen is a function of the incoming waste stream and the manner in which the waste is handled. The largest percentage of the material is soil-like fines resulting from site soil that is disposed with the rest of the C&D debris. Other possible contributing sources include dust and fines generated from the demolition of structures, and unused aggregate or sand from construction activities. Materials often observed as part of recovered soil fines include small pieces of dry-wall, wood, rock, paper, and plastic. A greater percentage of foreign materials tend to be observed with larger screens and in operations where the feed stream is size-reduced.

### 2.2. Organic pollutants in C&D waste

Although most of the materials in C&D waste streams are relatively inert, a potential environmental impact from hazardous materials in C&D waste does exist [8]. Table 1 presents a list of organic chemical categories and possible sources of hazardous chemicals encountered in the C&D waste stream. These hazardous or toxic components may either exist in a form in which they cannot be removed from the non-hazardous constituents (painted or treated wood), or they may be present in the mixed C&D waste stream as separate items (i.e. solvent containers, paint containers, and paint thinner containers). Contamination of recovered soil fines with organic chemicals may result from bulk chemicals which are disposed of in waste containers or on the soil at the site, from chemicals which are affixed to the surface of materials occurring in the soil fines, or from chemicals which leach from other materials in the C&D waste stream during storage. Contamination with

chemicals such as fuel and solvents may also occur as a result of the operation and maintenance of equipment and vehicles at C&D debris recycling facilities.

### 2.3. Risk assessment for organics in recovered soil fines — total analysis and leaching analysis

The criteria used for assessing risk in beneficial reuse applications of waste-derived materials is an evolving area. While federal risk-based standards have been established for municipal biosolids [9], no such standards exist for wastes such as C&D debris fines. Many states have adopted approaches that employ generic soil screening levels (SSLs). The SSLs are risk-based pollutant concentrations (mg/kg) derived from standardized equations combining exposure assumptions with toxicological information. A method frequently employed to determine SSLs was developed and is used by the US EPA superfund program [10].

Two general routes of risk must be evaluated. These include direct exposure (direct ingestion, inhalation of volatiles and fugitive dusts, and dermal absorption) and leaching to groundwater. In some cases ecological risk is of a concern, but guidelines for this are less developed. For direct exposure the total concentration of a pollutant (mg/kg) is compared to a risk-based level. The level may be calculated based on site- or media-specific conditions, but in most cases, a generic default concentration is used. The default values are typically used for assessment of beneficial reuse of waste materials because the material is destined for many possible locations, making site-specific assessment difficult. The generic number is usually a concentration conservatively determined for the different direct exposure routes, or in some cases, multiple risk pathways are factored into a simple concentration. Some states have different risk-based SSLs for different reuse scenarios. For example, in Florida default soil cleanup target levels (SCTLs) have been established for both residential and industrial reuse and contact scenarios [11].

The evaluation of the leaching route is performed via two different mechanisms. In one, a leaching test is performed, and the pollutant concentration in the leachate is compared directly to applicable groundwater standards. The common leaching test used is the synthetic precipitation leaching procedure (SPLP), which mimics leaching under conditions of acidic rainfall (US EPA SW 846 Method 1312) [10,12]. This is preferred over the toxicity characteristics leaching procedure (TCLP), which simulates leaching within a domestic waste landfill. The performance of a leaching test is an additional step beyond the measurement made for direct exposure. The second mechanism for assessing leaching risk involves comparing the total concentration (mg/kg) to a risk-based leaching level. The risk-based leaching level represents the theoretical concentration of a pollutant in a soil or waste (mg/kg) that results in a leachate concentration that exceeds the applicable groundwater standard for that pollutant. An equation for the determination of these leaching levels is described as follows [10]:

$$C_t = C_w \times \left\{ K_{oc} \cdot f_{oc} + \left( \frac{\theta_w + \theta_a \cdot H'}{\rho_b} \right) \right\}$$

where

$C_t$	=	screening level in soil (mg/kg)
$C_w$	=	target leachate concentration (mg/l) ( $C_{\text{standard}} \times \text{dilution factor}$ )
$C_{\text{standard}}$	=	risk-based groundwater guidance concentration or regulation
$K_{oc}$	=	soil organic carbon-water partition coefficient (l/kg)
$f_{oc}$	=	organic carbon content of soil (kg/kg)
$\theta_w$	=	water-filled soil porosity ( $l_{\text{water}}/l_{\text{soil}}$ )
$\theta_a$	=	air-filled soil porosity ( $l_{\text{air}}/l_{\text{soil}}$ )
$H'$	=	dimensionless Henry's law constant
$\rho_b$	=	dry soil bulk density (kg/l).

Table 1  
Organic chemicals potentially encountered in construction and demolition (C&D) waste

Class of organic chemicals	Potential contributor in the C&D waste stream
Polycyclic aromatic hydrocarbons	Asphalt cement, roofing tiles, pavement, petroleum distillates in paint thinner, waste oil and greases, oil filters, fuel products
Ethylbenzene, xylenes, toluene, other volatile organic compounds	Oil varnishes, stains, paint containers and waste paint products, solvent containers and waste, aerosol spray, other chemical additives
Phthalic acid esters	Tubings, hoses, other PVC products
Pesticides	Pesticide application to carpets, floors in demolition activities; pesticide application to soil removed in land-clearing and demolition, pesticide containers
Herbicides	Previous application to soil and plants removed in land-clearing and demolition activities, herbicide containers
Polychlorinated biphenyls	Electrical lighting ballasts

While the performance of a leaching test should better reflect actual leaching concentrations of a given soil, the generic risk-based leaching level provides a less expensive evaluation. In most circumstances, the leaching test is only conducted after the generic risk-based leaching concentration (mg/kg) is exceeded.

### 3. Methods and materials

Over a period of 18 months, five sampling trips were made to a total of 14 C&D waste recycling facilities in Florida. Recovered soil fines samples were collected from older stockpiles and from newly generated (or fresh) piles at the sites. The samples were analyzed for volatile organic compounds and semi-volatile organic compounds following standardized US EPA methodology. A SPLP test was also performed on selected samples to examine the leachability of organics.

#### 3.1. Sample collection

Samples were collected according to established procedures for laboratory operations and sample collection activities [13]. Table 2 presents the source of the C&D fines (fresh, stockpiled), the number of sites visited, the number of samples analyzed and collected, and the parameters analyzed. The objective of this study was to broadly characterize this waste stream and thus no repetitive sampling events to a particular site were performed to examine temporal trends. Stockpiled samples were collected using stainless steel augers at a depth of approximately 90 cm (3 ft) into the pile. In some cases a backhoe was used to excavate down to 1.5 m (5 ft). The sample locations from the stockpiles were selected using a simple random sampling method with a random grid technique outlined in US EPA protocols (US EPA SW-846 Chapter nine) [12]. Fresh pile samples

were collected directly from the pile underneath the conveyor belt supplying the recovered soil fines, typically by quartering the pile.

All samples were collected with stainless steel trowels or spoons. Samples for volatile organic chemicals were collected directly from stockpiles as well as from the fresh pile underneath the conveyor belt in 40-ml vials (I-Chem. Corp.) with Teflon lined septa. The volatile organic compound (VOC) vials were tapped slightly as they were filled to eliminate as much free air space as possible. Samples for semi-volatile organics were collected in 230-ml (8-oz.) glass jars (I-Chem Corp.) with Teflon lined lids. Semi-VOC samples were collected from composites of grab samples. A minimum of four grab samples were composited in stainless steel mixing bowl (7.4 liters) to obtain representative samples from the old and fresh piles. The samples were stored below 4°C in an iced container and transported to a walk-in cooler located at the University of Florida Solid and Hazardous Waste Laboratory prior to analysis.

To implement quality assurance (QA) practices in the field and laboratory analyses, trip blanks, field blanks, equipment blanks, and duplicate samples were carried or collected during sampling trips. The trip blank was prepared in 40-ml VOC vials and 230-ml glass jars in the laboratory using organic-free reagent blank. Trip blanks were taken to the sampling sites and returned to the laboratory to document contamination attributable to shipping and field handling procedures. Field blanks were collected into the vials and jars from baked silica sand, which was prepared from the laboratory (at 105°C in oven overnight) before the sampling trips. The field blank was used to document field contamination during sample collection. Equipment blanks were collected from an aliquot of organic-free reagent water, which was used to rinse the sample equipment (e.g. spatula, spoon, stainless steel bowl).

Table 2  
Sample collection and analysis<sup>a</sup>

Sample collection trip	Types of samples collected	A number of sites	A number of samples analyzed					
			Total analysis				Leaching analysis	
			VOC	Base/neutral, acids, pesticides	PAH	PCB	VOC	Semi-VOC
1	Stockpiled, fresh	3	24	23	NA	NA	NA	NA
2	Stockpiled, fresh	1	10	NA	NA	NA	NA	NA
3	Fresh	12	NA	12	12	12	NA	NA
4	Fresh	3	3	6	6	6	3	6
5	Fresh	1	6	6	6	6	3	6
Total			43	47	24	24	6	12

<sup>a</sup> VOC, volatile organic compounds; PAH, polycyclic aromatic hydrocarbons; PCB, polychlorinated biphenyl.

### 3.2. Sample preparation and analysis

The samples were extracted and analyzed using the EPA methods in Table 3. Gas chromatography/mass spectrometry (GC/MS) was used for a total analysis of volatile and semi-volatile organic compounds in the samples. Leaching tests were also performed on the samples using the synthetic precipitation leaching procedure.

#### 3.2.1. Total VOCs

VOC analysis was carried out using a Tekmar 3000 and Tekmar Precept purge-and trap concentrator attached to a Hewlett Packard 5890 gas chromatograph and a Hewlett Packard 5985 mass spectrometer (US EPA SW-846 Method 8260A). Table 4 presents the volatile organic compounds (halogenated organic compounds and aromatic hydrocarbons) analyzed. Approximately 5 g of a recovered soil fines sample was placed into a 40-ml VOC vial with a 0.32-cm (0.125 inch) thick Teflon lined cap. Deionized water (10 ml) was then added to the sample. An inert gas (helium) was bubbled through the sample at ambient temperature, transferring the volatile components from the aqueous phase to the vapor phase, where they were swept through an adsorbent trap. After purging (10 min), the sorbent trap (Tenax/Silica Gel/Charcoal GLT trap) was heated and backflushed with the inert gas to desorb trapped sample components. The desorbed analytes were cryofocused onto the capillary column for 4 min. The GC column used to separate the analytes was a 60 m×0.53 mm i.d. 30 μm film thickness DB 624 (J&W Scientific). The initial temperature was 35°C for 6 min, programmed at 10°C/min to 250°C, with a 5 min final hold. The analytes were detected with a mass spectrometer interfaced to the gas chromatography.

During Trips 4 and 5 a methanol extraction technique was used for the total analysis of VOCs (US EPA SW-846 Method 5030A-high concentration method). Methanol extraction has been reported as a more efficient and robust method for determining VOCs in soils [14]. A 5-g soil fines sample was placed into a volumetric flask with 50 ml of HPLC grade methanol. The flask was capped and shaken for 2 min. These steps were performed in a refrigerated room rapidly to avoid any possible loss of volatile organic chemicals. A 1-ml aliquot of the extract was transferred into a 50 ml volumetric flask, which was diluted with 49 ml deion-

ized water. An aliquot of this extract from the flask was placed into a 40-ml VOC vial with zero headspace for the GC/MS analysis of volatile organics according to EPA SW 846 Method 8260A.

#### 3.2.2. Leaching volatile organic compounds

A leaching test for volatile organics from recovered soil fines was conducted using a zero headspace extractor (ZHE; Analytical Testing Corporation). Approximately 25 g of a recovered soil fines sample was placed in the ZHE. In order to prevent the loss of volatile compounds, excessive sample manipulation was kept to a minimum, and sample loading was performed in a refrigerated room below 4°C. A 500-ml SPLP leaching solution was then added to the ZHE. The SPLP utilizes a slightly acidic solution (pH = 4.20±0.05) designed to simulate acidic rainwater by adding the 60/40 weight percent mixture of sulfuric and nitric acids. The test was performed in the same manner as the TCLP test (US EPA SW 846 Method 1311) [12]. The ZHE unit was placed in a rotary extractor and rotated for 18±2 h at 30 rpm at room temperature. After tumbling, the filtered leachate was collected into a glass syringe (Hamilton Gastight™ Syringe). The sample was then analyzed following the same VOC analytical method described above.

#### 3.2.3. Total semi-VOC: extraction and analysis

An ultrasonic extraction technique (Sonicator™ Model W-375 (Heat Systems-Ultrasonics, Inc.) was used for extracting Semi-VOCs from recovered soil fines samples (US EPA SW-846 Method 3550A). The ultrasonic process ensures intimate contact of the sample matrix with the extraction solvent (acetonitrile). A 30-g soil fines sample was weighed into a 400-ml erlenmeyer flask with 100 ml of the extraction solvent. The semi-volatile components were extracted ultrasonically for 3 min, with the output control knob set at 10 (full power) and with the percent duty cycle knob set at approximately 50% and a pulse energy mode. The extract was then filtered through Whatman No.41 filter paper using vacuum filtration. This ultrasonic extraction was repeated two times with two additional 100-ml portions of the solvent. A sodium sulfate filtration process was then used to remove water in the extract. After filtration, a solvent rotary evaporation apparatus (Rotavapor RE 111, Brinkman) was employed to further concentrate the extract to approximately 8–10 ml. A nitrogen blowdown

Table 3  
Analytical methods for total organics analysis and leaching analysis

Parameters	EPA Methods (Extraction/Analysis)
Total VOCs <sup>a</sup>	US EPA SW-846 Method 5030A/8260A
Total semi-VOCs	US EPA SW-846 Method 3550A/8270B
Leachable VOCs and semi-VOCs	US EPA SW-846 Method 1312/8260A/8270B

<sup>a</sup> Volatile organic compounds.

Table 4  
 Target organic compounds<sup>a</sup>

VOCs	1,4-Dichlorobenzene	Hexachlorocyclopentadiene
Bromochloromethane	1,2,3-Trichlorobenzene	Hexachloroethane
Bromodichloromethane	1,2,4-Trichlorobenzene	Isophorone
Bromoform	Bromomethane	Nitrobenzene
Carbon tetrachloride	Chloroethane	
Chloroform	Trichlorofluoromethan	Pest/herb
Dibromochloromethane	Semi-VOCs acids	Alpha-BHC
Dibromomethane		Gamma-BHC
Methylene chloride	2-Chlorophenol	4,4'-DDD
1,2-Dibromoethane	2,4-Dichlorophenol	Dieldrin
1,1-Dichloroethane	2,4-Dimethylphenol	Endrin
1,2-Dichloroethane	2-Methyl-4,6-dinitrophenol	Heptachlor
1,1-Dichloroethene	2,4-Dinitrophenol	Heptachlor epoxide (isomer A)
<i>cis</i> -1,2-dichloroethene	2-Nitrophenol	Aldrin
<i>trans</i> -1,2-dichloroethene	4-Nitrophenol	Beta-BHC
1,1,1,2-Tetrachloroethane	Pentachlorophenol	4,4'-DDE
1,1,2,2-Tetrachloroethane	Phenol	Endosulfan I
Tetrachloroethene	2,4,6-Trichlorophenol	Endosulfan II
1,1,2-Trichloroethane	Base/neutrals	PAHs
Trichloroethene		Acenaphthene
1,3-Dichloropropane	<i>bis</i> (2-chloroethoxy)methane	Anthracene
<i>cis</i> -1,3-dichloropropene	<i>bis</i> (2-chloroethyl)ether	Benzo[ <i>k</i> ]fluoranthene
<i>trans</i> -1,3-dichloropropene	<i>bis</i> (2-chloroisopropyl)phthalate	Chrysene
1,2,3-Trichloropropane	<i>bis</i> (2-ethylhexyl)phthalate	Fluorene
Benzene	4-Bromophenyl phenyl ether	Indeno[1,2,3- <i>cd</i> ]pyrene
<i>Sec</i> -butylbenzene	Butyl benzyl phthalate	Pyrene
<i>Tert</i> -butylbenzene	4-Chlorophenyl phenyl ether	Benz[ <i>a</i> ]anthracene
Ethylbenzene	Di- <i>n</i> -butyl phthalate	Benzo[ <i>b</i> ]fluoranthene
Isopropylbenzene	Diethyl phthalate	Benzo[ <i>ghi</i> ]perylene
4-Isopropyltoluene	Dimethyl phthalate	Benzo[ <i>a</i> ]pyrene
Naphthalene	Di- <i>n</i> -octyl phthalate	Dibenz[ <i>a,h</i> ]anthracene
<i>n</i> -Propylbenzene	<i>n</i> -nitrosodimethylamine	Fluoranthene
Styrene	<i>n</i> -nitrosodiphenylamine	Phenanthrene
Toluene	<i>n</i> -nitroso-di- <i>n</i> -propylamine	
1,2,4-Trimethylbenzene	Azobenzene	PCBs
1,3,5-Trimethylbenzene	2-Chloronaphthalene	
<i>o</i> -xylene	1,2-Dichlorobenzene	2-Chlorobiphenyl
<i>m/p</i> -xylene	1,3-Dichlorobenzene	2,3-Dichlorobiphenyl
Bromobenzene	1,4-Dichlorobenzene	2,2',4,6-tetrachlorobiphenyl
Chlorobenzene	2,4-Dinitrotoluene	2,2',3,4,5'-Pentachlorobiphenyl
2-Chlorotoluene	2,6-Dinitrotoluene	2,2',3,3',4,5',6,6'-octachlorobiphenyl
4-Chlorotoluene	Hexachlorobenzene	
1,2-Dichlorobenzene	Hexachlorobutadiene	

<sup>a</sup> VOC, volatile organic compounds; PAHs, polycyclic aromatic hydrocarbons; PCBs, polychlorinated biphenyl.

apparatus (N-EVAP Model 111, Organomation Assoc. Inc.) was then used to reduce the solvent volume to approximately 2 ml using a gentle stream of clean, dry nitrogen. During nitrogen blowdown evaporation, the concentrated extract tube was placed in a warm water bath (approximately 35°C).

One micro liter of the concentrated sample was analyzed by GC/MS (US EPA SW 846 Method 8270B) using a Perkin-Elmer GC (Model 8500) attached to an ion trap detector (Perkin-Elmer ITD Model 6210). The GC column was a 30 m×0.25 mm i.d. 0.25 μm film thickness DB5 (J&W Scientific). The initial temperature set at 40°C for 4 min, and then ramped to 300°C at 10°C/min, with a 5-min final hold. Semi-VOCs analyzed using this

method include base/neutral and acid extractables (BNAs), pesticides, polycyclic aromatic hydrocarbons (PAHs), and polychlorinated biphenyls (PCBs; Table 4).

### 3.2.4. Leaching semi-VOC: SPLP leaching, extraction, analysis

A SPLP was also conducted for semi-volatile organic compounds on samples from Trips 4 and 5. The SPLP extraction process for semi-VOC includes placing 100-gram of sample in a 2-liter glass extraction vessel, adding 2 liters of SPLP leaching solution in the vessel, tumbling for 18±2 h, and filtering the extract using a pressurized filtration apparatus with a 0.7-μm borosilicate glass fiber filter (Whatman TCLP Filters). The filtered lea-



chate was extracted using a separatory funnel liquid-liquid extraction (US EPA SW 846 Method 3510B). The extract was then analyzed using the same method as the semi-volatile compound analysis (US EPA SW 846 Method 8270B).

During VOC and semi-VOC total analyses, samples spiked with known concentrations of selected chemicals (matrix spikes and surrogate spikes) were analyzed to measure matrix effects, to check the recovery of the organic chemicals from the sample matrix, and to track analytical error. In order to monitor the performance of the analytical methods on VOC ZHE extracts and to assess the presence of interferences, matrix spikes were added to randomly selected recovered soil fines samples after SPLP ZHE leaching but prior to VOC analysis. During semi-VOCs SPLP leaching tests matrix spikes and surrogate spikes were added to randomly selected samples after leaching and filtration but prior to semi-VOC extraction.

## 4. Results and discussions

Total analysis results of trace organic compounds in recovered soil fines are presented and discussed. The results are then compared to Florida's risk-based SCTLs (residential direct exposure, industrial direct exposure, and leaching standards) [11]. Leaching results from SPLP tests are provided to examine the leachability of trace organics in recovered soil fines. The results are compared to Florida's groundwater cleanup target levels (GWCTLs) [11]. Other details such as quality assurance/quality control (QA/QC) results can be found elsewhere [5].

### 4.1. VOC analysis

#### 4.1.1. Total VOC

Results from the total VOC analyses are summarized in Table 5. Only a few VOC compounds, including trichlorofluoromethane, toluene, 4-isopropyltoluene, trimethylbenzene, xylenes, and methylene chloride, were routinely found in the recovered soil fines samples. Trichlorofluoromethane, a widely used halocarbon aerosol propellant and refrigerant [15], could have resulted from old air conditioning units or refrigerators. Toluene and xylenes are commonly used as solvents for oil varnish, paints, resins, and enamels. They are also common fuel components. The concentrations of VOCs contained in paint thinner, oil stain, and oil varnish range typically from 500 to 700 g/l. Trimethylbenzenes are widely used as solvents in industrial and commercial dyes and varnishes. Methylene chloride is commonly used in solvents, as an ingredient in paint and varnish removers, and as blowing agents in foams [15]. Possible sources of methylene chloride, which were measured at high levels in some samples (Trip 1: Site B and Site C) might have

been solvents and discarded foam sprays. Some methylene chloride may have resulted from laboratory contamination. The chemical was found in some of the laboratory blanks at detectable levels (5.0–10.0 ug/kg).

#### 4.1.2. Leaching of volatile organics

The VOC leaching test using a SPLP ZHE was conducted on samples from Trips 4 and 5. The results of the leaching tests showed a total of nine VOCs out of 50 target volatile organics being detected (Table 6). Among the commonly leached compounds were ethylbenzene, trimethylbenzene, toluene, xylenes, and trichlorofluoromethane. Comparisons between total results and leaching results were made to examine the leachability of each compound. Toluene showed the highest leachability among the compounds (61.3–92.0%), whereas trichlorofluoromethane, the most commonly detected compound from both total and leaching tests, demonstrated the lowest leachability. The range of the leaching percentage of trichlorofluoromethane was from 1.4 to 39.9%.

### 4.2. Semi-VOC analysis

#### 4.2.1. Total semi-volatile organic compounds

Results of the total semi-VOC analyses are shown in Table 7. Of a total of 47 recovered soil fines semi-VOC samples from 14 C&D recycling facilities, no acid extractable compounds, or pesticides were detected with the exception of one acid compound detected from one sample at one site (phenol=7.9 mg/kg). Three base-neutral compounds, bis(2-ethylhexyl)phthalate, butyl benzyl phthalate, and di-n-butyl phthalate were commonly detected in many of the soil fines samples. These compounds are phthalic acid esters (PAE), which are used as plasticizers to enhance the flexibility of polyvinyl chloride (PVC) [15,16]. High concentrations of PAE are found both in plastics and in soft PVC products, such as cables, tubings, toys, or carpets [16]. The probable source of the phthalates was plastics in recovered soil fines samples. In a composition study of recovered soil fines from 12 different C&D recycling facilities in Florida from Trip 3, the percentage of readily identified as materials other than soil (greater than 0.64 cm or 0.25 inch sieve size) was 27% (by mass). Plastics accounted for approximately 0.2% (by mass) of the readily identifiable waste components by average [5].

Additional chemical parameters, PAHs and PCBs, were targeted for the analysis of recovered soil fines samples from Trips 3, 4 and 5. No PCBs were detected in the samples. However, several PAHs, such as acenaphthene, pyrene, fluoranthene, and phenanthrene, were commonly detected in the samples. In a study conducted to examine leachability of organics in building waste using the TCLP, several PAHs were leached from roofing felt [17]. The probable sources of PAHs

Table 5  
Results of total volatile organic compounds (VOC) analysis (unit: µg/kg)

Compounds	Trip 1			Trip 2	Trip 4 <sup>a</sup>			Trip 5 <sup>a</sup>	Florida soil cleanup target levels: direct exposure	
	Site A (12 samples)	Site B (five samples)	Site C (seven samples)	Site D (10 samples)	Site A (one sample)	Site E (one sample)	Site F (one sample)	Site D (six samples)	Residential	Industrial
Carbon tetrachloride	15.3 (1) <sup>b</sup>		11.6–21.2 (4)	27.4 (1)					400	600
Methylene chloride	6.3–10.9 (4)	137.9–3,760 (4)	12.0–3,430 (6)	5.0 (1)					1600	23,000
1,1,2,2-tetrachloroethane	18.7–35.9 (3)								700	1100
tetrachloroethene	21.7–24.4 (2)	9.3–21.3 (2)		27.4 (1)					8900	17,000
1,1,2-trichloroethane	5.8 (1)								1300	1800
1,1-dichloropropene	10.0 (1)		7.2–21.2 (4)						NA	NA
Benzene	11.4–14.3 (2)	16.0 (1)	5.2–12.9 (2)	5.2–20.9 (3)				1100	1600	
Ethylbenzene				13.5–102.4 (7)		350		80–100 (2)	1,100,000	8,400,000
Isopropylbenzene				11.4–391.3 (4)				1,720 (1)	NA	NA
4-isopropyltoluene	12.6–141.2 (4)	23.9–38.8 (2)	11.4–143.0 (5)	18.8–193.4 (7)				240–590 (3)	NA	NA
Naphthalene				55.9–446.1 (9)					40,000	270,000
<i>n</i> -propylbenzene	10.9–47.1 (6)	20.5–56.4 (2)	10.5–57.5 (5)	5.6–9.8 (3)					NA	NA
Styrene	10.4–30.5 (2)	26.3–1,110 (2)	157.7 (1)	7.4 (1)					2,700,000	21,000,000
Toluene	14.6–219.9 (7)	28.5–133.3 (2)	17.8–47.2 (4)	8.0–45.6 (7)		140		220–2170 (4)	380,000	2,600,000
1,2,4-trimethylbenzene	6.9–268.3 (5)	70.1–73.4 (2)	17.7–146.9 (5)	8.7–62.6 (8)		240	103.0	220 (1)	13,000	88,000
1,3,5-trimethylbenzene	22.9–133.9 (7)	78.9–84.4 (2)	19.9–165.3 (6)	6.6–52.7 (8)		420	125.0		11,000	74,000
<i>o</i> -xylene	15.3–366.7 (7)	258.4–541.1 (2)	5.4–24.0 (3)	7.3–78.9 (7)		1,020	105.0	100–180 (2)	5,900,000	4,000,000
<i>m/p</i> -xylene	7.7–56.2 (5)	48.6–124.2 (2)	6.6–11.1 (3)	13.8–74.7 (7)	150	4,150	260	510–720 (3)	5,900,000	4,000,000
Chlorobenzene				6.0–18.9 (3)					30,000	200,000
2-chlorotoluene	30.0 (1)		35.1 (1)						120,000	850,000
4-chlorotoluene	20.38–31.0 (3)		34.5 (1)						100,000	730,000
1,2-dichlorobenzene	17.0 (1)								650,000	4,600,000
1,2,3-trichlorobenzene		92.4 (1)							560,000	7,400,000
trichlorofluoromethane	6.1–198.2 (7)	8.1 (1)		9.3–790.8 (2)	1940	12,850	700	1480–12,180 (4)	200,000	1,300,000

<sup>a</sup> Methanol extraction.

<sup>b</sup> Parenthesis represents the number of detected sample. Note: detection limits for volatile organic analysis from purge and trap method of solids and from methanol extraction method of solids; 5.0 ug/kg and 50 ug/kg.

Table 6  
Comparisons of total volatile organic compounds (VOCs) and leaching VOCs<sup>a</sup>

Compounds	Trip 4									Trip 5			Florida soil cleanup target levels—leaching (ug/kg)
	Site A			Site E			Site F			Site D			
	Total (ug/kg)	Leaching (ug/l)	% Leaching	Total (ug/kg)	Leaching (ug/l)	% Leaching	Total (ug/kg)	Leaching (ug/l)	% Leaching	Total (ug/kg)	Leaching (ug/l)	% Leaching	
Ethylbenzene				350.0	10.5	60.0%		1.0	NA	80–100 (2)	2.4–2.7 (2)	47.8–66.4%	600
Isopropylbenzene										1720 (1)		NA	NA
4-isopropyltoluene										240–590 (3)	2.0–5.2 (3)	46.4–20.1%	NA
Toluene				140.0	4.3	61.3%				440–2170 (4)	20.3–94 (2)	86.7–92.0%	500
1,2,4-Trimethylbenzene				240.0	2.9	24.5%	103.0	1.2	23.9%	220 (1)	2.2 (1)	19.70%	300
1,3,5-Trimethylbenzene				420.0	7.3	34.8%	125.0	2.8	44.3%				300
<i>o</i> -xylene				1020	27.8	54.5%	105.0	2.3	43.1%	100–180 (2)	3.0–5.4 (2)	59.2–60.0%	200
<i>m/p</i> -xylene	150.0	1.0	13.3%	4150	91.0	43.9%	260.0	9.4	72.3%	510–720 (3)	17.4–24.3 (2)	66.4–68.1%	200
Trichlorofluoromethane	1940	1.4	1.4%	12,820	16.9	2.6%	700.0	1.8	5.3%	1480–12,180 (4)	20.5–29.5 (3)	13.1–39.9%	33,000

<sup>a</sup> Note, detection limit for leaching: 20 ug/kg or 1 ug/l.

Table 7  
Results of total semi volatile organic compound analysis

Compounds	Trip 1			Trip 3	Trip 4			Trip 5	Florida soil cleanup target levels: direct exposure (mg/kg)	
	Site A 12 samples (mg/kg)	Site B five samples (mg/kg)	Site C six samples (mg/kg)	12 Sites 12 samples (mg/kg)	Site A two samples (mg/kg)	Site E two samples (mg/kg)	Site F two samples (mg/kg)	Site D six samples (mg/kg)	Residential	Industrial
Acids										
Base/ neutrals										
Phenol		7.9 (1) <sup>a</sup>							900	390,000
Bis(2-ethylhexyl)phthalate	1.3-40.0 (11)	1.6-16.4 (4)	2.6-7.7 (3)	1.1-38.4 (10)				9.8-46.6 (6)	76	280
Butyl benzyl phthalate	1.6-35.6 (12)	2.6-17.6 (3)	4.0-30.1 (3)	1.1-13.3 (5)				1.0-20.5 (6)	15,000	320,000
Di- <i>n</i> -butyl phthalate	1.5-4.0 (9)	2.6 (1)	7.8 (1)					0.4-6.1 (6)	7300	140,000
Polycyclic aroatic hydrocarbons										
Acenaphthene				8.5-73.3 (2)		0.7 (1)		0.3-42.7 (5)	1900	18,000
Indeno [1,2,3-cd] pyrene				27.1 (1)					1.5	5.3
Pyrene				1.8-29.6 (9)	2.9 (1)	0.5-0.9 (2)	5.1-9.0 (2)	6.4-18.6 (6)	2200	137,000
Benzo [ghi] perylene				17.2 (1)					2300	41,000
Benzo [a] pyrene				55.1 (1)					0.1	0.5
Fluoranthene				1.3-225.1 (8)	1.3 (1)	0.6-0.7 (2)	3.9-11.5 (2)	11.5-68.4 (6)	2900	48,000
Phenanthrene				2.9-45.3 (6)	2.9 (1)	0.4-1.4 (2)	8.4-12.8 (2)	3.9-32.6 (6)	2000	30,000

<sup>a</sup> Parenthesis represents the number of detected samples. Note; detection limit for semi-volatile organics including polychlorinated biphenyl and pesticides in solids: 0.3 mg/kg.

Table 8  
Comparisons of total semi-volatile organic compounds (VOC) analysis and leaching semi-VOC analysis

Compounds	Trip 5 (site D)			Florida soil cleanup target levels—leaching (mg/kg)
	Total six samples (mg/kg)	leaching 6 samples (ug/l)	%Leaching	
Base/neutrals				
Bis(2-ethylhexyl)phthalate	9.8–46.6 (6) <sup>a</sup>	6.0–280	0.3–0.8%	3600
Butyl benzyl phthalate	1.0–20.5 (6)	3.0–110	0.1–10.0%	310
Di- <i>n</i> -butyl phthalate	0.4–6.1 (6)	8.0–180	1.5–25.0%	47
Polycyclic aromatic hydrocarbons				
Acenaphthene	0.3–42.7 (5)	1.0–6.0	0.0–5.0%	2.1
Pyrene	6.4–18.6 (6)	1.0–3.0	0.0–0.5%	880
Fluoranthene	11.5–68.4 (6)	1.0–3.0	0.0–0.3%	1200
Phenanthrene	3.9–32.6 (6)	3.0–9.0	0.2–2.3%	250

<sup>a</sup> Parenthesis represents the number of detected samples Note; detection limit for semi-volatile organics including polychlorinated biphenyl and pesticides for leaching: 0.01 mg/kg or 2 ug/l.

Table 9  
Organic compounds in recovered soil fines exceeding soil cleanup target levels for direct exposure

Compounds	Concentration range (mg/kg)	Number of exceeded	Direct exposure (mg/kg)	
			Residential	Industrial
Indeno[1,2,3- <i>cd</i> ]pyrene	27.1	1	1.5	5.2
Benzo( <i>a</i> )pyrene	55.1	1	0.1	0.5

might be roofing felt, asphalt roofing products, petroleum distillates from disposed paint thinner and other petroleum products such as oil filters, waste oil and greases, and fuel spills.

#### 4.2.2. Leaching of semi-volatile organics

The semi-VOC leaching tests were performed on samples from Trips 4 and 5. No semi-VOCs from a total

of six samples from the sampling Trip 4 were leached, despite the fact that several PAHs (acenaphthene, pyrene, fluoranthene, and phenanthrene) were detected in total analysis from Trip 4. Several phthalates and PAHs from Trip 5 samples leached, however (Table 8). These compounds were detected in total analysis of Trip 5. Leachability ranged from 0.1 to 25% and from 0 to 5% for phthalates and PAHs, respectively. The low leach-

Table 10  
Organic compounds in recovered soil fines exceeding soil cleanup target levels for leaching

Compounds	Concentration range of exceeded samples (mg/kg)	Number of exceeded	Leachability based on groundwater criteria (mg/kg) <sup>a</sup>
<i>Total volatile organic compounds (VOC)</i>			
Benzene	0.008–0.021	6	0.007
Isopropyl benzene	0.391–1.72	2	0.2
Methylene chloride	0.034–3.76	8	0.02
1,1,2,2 Tetrachloroethane	0.019–0.036	3	0.002
1,2,4 Trimethylbenzene	0.220–0.268	3	0.3
1,3,5 Trimethylbenzene	0.420	1	0.3
Xylenes	0.343–4.15	7	0.2
<i>Total semi-VOC</i>			
Bis(2-ethylhexyl) phthalate	33.5–46.6	0	3600
Phenol	7.9	1	0.05
Acenaphthene	8.5–42.7	4	2.1
Benzo( <i>a</i> )pyrene	55.1	1	8.0

<sup>a</sup> Florida soil cleanup targets levels [11]. The values were calculated using generic assumptions regarding exposure and soil characteristics and are intended to be applicable to sites in Florida. The equation described earlier in Section 2.3 was used to determine soil cleanup target levels for leaching in Table 10. The variables and assumed values in the equation are as follow:  $C_w K_{oc}$ ,  $H'$ — chemical specific,  $f_{oc} = 0.002$ (g/g),  $\theta_w = 0.3$ ,  $\theta_a = 0.13$ ,  $\rho_b = 1.5$  g/cm<sup>3</sup>

ability of PAHs under the SPLP test results primarily from their low aqueous solubilities and vapor pressures. Because of their hydrophobic nature, PAHs display a tendency to partition onto the solid phase (recovered soil fines) rather than the liquid phase (SPLP solution).

#### *4.3. Risk assessment and organics in recovered soil fines*

The results of this study were used to assess the risk from land applied recovered soil fines by comparing the results to Florida's generic SCTLs (total concentration) and Florida's GWCTLs (leaching concentration). For the leaching risk assessment, two methods were available; a comparison of the SPLP leaching results (mg/l) with applicable groundwater standards and a comparison of total concentrations (mg/kg) with risk-based SCTLs.

Table 9 presents the organic compounds measured in the recovered soil fines that exceeded SCTLs for direct exposure in Florida. In this study, two PAH compounds (indeno[1,2,3-cd]pyrene and benzo(a)pyrene) were detected at concentrations above the direct human exposure SCTLs. This observation was made for only one sample. The total concentrations of the other organic compounds detected were below the SCTLs for direct exposure.

Table 10 summarizes a list of organic compounds that exceeded SCTLs for leaching to groundwater. The leaching SCTL comparison indicates that these chemicals may be of concern in regard to groundwater contamination. However, no organic compounds evaluated during the SPLP leaching tests exceeded the GWCTLs. This observation may have a number of causes. One reason for this may be the relatively high organic carbon content in the recovered soil fines (2.0–5.0%) [5]. A 0.2% organic content is used as an EPA default value to calculate the organic leaching goals in the equation. The lower leachability is explained by a greater affinity of hydrophobic compounds to bind with organic carbon rather than enter the aqueous phase (or SPLP leaching solution). This is especially true for PAHs. A second reason stems from differences between actual leaching tests and theoretical calculations.

Based on these results, the Florida Department of Environment Protection (FDEP) concluded that the organic chemicals in C&D soil fines were not a major concern with regard to direct exposure and leaching to groundwater. They found that heavy metals, primarily arsenic in the fines, were a greater risk. The guidelines that were developed for the management of recovered screen material from C&D debris recycling facilities in Florida therefore included more frequent monitoring for metals such as arsenic, and less frequent monitoring for organic chemicals [18].

## **5. Conclusions**

Over recent years, the recycling of C&D materials has been a growing industry in the USA. C&D material recycling facilities separate mixed C&D debris using a series of mechanical operations, manual labor, or a combination of both. During recycling processes screened fines represent one of the major recovered materials at C&D debris processing facilities. Several reuse options have been proposed for the recovered screened materials, including fill material in roads and soil amendment. Before applications where the material is placed in direct contact with the environment, the C&D material must be adequately characterized for chemical composition.

Recovered fines samples from a total of 14 C&D waste recycling facilities in Florida were collected over a period of 18 months and then analyzed for organic pollutants (volatile and semi-volatile organics). Both total extractable organic analysis and leaching analysis were conducted to chemically characterize the recovered soil fines. For leaching analysis a synthetic precipitation leaching procedure (SPLP) test was performed to evaluate the leachability of organics.

Only a few volatile organic compounds (VOCs) out of 50 target VOCs, including trichlorofluoromethane, toluene, 4-isopropyltoluene, trimethylbenzene, xylenes, and methylene chloride, were commonly found in the samples. Among the commonly leached compounds were ethylbenzene, trimethylbenzene, toluene, xylenes, and trichlorofluoromethane. Toluene showed the highest leachability among the compounds (61.3–92.0%), while trichlorofluoromethane, the most commonly detected compound from both total and leaching tests, resulted in the lowest leachability (1.4–39.9%). Three base-neutral semi-VOCs (bis(2-ethylhexyl)phthalate, butyl benzyl phthalate, and di-n-butyl phthalate) and several PAHs (acenaphthene, pyrene, fluoranthene, and phenanthrene) were commonly detected in many samples. These compounds also leached during the leaching test. However, the range of percent leaching from the compounds was from only 0.1 up to 25%.

This study demonstrates the use of generic risk-based soil screening levels. It also shows that leaching tests are warranted because they consider specific soil/media interaction with a leaching solution. Based on this study's results, the organic chemicals in recovered soil fines from C&D debris recycling facilities were not of a major concern in terms of human exposure and leaching to groundwater under reuse and contact scenarios.

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ATTACHMENT D



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## Heavy metals in recovered fines from construction and demolition debris recycling facilities in Florida

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### Abstract

A major product recovered from the processing and recycling of construction and demolition (C&D) debris is screened soil, also referred to as fines. A proposed reuse option for C&D debris fines is fill material, typically in construction projects as a substitute for natural soil. Waste material that is reused in a manner similar to soil must first be characterized to examine potential risks to human health and the environment. In Florida, samples of C&D debris fines from 13 C&D debris recycling facilities were characterized for 11 total and leachable heavy metal concentrations. Total metal concentration results (mg/kg) were compared to existing data on background Florida soil concentrations and to Florida's risk-based soil cleanup target levels (SCTLs). All of the detected metals were found to be elevated with respect to background. The 95% upper confidence level of arsenic from 99 samples was 3.2 mg/kg; arsenic presented the greatest limitation to reuse when compared to the SCTLs. Lead was not found to pose a major problem, likely because of the relatively new building infrastructure in Florida, which results in less demolition debris and less material impacted by lead-based paint. The results of batch leaching tests conducted using simulated rainwater (mg/l) were compared directly to risk-based groundwater levels for Florida and were found not to pose a risk using existing risk assessment policies.

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*Keywords:* Construction debris; Demolition debris; C&D debris; Soil fines; Recycling; Arsenic; Heavy metals; Leaching; SPLP; Recovered fines

### 1. Introduction

Construction and demolition (C&D) debris represents one of the largest components of solid waste generated from municipal activities. Major components of C&D debris include wood, concrete (including masonry products), asphalt (pavement and

roofing), gypsum wallboard, cardboard, metal, soil, rock and vegetative debris (from land clearing). A US Environmental Protection Agency (EPA) sponsored study estimated that 127 million metric tons of building-related C&D debris were generated in the US in 1996 (US EPA, 1998a). While C&D debris has often been managed by landfill disposal, recovery, processing and recycling of C&D debris components have become standard practice in some locations. In Florida, where construction continues to be very active, C&D debris comprised almost

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one-fourth of the weight of the municipal solid waste (MSW) stream in 1998, with a reported 5.3 million metric tons collected (FDEP, 2000). Since C&D debris represents a significant portion of the waste stream, it is often targeted for waste reduction and recycling. Several factors dictate the success of C&D debris recycling including landfill tipping fees, the magnitude of C&D activity occurring in an area, and viability of reuse markets for recovered C&D debris components.

Although some separation of C&D debris components may occur at the construction or demolition site, most mixed C&D debris that is recycled is separated and recovered at central processing facilities (Townsend, 1998). Central processing operations utilize a combination of mechanical equipment (e.g. conveyor belts, float tanks, trommel screens, magnets) and manual methods (e.g. hand picking of specific materials) to separate mixed C&D debris into a number of reusable commodities. Typical operations utilize a screen to remove soil and other fine materials prior to subsequent processing. While wood, concrete, and metal are common C&D debris materials targeted for recovery, a large percentage of the recovered mass is comprised of fines.

While C&D debris for the most part is inert, it may contain some materials with the potential to harm human health and the environment (US EPA, 1998a). Because of concerns over possible risks posed by C&D debris fines when recycled through land application, a study was initiated to characterize and assess the magnitude and range of concentrations of chemicals in recovered C&D debris fines. Organic compounds (Jang and Townsend, 2001a), sulfate leachability (Jang and Townsend, 2001b), and heavy metals were examined. This paper reports the results of the heavy metal characterization of the C&D debris fines in Florida. The total and leachable concentrations of 11 metals were measured. The total metal concentrations (mg/kg) were compared to Florida's SCTLs that are regularly used for the assessment of contaminated sites. The leachable metal concentrations ( $\mu\text{g/l}$ ) were compared directly to Florida's risk-based groundwater cleanup target levels (GWCTLs) (Saranko et al., 1999). The results are not only of interest to those dealing with C&D debris processing facilities, but also provide an example of the application of generic risk-based target levels or guidance

concentrations for waste materials reused in a beneficial manner.

## 2. Background

### 2.1. Recovered fines from C&D debris recycling

The processing of C&D debris has evolved over recent years. Demolition contractors have historically processed waste materials for the recovery of concrete and metal, but modern C&D debris recyclers accept commingled and mixed loads of C&D debris for processing and recovery. A variety of approaches and unit operations are employed at C&D debris processing facilities. Some facilities process waste up-front using compactors or impactors to reduce particle size, which allows for efficient mechanical separation. Other facilities minimize size-reduction processes to allow for easier manual separation of bulk materials (Townsend, 1998).

In both processing scenarios, screening is an essential element of the separation and recovery process. Mixed C&D debris often contains large amounts of soil from the construction or demolition site. Large screening devices, such as trommel screens, bar screens, or vibrating screens, separate fine materials from the bulk waste. Typical screen sizes range from 0.6 to 5 cm (0.25 to 2 inches). Fig. 1 presents a typical layout of a C&D debris processing facility, including the production of C&D debris fines. Recovered C&D debris fines have been found to make up 20% or more of the mass of recovered material at C&D debris processing facilities (Townsend, 1998). In addition to soil, C&D debris fines also contain other waste materials, including small pieces of wood, concrete, brick, asphalt (pavement and shingles), gypsum dry-wall, paper, and plastic. The composition of recovered C&D debris fines is a function of the separation process (mechanical or manual) and the types of waste processed. The objective of C&D debris processing facilities is to recycle the maximum amount of materials from the waste stream; thus the disposal of the recovered fines in a landfill is avoided if at all possible. Several reuse and recycling options for the recovered C&D debris fines have been considered; daily cover at landfills is one practiced option. Since C&D debris fines contain primarily soil and are soil-

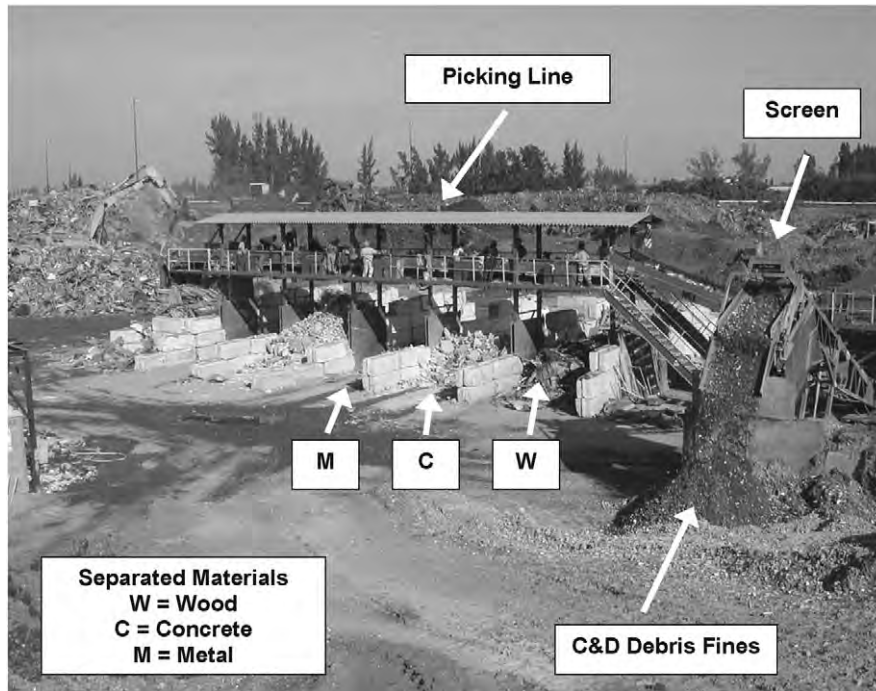


Fig. 1. Typical C&D debris recycling operation. Production of C&D debris fines is indicated.

like in appearance, the possibility of beneficial use as a soil substitute or as a fill material has been proposed. The preferred choice by many facility operators is utilization as construction fill (e.g. embankments, berms). In this manner, the recovered C&D debris fines can be returned directly to the same site (or a similar area) as they were generated.

## 2.2. Heavy metals in construction and demolition debris

Recovered C&D debris fines can become contaminated with heavy metals in three primary ways: (1) the contamination may come from the soil in the C&D debris stream itself; (2) from small pieces of hazardous building materials (e.g. treated or painted wood); and/or (3) from leaching of hazardous materials commingled with the waste stream.

One source of heavy metal contamination is the soil in the C&D debris itself. If demolition or construction takes place at sites previously used for residential, commercial, or industrial facilities, the on-site soil may already contain elevated levels of

heavy metals from past use. Sources of contamination might include spills from historical chemical handling or storage at the site, application of pesticides, or atmospheric deposition from local industries. For example, elevated lead concentrations have been documented in soils adjacent to structures with exterior lead-based paint (US EPA, 1998b).

Heavy metals in C&D debris fines may also result from small pieces of heavy metal-containing debris. For example, paint chips or dust associated with lead-based paint debris may be commingled with C&D debris. The EPA defines lead-based paint as paint containing 0.5% lead (500 mg/kg) (FR, 2001), however concentrations can be much higher. In a study of wood chip fines from several C&D debris processing facilities operating in the northeast United States, lead was encountered at levels exceeding state guidelines (Beebe and England, 1998). Mercury-containing fluorescent lamps and high intensity discharge lamps in C&D debris would likely be crushed to sizes small enough to remain in the fines as well. Chromated copper arsenate (CCA) treated wood typically contains several thousand

mg/kg arsenic and chromium (Tolaymat et al., 2000); therefore, the presence of sawdust or CCA-treated wood fragments could impact the overall concentrations of these metals in the fines (Lebow, 1996).

While hazardous materials should be removed prior to a building's demolition (Sheridan et al., 2000), in many cases hazardous materials remain and often become commingled with demolition debris. The act of demolition can also expose some chemicals that were otherwise contained (e.g. batteries in exit signs and emergency lighting). Because debris is often stockpiled for a period of time before processing, components in the pile that would otherwise be removed in subsequent processing have the potential to leach and contaminate the surrounding soil. Damaged batteries may leach chemicals such as lead, nickel and cadmium. Lead flashing and sheeting may leach lead, and painted wood could leach a variety of metals. Even if CCA-treated wood is later removed, it may leach copper, chromium and arsenic into the surrounding soil and debris when stockpiled for an extended period. Even under normal use (decks, fences) CCA-treated wood leaches arsenic, chromium and copper into underlying soil (Stilwell and Gorny, 1997) and when it is a size-reduced, it leaches at even greater levels (Townsend et al., 2001).

### 3. Methodology

Recovered C&D debris fines were sampled from 13 C&D debris processing facilities in Florida over a period of 14 months. Researchers conducted five sampling events; different facilities were sampled in each sampling event, though some facilities were visited separately during two different events [see Townsend et al. (1998) for more details]. The total recoverable concentrations of selected heavy metals (mg/kg) were measured in every sample; leachable metal concentrations (mg/l) were only measured on a subset of these samples. Field sampling was performed following standardized procedures published by the state regulatory agency (FDEP, 1992). Laboratory analysis was performed following standard procedures for characterizing solid wastes (US EPA, 1996).

#### 3.1. Sample collection

In the first two sampling rounds, stockpiled fines were sampled by a random grid method. In the last three sample rounds, piles of fresh C&D debris fines were divided into four sections, and each quadrant was sampled. The first 5 cm of C&D debris fines at each location were removed to expose the sampling area. Two to four kg of C&D debris fines were placed in a stainless steel bowl and mixed thoroughly. Approximately 0.5 to 1 kg of fines were placed in either glass jars or polyethylene containers, stored below 4 °C, and transported to a walk-in cooler prior to digestion and analysis. In accordance with the laboratory's quality assurance (QA) plan, trip, field, and equipment blanks, as well as field sample duplicates, were collected and analyzed.

#### 3.2. Sample preparation and analysis

Eleven metals were targeted for total concentration analysis (aluminum, arsenic, cadmium, chromium, copper, mercury, nickel, lead, selenium, silver and zinc). Not every metal was analyzed in each sample; see Townsend et al. (1998) for details regarding which metals were analyzed on which samples. The digestion method used for most metals was EPA method 3050B. This method involves weighing 1–2 g of C&D debris fines into a glass flask and digesting them using an open-vessel hot plate acid digestion procedure with both nitric and hydrochloric acid followed by hydrogen peroxide addition. Silver was digested following method 7760A. After digestion, samples were analyzed using a Perkin–Elmer 5100 flame atomic absorption spectrophotometer (AAS). Some metals, such as arsenic and selenium, were analyzed using a Perkin–Elmer 5100 graphite furnace AAS equipped with Zeeman background correction. Samples were digested for mercury and analyzed following EPA method 7471A (US EPA, 1996); a Perkin–Elmer MHS-10 manual cold vapor apparatus was used. In all cases, samples were digested and analyzed at their original moisture content; results were converted to and reported on a dry weight basis using data from independent moisture measurements.

Samples from the third sampling round were leached and the leachates were analyzed for all metals

discussed above, except for mercury. Because of quality control concerns with chromium results from round 3, these results were discarded. Samples from rounds 4 and 5 were leached and analyzed for chromium and mercury. The leaching test employed was the SPLP, EPA method 1312 (US EPA, 1996). This procedure involved the addition of simulated acid rainwater (pH of  $4.20 \pm 0.05$ ) to the solid matrix at a 20 to 1 liquid–solid ratio. The SPLP simulated rainfall is made using 60/40 percent by weight mixture of nitric and sulfuric acid; no buffering capacity is provided by these strong mineral acids. One hundred grams of C&D debris fines were placed in a 2 l plastic container and 2 l of SPLP solution were added. The slurry was agitated in a rotary extractor (31 rpm) for  $18 \pm 2$  h. After filtration, samples were digested using EPA method 3010 (US EPA, 1996) and analyzed using the methods described for the total recoverable analysis (arsenic was digested using EPA method 7076A). The SPLP leachates were analyzed for mercury using EPA method 7470A (US EPA, 1996) with the same apparatus used for the solid samples.

## 4. Results and discussion

### 4.1. Total metals

Nine of the eleven metals analyzed (aluminum, arsenic, cadmium, chromium, copper, mercury, nickel, lead and zinc) for total recoverable concentrations (mg/kg) were detected in some samples above the detection limit. Silver and selenium were not detected in any of the samples above detection limits. For purposes of summarizing the data and statistical analysis, sample measurements that were below detection limit were treated as equal to one-half of the detection limit. To determine whether the data were normally or log-normally distributed, several statistical procedures were used (US EPA, 2000). Each sample collected from every sampling event was included in the analysis (results from field duplicates were averaged and treated as one sample). If the number of samples analyzed was less than 50, the Shapiro–Wilk *W*-test (Shapiro and Wilk, 1965) was performed. If the number of samples was greater than 50, the Filliben statistic was used (Filliben, 1975). The concentrations of aluminum and chromium were

found to be normally distributed. The concentrations of arsenic, copper, lead, nickel and zinc were found to be log-normally distributed. The distribution of aluminum and chromium suggests that these elements are more ubiquitous in the soils that make up the bulk of C&D debris fines, while the other elements are more sensitive to the particular source of debris. Mercury and cadmium were not observed to be normally or log-normally distributed using the tests performed. This might be the result of the relatively small number of samples analyzed in the case of mercury, or the small number of detected concentrations in the case of cadmium. Statistical data for both distributions are presented in Table 1. Since the majority of the elements were log-normally distributed and because it is common for environmental data involving trace pollutants to be log-normally distributed, mercury and cadmium were also assumed to be distributed as such for statistical comparison purposes.

Table 1 summarizes the total recoverable metal concentrations. Both the arithmetic and geometric means and standard deviations are reported. Statistics reported in other sections of this paper correspond to the respective distributions (normal or lognormal) discussed above. Fig. 2 presents histograms for all of the detected metals except mercury. Aluminum was found to have the highest mean concentration of all metals measured (3100 mg/kg), followed by zinc (200 mg/kg), and lead (66 mg/kg). The metals with the smallest mean concentrations were arsenic (2.7 mg/kg), cadmium (1.1 mg/kg) and mercury (0.16 mg/kg). Cadmium was detected in only 39 out of 81 samples.

### 4.2. Leachable metals

Table 2 presents the SPLP leaching results. Many of the elements were not detected or were detected very infrequently. Only the arithmetic mean and standard deviation of the detected concentrations are reported. Because none of the leaching results were found to exceed risk-based groundwater thresholds (to be discussed later in this paper), a more rigorous statistical analysis was not performed. The concentrations of lead, selenium, and silver were below their respective detection limits of 10, 5.0, and 50  $\mu\text{g/l}$  in every sample. Aluminum, arsenic and zinc were the only metals detected in more than half the SPLP samples, leaching

Table 1  
 Total metal concentrations in Florida C&D debris fines (mg/kg)

Metal	Number detected/ Number analyzed	Detection limit (mg/kg)	Metal concentration (mg/kg) <sup>a</sup>			
			Mean <sup>b</sup> ± S.D. <sup>c</sup>	Geo mean <sup>d</sup> ± S.D. <sup>e</sup>	Max	Min
Al	47/47	50	3100 ± 600	3000 ± 1.2	4300	1400
As	96/99	0.3	4.4 ± 6.1	2.7 ± 2.6	51	<0.5
Cd	39/81	0.5	2.0 ± 1.7	1.1 ± 3.6	5.6	<0.25
Cr	52/52	5.0	21 ± 13	18 ± 1.6	98	5.0
Cu	65/65	5.0	50 ± 60	34 ± 1.9	420	7.2
Pb	98/99	5.0	92 ± 111	66 ± 2.3	1000	<50
Hg	16/16	0.002	0.33 ± 0.6	0.16 ± 3.1	2.5	0.023
Ni	65/65	5.0	76 ± 62	51 ± 1.7	300	6.8
Zn	81/81	0.5	290 ± 381	200 ± 2.0	3100	27

<sup>a</sup> Half the detection limit is used for samples with concentrations below detection limits.

<sup>b</sup> Arithmetic mean.

<sup>c</sup> Arithmetic S.D.

<sup>d</sup> Geometric mean.

<sup>e</sup> Geometric S.D.

83%, 76% and 57% of the time, respectively. Mercury was detected (1.1 µg/l) in only one sample out of 7 analyzed. The pH of the leaching solution likely contributed to the relatively low metal leachability. Although the initial pH of the SPLP solution was 4.2, the final slurry pH (after 18 h contact time) ranged from 6.4 to 10.4. Most metals tend to be least soluble in this pH range (van der Sloot et al., 1997). Several components of C&D debris fines would contribute alkalinity to the samples; most notable would be unreacted cement or concrete dust, which are certainly present to some extent in a matrix of this type.

For arsenic, aluminum and zinc, samples with leachable concentrations above the detection limit were compared to the total recoverable metal concentrations of those same samples. The average fraction leached for the three metals was calculated. Arsenic leached to the greatest extent, leaching an average of 6.5% in the SPLP when compared to the total measured arsenic in the fines. Aluminum leached 0.13% of the total available metal concentration into solution while zinc leached 0.8%.

#### 4.3. Comparison to background soil concentrations

Since the C&D debris fines represented a composite of multiple sources and multiple locations, it was not possible to sample any specific sites to determine background soil concentrations. One objective of the

research, however, was to determine whether the metal concentrations in the C&D debris fines were elevated with respect to naturally occurring soils in Florida. Thus, the total recoverable metal concentrations were compared to existing data on the naturally occurring background concentrations of metals in Florida soils. Chen et al. (1999) reported the results of 448 soil samples that were collected from throughout Florida; they reported both the arithmetic and geometric means and standard deviations. To determine the appropriate statistical test to perform, the variance of each set of samples (the C&D debris fines and the reported Florida data) were compared using the *F*-test for the equality of two variances. The variances were determined not to be equal, so an unequal variance *t*-test was used to compare the two sample sets (US EPA, 2000).

All nine of the metals that were detected in the C&D debris fines were found to be statistically greater ( $\alpha=0.05$ ) than the metals concentrations reported by Chen et al. (1999). As described earlier, several sources could account for elevated metals in soil fines screened out of loads of C&D debris. As a result of the composite nature of the C&D debris fines collected, it is not possible to distinguish the source of metals in the samples. One interesting observation that will be further elaborated upon when the results are compared to risk-based clean soil concentrations is that lead concentrations are not as high as has been

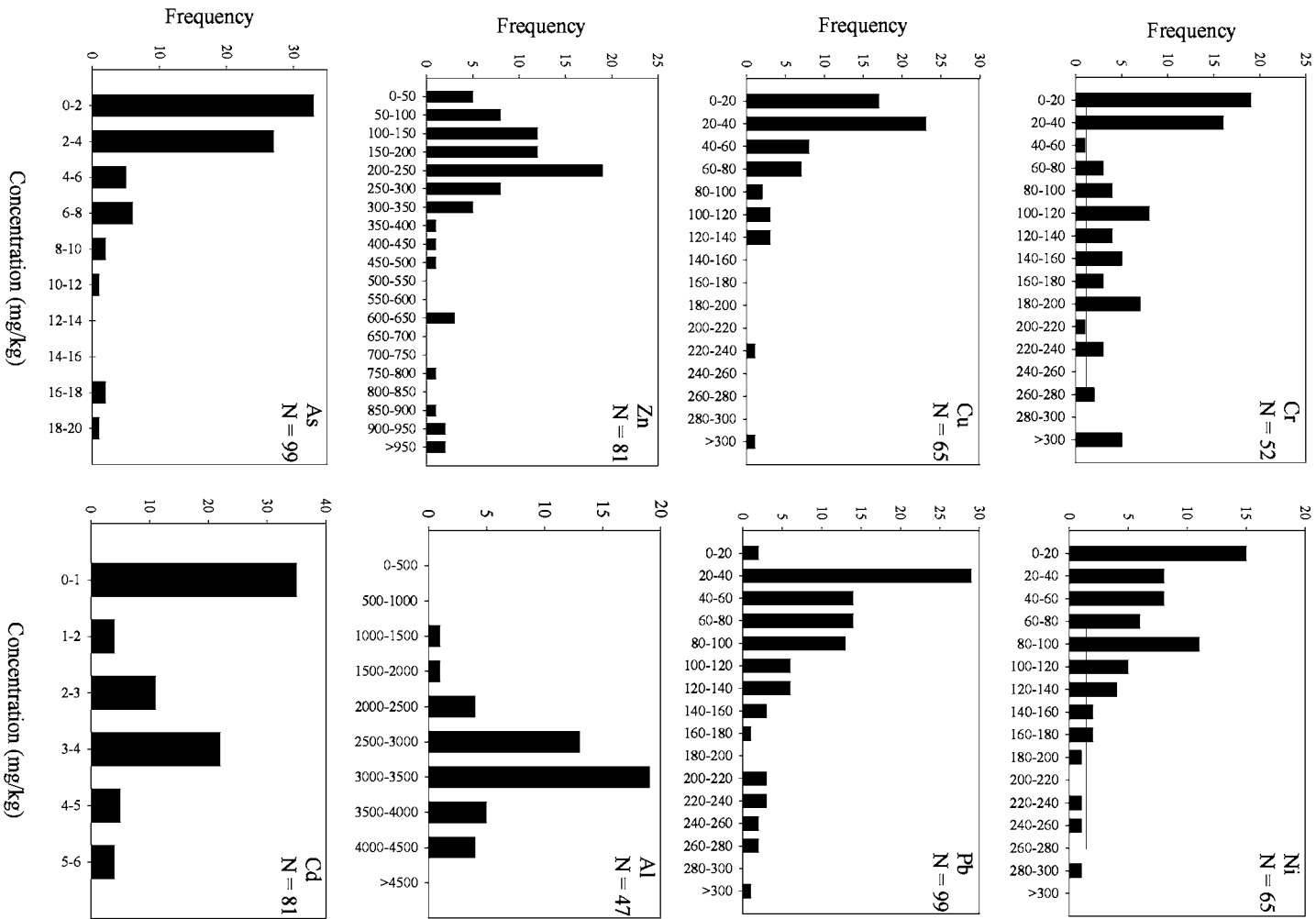


Fig. 2. C&D debris fines metal distributions histograms.

Table 2  
 C&D debris fines metal leaching concentrations (mg/l)

Metal	Number detected/ Number analyzed	Detection limit	Mean <sup>a,b</sup> ± S.D. <sup>c</sup>	Max
Al	38/46	20	52 ± 36	180
As	35/46	5.0	10 ± 1.8	39
Cd	3/46	1.0	2.0 ± 1.4	3.1
Cr	8/18	5.0	14 ± 15	50
Cu	4/46	100	120 ± 13	130
Hg	1/7	0.13	1.1	1.1
Ni	3/46	10	14 ± 3.3	16
Zn	25/46	50	80 ± 20	120

<sup>a</sup> Detected concentrations only.

<sup>b</sup> Arithmetic mean.

<sup>c</sup> Arithmetic S.D.

reported in the limited data on C&D debris fines from other areas of the US; Beebe and England (1998) reported that lead was the most limiting chemical in C&D debris wood fines from the Northeast US This is likely a result of the fact that Florida C&D debris is much more represented by construction debris, as opposed to demolition debris. Florida's infrastructure is much younger than other areas and thus lead is less likely to be encountered. The Chen et al. data Florida soil baseline concentrations (geometric means) are presented in Table 3 along with Florida's risk-based target levels for comparison purposes.

#### 4.4. Comparison to Florida's risk-based target levels

The total and leachable sample results were compared to risk-based guidelines for clean soil and

groundwater developed by the Florida Department of Environmental Protection (FDEP, 2000). The total recoverable concentrations were compared to the SCTLs and the SPLP leaching concentrations were compared to the GWCTLs.

The SCTLs were developed to assess the risk associated with direct human contact with a soil (or similar matrix) and the risk presented by the leaching of chemicals from that soil to groundwater. Table 3 presents three types of SCTLs for the metals detected in the C&D debris fines: residential and industrial direct human exposure SCTLs and a leaching-to-groundwater SCTL. For several of the metals, no leaching-to-groundwater SCTL was available; in these cases the regulations say that a SPLP must be performed (FAC, 2000). Also presented in Table 3 are the 95% upper confidence levels (UCL) for the detected metals. The 95% UCL represents a conservative estimate of the mean and is often used when comparing results to regulatory benchmarks or standards (US EPA, 2000). The 95% UCL for each element was calculated using the mean and standard deviation presented in Table 1 (for the distribution determined appropriate for each element). Arsenic was the only element with a 95% UCL (3.2 mg/kg) above the residential SCTL (0.8 mg/kg). When all samples were considered, arsenic concentrations exceeded the residential SCTL in 93% of the samples and exceeded the industrial SCTL (3.7 mg/kg) in 34% of the samples. C&D debris fines samples exceeded the residential SCTL for nickel in 14% of all of the samples, while chromium and copper concentrations exceeded their

Table 3  
 Comparison of total metal concentrations in Florida C&D debris fines to Florida's SCTLs and soil baseline concentrations

Metal	C&D debris Fines UCL <sup>a</sup> (mg/kg-dry)	Florida's SCTLs (mg/kg)			FL soils (mg/kg) <sup>b</sup>
		Residential	Industrial	Leaching	
Al	3300	72 000	NA	NA	NA
As	3.2	0.8	3.7	29	0.42
Cd	1.3	75	1300	8	0.01
Cr	24	210	420	38	8.45
Cu	41	110	76 000	NA	2.21
Hg	0.25	3.7	26	2.1	5.45 × 10 <sup>-3</sup>
Pb	75	400	920	NA	5.38
Ni	63	110	28 000	130	9.08
Zn	200	23 000	560 000	6000	5.12

<sup>a</sup> Half the detection limit is used for samples with concentrations below detection limits.

<sup>b</sup> The geometric mean of FL soil baseline concentrations as reported by Chen et al., 1999.

respective residential SCTLs in 11% and 7% of the samples.

Two different methods are typically used to assess the potential risk of a contaminated soil or land-applied waste to contaminate groundwater. If total concentrations (mg/kg) are available, they can be compared to SCTLs derived for the leaching-to-groundwater pathway. The leaching-to-groundwater SCTLs developed by the regulatory agencies (Saranko et al., 1999; US EPA, 1998b) were calculated by assuming a metal partition coefficient ( $k_d$ ) and back-calculating the total metal concentration that must be present for that metal to leach at the appropriate GWCTL. A dilution factor is sometimes applied to account for dilution and attenuation that may occur in the subsurface before contamination reaches the compliance point of interest (Saranko et al., 1999). When the C&D debris fines total concentrations were compared to the leaching-to-groundwater SCTLs, the 95% UCLs for all of the metals were below their respective GWCTLs. However, 46% and 9% of the individual samples exceeded the leaching SCTL for chromium and nickel, respectively. The apparent leaching risk posed by chromium when using this methodology is likely a result of the assumptions used to derive the leaching-to-groundwater SCTL for chromium, which is based on hexavalent chromium. While speciation was not performed to determine the form of chromium in the C&D debris fines samples (hexavalent vs. trivalent), it is very unlikely that any of the chromium in the samples was in the hexavalent form. Hexavalent chromium, which is generally more mobile in the environment, exists only under alkaline (primarily above a pH of 10) and oxidizing conditions; it is rarely encountered in the natural environment.

Evaluating the risk to groundwater using the above approach is only valid if the element's leaching characteristics are similar to that predicted by the assumed partition coefficient. Because of the many different forms that a metal may occur as in waste products, a preferred approach is to evaluate leaching risk using a leaching test. In Florida and many other states, leaching risk is evaluated by conducting the SPLP and comparing the results to risk-based groundwater standards or guidelines (Saranko et al., 1999). While a laboratory leaching test certainly has an advantage over using assumed leaching characteristics, this approach also has certain limitations that should be

considered when evaluating the data. One dilemma faced by the risk assessor is determining what the SPLP leaching concentrations truly represent. If they represent the pore water concentration that would be expected to occur, then the application of a dilution factor might be appropriate. In some cases, the SPLP leachate concentrations may represent a diluted pore water concentration, and thus the application of dilution factor might underestimate potential risk. The chemical environment in a layer of land applied waste (such as in an embankment) may also differ than that predicted in a short term batch leaching test such as the SPLP. The development of reducing conditions, for example, could impact metal mobility. A more preferred approach to address these concerns would be to simulate leaching using a column test, but the cost and time associated with such procedures precludes their use for routine regulatory applications.

When the C&D debris fines SPLP results were compared to Florida's GWCTLs, no single leaching sample ever exceeded a GWCTL for any metal. While chromium was found to present a potential leaching risk when comparing the total concentrations to the leaching-to-groundwater SCTLs, no such leaching risk was predicted using the SPLP. Another item worth noting is the new US drinking water standard for arsenic was recently lowered from 50  $\mu\text{g/l}$  to 10  $\mu\text{g/l}$ ; all drinking water facilities must be in compliance with this new standard by 2006. The current GWCTL is based on the previous drinking water standard and in all likelihood the FDEP will follow suit and lower the arsenic GWCTL. This should also result in a lowering of the leaching-to-groundwater SCTL (from 29 mg/kg to 5.8 mg/kg). When the C&D debris fines concentrations (both the total concentrations and in the SPLP leachate) are compared to the possible future arsenic risk-based target levels, arsenic concentrations do exceed in a number of cases. Therefore, the risk posed by arsenic leaching may sometimes limit reuse options. If the potential for arsenic leaching does turn out to limit reuse, additional leaching procedures such as column tests should be considered to better assess risks.

#### *4.5. Management issues*

C&D debris fines are a large component of materials recovered by C&D debris recycling facil-



ity operators (perhaps not by volume, but certainly by mass). The successful economic performance of such facilities is dependent on minimizing the amount of waste received that must be disposed of in a landfill. The results of this study found that with respect to heavy metals, arsenic most frequently exceeded the risk-based target levels and thus was the element most likely to limit reuse. Unlike results reported for fines from C&D debris processing activities in other US states (Beebe and England, 1998), lead concentrations did not limit reuse. The 95% UCL for arsenic of 99 C&D debris fines samples was 3.2 mg/kg. The residential risk-based SCTL for arsenic is currently 0.8 mg/kg in Florida. While the residential arsenic SCTL is relatively low (with respect to natural background concentrations), it is in line with many other states estimates (AEHS, 1998). The C&D debris fines were more suited to application in commercial or industrial settings where the industrial SCTL would be applied. The difficulty with this, however, is ensuring that the sites where the material is used stay industrial, and do not get changed (e.g. rezoned) to residential land use in the future. Deed restrictions could be placed on those sites that use this material; however, this practice is not appealing to most land owners because of fears of future property marketability. As a result of the work presented here, and other compliance data reported by the C&D debris recycling industry in Florida, the FDEP developed a guidance document outlining requirements for the beneficial use of C&D debris fines (FDEP, 1998). The document provides guidance on establishing baseline concentrations for new C&D debris recycling facilities; outlines routine sampling and reporting requirements for all facilities; and specifies off-site use requirements and restrictions.

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## Waste Management

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## Mobilization of iron and arsenic from soil by construction and demolition debris landfill leachate

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### ABSTRACT

Column experiments were performed to examine (a) the potential for leachate from construction and demolition (C&D) debris landfills to mobilize naturally-occurring iron and arsenic from soils underlying such facilities and (b) the ability of crushed limestone to remove these aqueous phase pollutants. In duplicate columns, water was added to a 30-cm layer of synthetic C&D debris, with the resulting leachate serially passed through a 30-cm soil layer containing iron and arsenic and a 30-cm crushed limestone layer. This experiment was conducted for two different soil types (one high in iron (10,400 mg/kg) and the second high in iron (5400 mg/kg) and arsenic (70 mg/kg)); also monitored were control columns for both soil types with water infiltration alone. Despite low iron concentrations in the simulated C&D debris leachate, elevated iron concentrations were observed when leachate passed through the soils; reductive dissolution was concluded to be the cause of iron mobilization. In the soil containing elevated arsenic, increased iron mobilization from the soil was accompanied by a similar but delayed arsenic mobilization. Since arsenic sorbs to oxidized iron soil minerals, reductive dissolution of these minerals results in arsenic mobilization. Crushed limestone significantly reduced iron (to values below the detection limit of 0.01 mg/L in most cases); however, arsenic was not removed to any significant extent.

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### 1. Introduction

A documented occurrence at waste disposal sites (landfills) is the mobilization of naturally-occurring iron (Fe) and arsenic (As) from native soils into the groundwater (Keimowitz et al., 2005; Delemos et al., 2006; Parisio et al., 2006; Di Palma and Mecozzi, 2010). In soils that naturally contain arsenic and iron, arsenic will typically bind to oxidized iron (Fe(III)) minerals, such as ferrihydrite, hematite, and goethite (Heron et al., 1994; Altundogan et al., 2002; Mandal and Suzuki, 2002; Catalano et al., 2008; Perez-Lopez et al., 2011). When landfill leachate infiltrates the underlying environment (soil, aquifer), the leachate provides a carbon source for biological activity and creates reducing conditions (as a result of the high organic matter content and reduced compounds present in leachate that enters the soil beneath the landfill). The formation of distinct groundwater redox zones downgradient of landfills, including iron-reducing zones, is well established (Lyngkilde and Christensen, 1992; Heron and Christensen, 1995; Christensen et al., 2001). The reductive dissolution of iron, where the insoluble state of ferric iron (Fe(III)) is converted to the soluble state of ferrous iron (Fe(II)), can result in the

mobilization of arsenic (Cummings et al., 1999; Delemos et al., 2006; Ghosh et al., 2006; Minyard and Burgos, 2007; Pique et al., 2010). While the waste may not be the source of the arsenic, the geochemical conditions created by the discharge from uncontrolled waste disposal (e.g., no liner system, leachate discharge to the environment) result in iron and arsenic mobilization (Delemos et al., 2006).

Elevated concentrations of arsenic, regardless of their source (waste or native soil), raise concern because of arsenic's known toxicity and low risk-based water quality thresholds (e.g., the World Health Organization (WHO) and US EPA drinking water standards for arsenic = 0.01 mg/L; WHO, 2008; US EPA, 2010). Despite the less severe human toxicity, elevation of iron concentrations in groundwater can also pose a concern. In Florida, US, for example, several unlined landfills have been required to undergo corrective action to remediate groundwater, largely because of elevated iron concentrations. While the US EPA secondary drinking water standard for iron of 0.3 mg/L (US EPA, 2011) is frequently exceeded in landfill groundwater monitoring wells, such exceedances did not historically receive regulatory scrutiny as secondary standards exist to address aesthetic concerns (e.g., taste, color, staining). However, given that a newer health-based water quality threshold for iron of 4.2 mg/L (FDEP, 2005) was developed for Florida and is often exceeded in groundwater at landfill sites, an

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increase in monitoring requirements and implementation of remedial action has occurred. Iron concerns have been exacerbated when high-Fe(II) groundwater intercepts a surface water body, causing oxygen depletion and producing iron-rich flocs as a result of oxidation of reduced iron, where Fe(II) is oxidized to Fe(III) (Dalzell and Macfarlane, 1999; Randall et al., 1999; Parisio et al., 2006).

In this research, we investigated the potential for leachate from unlined construction and demolition (C&D) debris landfills to promote reductive dissolution of naturally occurring iron and arsenic in underlying soils. Many US states do not require liners for C&D debris landfills (Clark et al., 2006), thus leachate may directly enter the underlying soil system and groundwater. As a result of the types of waste found in C&D debris (building materials such as concrete, asphalt, wood, drywall), leachate from C&D debris landfills is expected to contain less readily biodegradable organic matter compared to landfills used for disposing household waste containing food and paper products (Townsend et al., 1999; Weber et al., 2002). The organic matter present in C&D debris leachate largely originates from woody materials such as lumber and vegetative matter and should be more recalcitrant to biodegradation.

The research presented here was motivated by uncertainty in the regulatory and waste disposal community in Florida regarding the cause of elevated iron and arsenic concentrations in monitoring wells at unlined C&D debris landfills. Although iron and arsenic are known to naturally occur in Florida soils, and evidence from elsewhere suggests that municipal solid waste (MSW) landfill leachate can result in mobilization, C&D debris components include both iron- and arsenic-containing materials (e.g., steel, arsenic-treated wood; Kartam et al., 2004; Jambeck et al., 2007; Hawley et al., 2009; Cochran and Townsend, 2010), and thus represents another possible contamination source. Thus, one objective of this work was to assess whether C&D debris leachate could promote reductive dissolution of soil iron minerals, and if present, the subsequent mobilization of arsenic. A second objective was to examine the approach of using calcium carbonate materials (e.g., limestone) to remove iron (and possibly arsenic) from groundwater plumes containing mobilized Fe(II). Calcium carbonate is an abundant material in Florida; such minerals have been noted to possess the potential for removing metals from groundwater (Aziz et al., 2001), and under certain environmental conditions Fe(II) can precipitate as a carbonate mineral (e.g., siderite).

## 2. Materials and methods

### 2.1. Column materials

Four different types of materials were employed in the columns: synthetic C&D debris, soil, drainage gravel, and crushed limestone. A simplified C&D debris composition was used to provide the major components of this waste stream (percentages by mass): concrete (40%), wood (45%), and drywall (15%). While other components also occur in C&D debris (e.g., metal, asphalt, cardboard), based on experience from similar studies, use of only the major components (i.e., concrete, wood, drywall) would meet the objectives of creating leachate with the typical organic matter composition (primarily from wood waste) and chemistry of C&D debris landfill leachate (Yang et al., 2006; Jambeck et al., 2008; Dubey et al., 2009). Wood (untreated southern yellow pine) and drywall were purchased from building material stores and the concrete was collected from a concrete processing facility. Each material was size-reduced (as needed) and screened to pass a No. 4 sieve (1.27 cm).

Two soils were used, each in separate columns. One soil (S1) was collected from a solid waste facility in Florida known to have

elevated iron issues in the surrounding groundwater monitoring wells. This soil possessed an average iron concentration of 10,400 mg/kg, was low in arsenic (<0.5 mg/kg), had a pH of 4.7, and was yellow in color. Goethite was detected by XRD analysis. The other soil (S2) was collected from a site in New Hampshire known for elevated arsenic concentrations. This soil, S2, possessed an average iron concentration of 5820 mg/kg, an arsenic concentration of 72.4 mg/kg, had a pH of 6.2, and was brown in color. All soils used in the columns were sieved to less than 2 mm. A DI water extraction test was performed on both soil samples using a 1:1.4 soil/water ratio for 12 h. The Fe(II) and Fe(III) ion concentrations released into water were below detection limit (BDL). The content of amorphous iron contents was analyzed using the method of McKeague and Day (1966). S1 contained 988 mg/kg and S2 contained 427 mg/kg amorphous iron content.

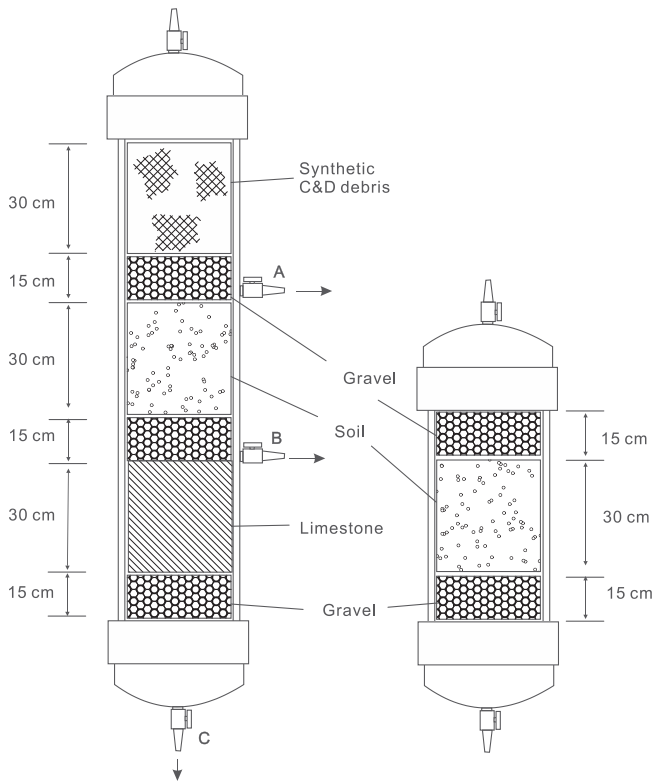
Other than the detection of goethite in S1 through XRD analysis, specific information regarding the mineralogy of the iron and arsenic species in the soil was not determined. However, based on soil color, the primary Fe(III) minerals present in S1 were likely goethite, hematite, and akageneite, and those present in S2 were likely ferrihydrite, goethite, akageneite, hematite, and maghemite (Heron et al., 1994; Arimoto et al., 2002). Both arsenate and arsenite are known to sorb to iron oxides in soils and could therefore have both been present sorbed to the soil minerals (Cummings et al., 1999). The reductively soluble arsenic is often assumed to exist as the As(III) species (Cummings et al., 1999; Masscheleyn et al., 1991; Pique et al., 2010).

The limestone was purchased from a mineral supply store and was screened to a particle size less than 0.8 mm. The composition of the limestone was determined to be 95% CaCO<sub>3</sub>, 2.7% MgCO<sub>3</sub>, and 2.3% other minerals by a Bruker AXS ARTAX 800 micro-XRF spectrometer. For the drainage layers, washed pea gravel from a building supply store was utilized.

### 2.2. Column construction and operation

Eight laboratory leaching columns were constructed of 10.2-cm (4-inch) diameter PVC pipe (four experimental scenarios were simulated in duplicate columns). Column construction details and configurations are presented in Fig. 1. Columns 1 and 3 (and their duplicates) included a waste layer for producing C&D debris leachate, a soil layer (S1 was used for column 1 and S2 was used for column 3), and a remedial layer (limestone). Drainage layers (used for sample collection) were placed below each layer. From the top to the bottom of the column, the strata consisted of the C&D waste layer (30 cm), a drainage layer (15 cm gravel), a soil layer (30 cm), a drainage layer (15 cm), a limestone layer (30 cm), and a final drainage layer (15 cm). Additional columns were prepared to serve as control columns, containing only a soil layer (30 cm) with a drainage layer (15 cm gravel) above and beneath (column 2 for S1 and column 4 for S2). A piece of nonwoven geotextile was used to separate the different layers in the columns. Ports for sample collection (labeled A, B, and C in Fig. 1) were installed in the drainage layers of the test columns. For the control columns, samples were collected only from the bottom of the drainage layer, Fig. 1.

During operation, a total of 2 L of DI water was added to the top of each column weekly. A peristaltic pump (Cole-Parmer Instrument Co.) was used to feed the water at a flow rate of 10 mL/min until the 2-L reservoir was exhausted. The water passed through the column by gravity, taking approximately 30 min to pass through the waste layer, 40 min to pass through the soil layer, and 120 min to pass through the limestone layer. A 100-mL leachate sample was collected from all three ports of the experimental columns weekly. Remaining liquids from the bottom of the column were discarded; no liquids were recirculated and only DI water was added to the top of the column. For the control columns,



**Fig. 1.** Schematic of testing (left) and control (right) columns used in the study (for the testing columns ports A, B, and C) shows the sampling ports for leachate after it passes through the different layers of synthetic construction and demolition (C&D) debris, soil, and limestone.

one sample was collected from the bottom of the column. The total operation time for columns 1 and 2 was 34 weeks and the total operation time for columns 3 and 4 was 27 weeks. The experiments were concluded at the same time.

### 2.3. Chemical analysis

The pH, oxidation-reduction potential (ORP) (Accumet Co., Model 20), dissolved oxygen (DO) (Thermo Inc., Model 23), and Fe(II) (Standard Method 3500) (APHA, 1995) were measured in all leachate samples. The detection limit of Fe(II) was 0.01 mg/L. The concentrations of iron, arsenic, and other cations (e.g., calcium and sodium) in the soil and leachate were analyzed by digesting the samples following US EPA SW846 3050B method and US EPA SW846 3010A method, respectively, followed by analysis using inductively coupled plasma atomic emission spectroscopy per US EPA SW846 6010C method (ICP-AES, Thermo Jarrell Ash Corp. Model 95970). Detection limits of iron, arsenic, calcium, and sodium were 0.01 mg/L, 0.004 mg/L, 0.1 mg/L, and 0.3 mg/L, respectively. Soil pH values were analyzed per US EPA method SW846 9145D. Additional leachate was sampled on days 7, 49, 119, and 189 and analyzed for non-purgeable organic carbon (NPOC) (Shimadzu total organic carbon analyzer, TOC-V<sub>CPH</sub>), total dissolved solids (TDS) (Standard Method 2540), and alkalinity (Standard Method 2320) (APHA, 1995). During analysis, laboratory blanks, matrix spikes, and calibration checks were performed as appropriate for quality control. Analyses of blank samples were consistently below detection limits; matrix spike samples and calibration check samples showed recoveries between 90 and 110%. Statistical analyses were performed using one-way analysis of variance (ANOVA) followed by a post hoc Dunnett's test utilizing the control columns as a reference. A *p* value of less than 0.05 was considered to be statistically significant.

## 3. Results and discussion

### 3.1. C&D debris leachate quality

General water quality characteristics of leachate collected from the bottom of the waste layers (port A of columns 1 and 3) fell within the typical range of C&D debris leachate from previous lab experiments and field measurements (Table 1). The average pH and ORP values of the samples from the experimental columns are presented in Fig. 2. The pH ranged from 6.1 to 6.9. Leachate pH from the waste layers of columns 1 and 3 varied until day 100 of the experiment, but stabilized at approximately 6.4 for the rest of the experimental period, on the lower end of the range typical of C&D debris landfill leachate (6.5–7.6) as shown in Table 1. The ORP of the C&D debris leachate in both columns shifted from positive to negative values within two weeks. These results, coupled with the increase in alkalinity, indicate a shift in the C&D debris layer from oxidizing to reducing conditions. When drywall or similar gypsum wastes are present, reducing conditions in C&D debris landfills develop as a result of the activity of sulfate-reducing bacteria that consume organic matter and utilize sulfate (present in large amounts because of gypsum) as an electron acceptor (Yang et al., 2006; Jambeck et al., 2008). The concentrations of NPOC, TDS, sodium, and calcium fell in the range of those found in typical C&D debris leachate.

Arsenic concentrations in the C&D debris leachate were below detection limit (4 µg/L) during the experimental period. The majority of the Fe(II) leachate measurements throughout the experimental period were below detection limit (0.01 mg/L). Iron and arsenic have routinely been detected in other C&D debris leachate evaluations (Table 1), but the absence of these two chemicals in the present study was not surprising. Previous studies reported total iron and not Fe(II), so it is difficult to assess the true extent of dissolved iron concentrations. Particulate-bound iron is common in leachate and small amounts were observed periodically in this study (<1 mg/L). In the form of steel, iron will exist in the zero-valent form and thus is not expected to undergo reductive dissolution. However, some oxidized iron (surfaces of steel wastes, landfill cover material) will be present in such landfills and thus some leachate iron is expected; no steel materials were included in the waste stream in this study since a major objective was to assess the potential for the soil to act as the source of dissolved iron. The concentrations of arsenic measured in previous studies have most often been a result of arsenic-treated wood (Jambeck et al., 2007; Dubey et al., 2009), and in this research, no such material was added.

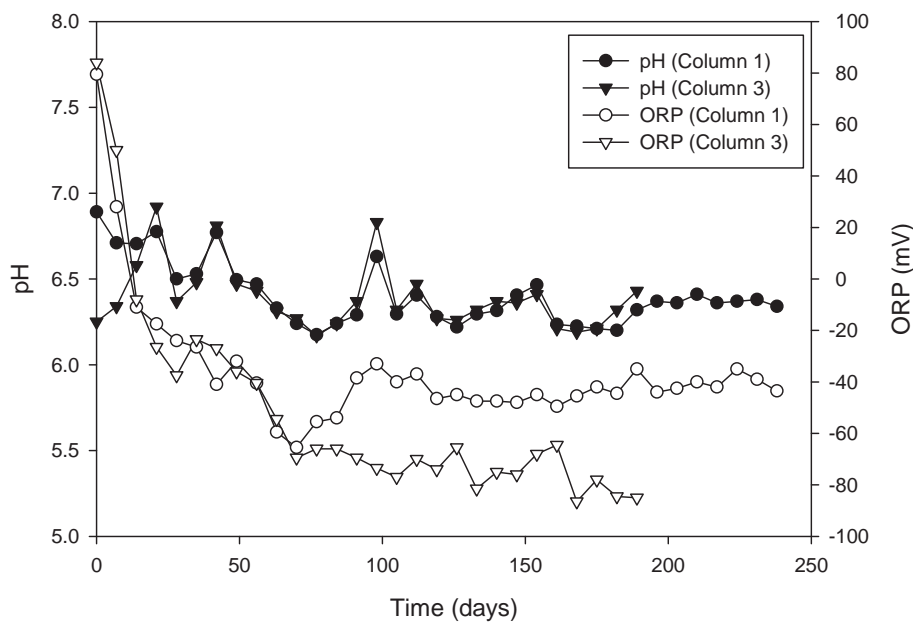
### 3.2. Iron mobilization

Fig. 3 shows the Fe(II) concentrations with time in samples collected from ports A and B for the columns containing the S1 soil. Dissolved Fe(II) concentrations measured after water passed through the layer simulating C&D debris (port A) were in nearly all cases lower than the detection limit (0.01 mg/L), whereas the Fe(II) concentrations in the leachate after it passed through the soil were in the range of 3–6 mg/L. These concentrations were more than an order of magnitude greater than the concentrations measured from the control columns (<0.01–0.03 mg/L) and considered statistically significant at the 95% confidence interval for 32 of 37 sampling events for S1 and 29 of 37 events for S2, including the initial event at 0 days for S1 and S2. Similar results were encountered in the columns containing the S2 soil (S2); see Fig. 4. Fe(II) concentrations increased above 1 mg/L after 30 days and remained in a range of 1–2 mg/L through the remainder of the experiment (a maximum concentration of 1.8 mg/L was measured). The Fe(II)

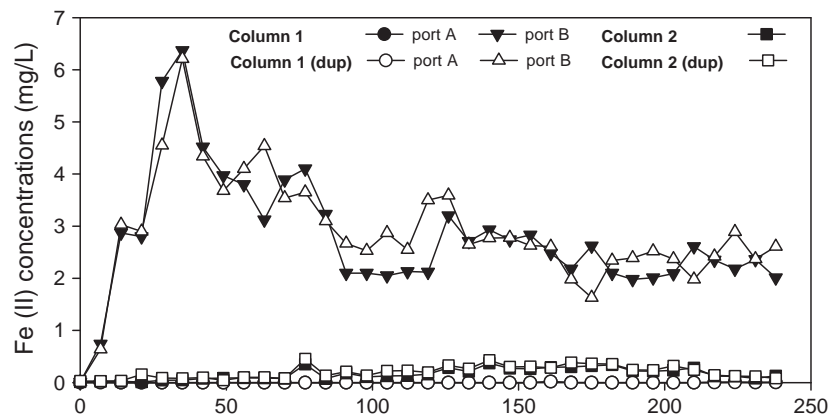
**Table 1**  
 Comparison of measured construction and demolition (C&D) debris leachate parameters with published literature values.

Parameters	This study <sup>a</sup>	Melendez, 1996 <sup>b</sup>	Jang and Townsend, 2003 <sup>c</sup>	Weber et al., 2002 <sup>d</sup>	Jambeck, 2004 <sup>e</sup>	Dubey, 2005 <sup>e</sup>
pH	6.40	6.45–7.60 (6.95)	6.5–7.0	6.90	6.5–7.0	6.5–7.0
Alkalinity (mg/L as CaCO <sub>3</sub> )	75–725	38.2–6520 (970)	–	530	550–2000	1500–2500
NPOC (mg/L)	20–36	19.0–1900 (310)	–	21.1	–	–
TDS (mg/L)	873–2010	990–3530 (2260)	1640–3000	2120	1720–4160	2200–7000
Sodium (mg/L)	21–37	11.0–1290	–	42.8	130–500	–
Calcium (mg/L)	274	90–600 (270)	300–690	470	210–790	–
Iron (mg/L)	<0.01 <sup>f</sup>	0.05–275 (36)	–	1.65	–	–
Arsenic (µg/L)	<4.0	1.4–24.6 (12.3)	10–380	41.4	12.5–38.1	50–300

<sup>a</sup> Average/ranges experimental columns 1 & 2.  
<sup>b</sup> Full-scale study. Concentration ranges from literature review of C&D leachate, values in parentheses indicate average value for each parameter.  
<sup>c</sup> Lab-scale C&D lysimeter study (0.5% CCA treated wood added).  
<sup>d</sup> Field-cell average.  
<sup>e</sup> Pilot-scale C&D lysimeter study (control column: no CCA treated wood added).  
<sup>f</sup> Measured as Fe(II).



**Fig. 2.** Changes in pH and oxidation–reduction potential (ORP) of C&D debris leachate (port A) over time. Values represent the average of duplicate columns.



**Fig. 3.** Change of ferrous iron concentration in leachate before (port A) and after (port B) passing through iron-rich soil (S1) as a function of time.

concentrations of the control columns were relatively stable and consistently lower than 0.4 mg/L.

Within the S1 columns a one-way ANOVA test showed that the Fe(II) concentrations from ports A, B, and C varied significantly at

the 95% confidence interval ( $p < 0.05$ ) for 34 of 37 sampling events. Further post hoc analysis (one-sided Dunnett's test) demonstrated that the variation between these ports showed that Fe(II) concentrations in port B varied significantly from Fe(II) concentrations in

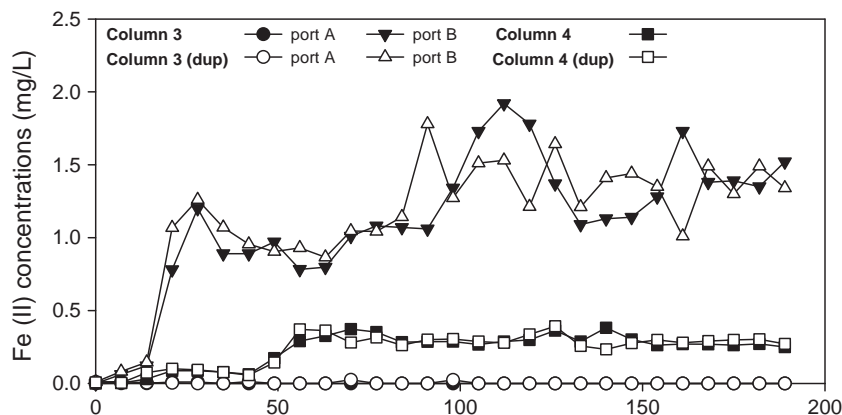


Fig. 4. Change of ferrous iron concentration in leachate before (port A) and after (port B) passing through iron- and arsenic-rich soil (S2) as a function of time.

ports A and C only during the first measurement in the study (at 0 days), when concentrations from ports A and C were below detection and resulted in a  $t$  statistic lower than  $t_{critical}$ . Also the relatively low degrees of freedom in the experiments influenced a relatively high value of  $t_{critical}$ . No significant difference in Fe(II) concentrations was observed between ports A and C for both the S1 and S2 soil columns.

A primary objective of our research was to assess whether the infiltration of C&D debris could promote the reductive dissolution of iron from soil, and the results demonstrate that this can occur. Previous research on MSW landfills, for example Di Palma and Mecozzi (2010), shows that leachate can result in the mobilization of naturally occurring elements in soil. Even though C&D debris landfill leachate does not contain the same readily degradable organic matter as expected with MSW landfill leachate, the present research demonstrates such leachate can similarly promote reducing conditions. Several factors could contribute to promoting reducing conditions (e.g., providing organic matter for consumption by iron-reducing bacteria, the chemically reducing nature of the C&D debris leachate), and this study was not designed to determine the relative role of these factors (future work should target this). Some degree of reductive dissolution and iron mobilization might occur at a landfill site regardless of leachate infiltration because of the other geochemical changes that develop (e.g., interruption of natural water cycles and soil vapor exchange); this merits further attention.

The results support that reductive dissolution was the primary cause of iron release, though no specific experiments were conducted to determine the role of biotic versus abiotic reduction; the authors suspect that biotic reductive dissolution was dominant. Changes in pH, which result from both the pH of the leachate as well as the subsequent reduction reactions, can also impact iron release. Figs. S1 and S2 in the Supplementary material depict the log Fe(II) concentrations over the range of pH values observed in the study. Among the different sample types (leachate from waste, leachate passing through soil, water passing through soil), pH values were relatively similar throughout the experiment and thus no dramatic difference in Fe(II) concentration as a function of pH was noted. In a study by Di Palma and Mecozzi (2010), column tests were conducted using soil taken from under a municipal landfill and groundwater from the same site; their results showed no significant difference in dissolved iron leached from soil at pH values of 6.5 and 8.5.

The concentration of iron observed in the leachates after passing through both soils was greater than the secondary drinking water standard (0.3 mg/L) and in the case of S1, at times greater than Florida's risk-based threshold (4.2 mg/L). However, care must be taken in comparing column results to regulatory thresholds, as

many other factors would influence the concentrations likely to be reached in the environment at a landfill site (e.g., waste deposit area, aquifer thickness, groundwater velocity). Measurements observed by the authors at Florida landfills are often 10 mg/L or greater. Delemos (2006) found iron levels in groundwater surrounding a closed, capped MSW landfill to range from 10 to 70 mg/L. Minyard and Burgos (2007) performed a column experiment using an iron-rich coastal sand (20,400 mg/kg extractable iron, predominantly hematite), synthetic groundwater, and a source of organic material and bacteria, and reported mobilized Fe(II) concentrations in the range of 33–40 mg/L.

While the solutions collected from the control columns (the soils exposed to water, not leachate) were lower in Fe(II) concentration compared to those exposed to leachate, some Fe(II) was mobilized from the soils; Fe(II) was not initially present in the pore water collected from the base of the columns, but increased over time to a steady magnitude. This was more pronounced for the S2 soil even though its total iron concentration was half that of the S1 soil (5800 mg/kg vs 10,400 mg/kg). The results suggest that the geochemical environment created in the soil layer with water addition supported reductive dissolution (though to a lesser extent than under similar conditions with leachate addition), and characteristics of S2 were more amenable to reductive dissolution occurring. Based on the current literature, it is speculated that iron release occurred primarily from the amorphous (poorly crystalline) iron oxides found in the soil, as they have been shown to reduce more readily under biological and chemical means (Lovley and Phillips, 1986; Wahid and Kamalam, 1993; Roden, 2003).

### 3.3. Arsenic mobilization

Since the S1 soil did not contain a detectable arsenic concentration, arsenic mobilization in relation to iron reductive dissolution was only assessed using the S2 soil (Fig. 5). As expected, arsenic was not detected in the C&D debris leachate, but it was observed after the leachate passed through the soil (port B). Detectable concentrations of arsenic were first observed at day 50, after which they remained in the range of 10–25 µg/L. For perspective, this is greater than the EPA and WHO drinking water standard of 10 µg/L (WHO, 2008; US EPA, 2011). Using a one-way ANOVA comparing experimental column As concentrations from ports A, B, and C, concentrations were found to vary significantly ( $p < 0.05$ ) for 10 of the 28 sampling events, which prompted a post hoc Dunnett's test. After 56 days, As concentrations in samples from Port B were significantly larger than Port A for 20 of 28 sampling events.

Arsenic mobilization did not immediately correspond to that observed with iron; arsenic release occurred at day 50, whereas iron release was observed at day 7 (Figs. 4 and 5). Several factors

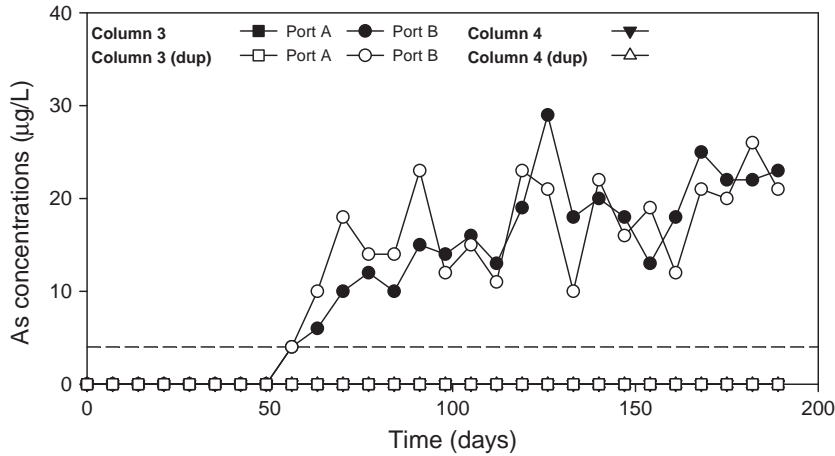


Fig. 5. Change of arsenic iron concentration in leachate before (port A) and after (port B) passing through iron- and arsenic-rich soil (S2) as a function of time.

may have contributed to this observation. As reductive dissolution occurs and arsenic is released into solution, some of the arsenic should sorb with remaining Fe(III) minerals, thus retarding travel through the soil layer. Carbonate ions have been reported to displace sorbed arsenic from Fe(III) minerals (Appelo et al., 2002), and in the present study, a large increase in leachate alkalinity between days 7 and 50 corresponded to the increase in arsenic concentrations measured in the leachate after passing through the soil layer (Fig. 5).

The discussion of arsenic thus far has been simplified, referring to arsenic as a single chemical species. However, much like the Fe(III)/Fe(II) redox couple, arsenic's fate and mobility are influenced by the As(V)/As(III) redox couple (Kumpiene et al., 2009). For example, Islam et al. (2004) discuss the mobilization of arsenic from aquifer sediments and describe the potential complex role of both Fe(III) and As(V) reduction after their subsequent fate. In batch studies, they observed an arsenic release that followed in time after an iron release (similar to observations from the present study) and attributed this in part to the redox potential of the Fe(III)/Fe(II) couple compared to the As(V)/As(III) couple. Similarly, Masscheleyn et al. (1991), in an experiment designed to test arsenic speciation and solubility at different Eh and pH values, observed the simultaneous reduction of As(V) to As(III) and increases in soluble As.

### 3.4. Efficacy of limestone

Limestone proved to be effective at removing Fe(II) from the leachate solutions that passed through the soil layers (columns 1 and 3), reducing Fe(II) concentrations from the range of 1–6 mg/L to less than the detection limit of 0.01 mg/L (see Fig. 6). Throughout the experiment, all of the iron concentrations measured in leachate passing through a limestone layer (columns 1 and 3, port C) were below the secondary drinking water limit (0.3 mg/L). Mettler et al. (2009) reported that Fe(II) sorbed on calcite particle surfaces rapidly, followed by co-precipitation to form Fe(II) minerals. Aziz and Smith (1992) reported that limestone has the ability to remove other bivalent metal cations; for example, manganese (II) was removed by greater than 95% from an initial concentration of 1 mg/L, and copper (II) was removed by greater than 90% from an initial concentration of 50 mg/L (Aziz et al., 2001). Future work will focus on identifying the responsible removal mechanisms, but candidate processing includes the formation of the mineral siderite (FeCO<sub>3</sub>), the sorption of Fe(II) onto the limestone surface, and sorption followed by co-precipitation.

The limestone layer was largely ineffective at decreasing the arsenic concentration (data not shown). Using a one-sided post hoc *t*-test at 95% confidence, no As concentrations from port B compared to port C were found to vary significantly. Other studies have

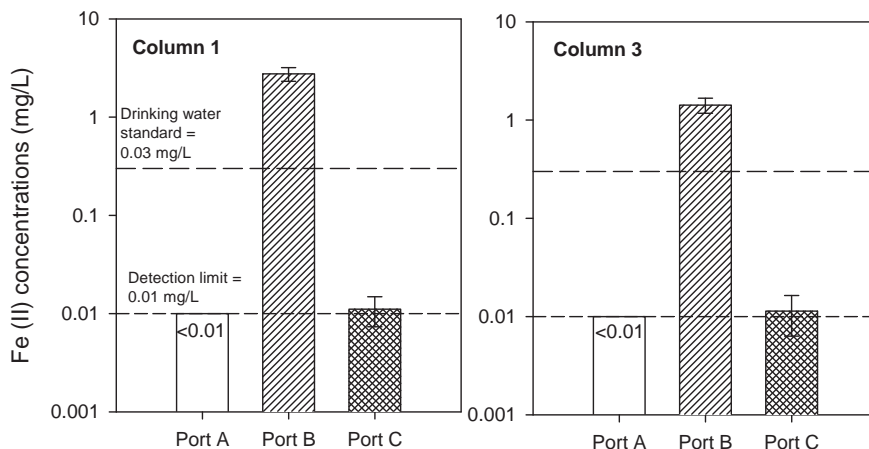


Fig. 6. Comparisons of Fe(II) concentrations in leachate collected from ports A, B, and C of columns 1 and 3 for days 100 through 150. In computing statistics, concentrations below the detection limit were set equal to the detection limit.



suggested that limestone has a weak affinity on arsenic (Hossain and Islam, 2008; Nillson et al., 1994). Gibert et al. (2010) studied arsenic removal by mixing zero-valent iron, limestone, and compost, and summarized that arsenic was removed through co-precipitation with iron and aluminum (oxy) hydroxides and even more so by sorption onto zero-valent iron, and not by sorption onto or reaction with limestone.

#### 4. Summary and conclusions

Column experiments were performed to examine whether C&D debris leachate (without iron and arsenic) can mobilize naturally-occurring iron and arsenic from soils that might exist at unlined landfill sites. In addition, the potential for limestone to react with and remove mobilized iron and arsenic was examined. This research was restricted to the analysis of two soils, one high in iron (10,400 mg/kg) and one high in iron and arsenic (5400 mg/kg iron and 70 mg/kg arsenic). The soils were exposed to the infiltration of C&D debris leachate created by DI water passing through synthetic C&D debris. Control columns were included for both soil types, allowing the water to only pass through the soils.

Low iron and arsenic concentrations (below US federal drinking water limits) were consistently encountered in water that had passed through the C&D debris layer to create leachate. The infiltration of this leachate through both soil types resulted in the mobilization of Fe(II). While leachate from C&D debris landfills is thought by some to be relatively inert in comparison to MSW landfill leachate, the results demonstrate that for landfills containing soils amenable to reductive dissolution of Fe(III) minerals, discharge of leachate from an unlined disposal area can result in Fe(II) mobilization. Although iron in a C&D debris landfill's leachate may represent a discharge source to the underlying aquifer, iron presence in the leachate is not required for Fe(II) mobilization. The environmental impact of elevated iron concentrations in groundwater at a C&D debris landfill may be debated, but the occurrence should not represent a surprise.

Arsenic mobilization from the high-arsenic soil examined also occurred, again indicating that at unlined C&D debris landfill sites with certain soils (containing arsenic, subject to reductive dissolution), elevations of arsenic concentrations in groundwater well samples should be anticipated, regardless of whether arsenic-containing materials are present in the waste. Measurements of arsenic release did not immediately correspond to iron mobilization; additional work will be needed to assess the specific nature of arsenic release and mobilization, and its interrelation with iron reductive dissolution.

Crushed limestone was successfully utilized to remove iron from the soil-impacted leachate (from a maximum concentration of 6 mg/L to below 0.01 mg/L). However, arsenic was not removed appreciably. The ability of limestone to remove Fe(II) from groundwater opens the door for alternative remedial strategies to more commonly practiced pump-and-treat systems. One example currently under investigation is the use of calcium carbonate or similar materials as a passive reactive barrier, especially at key points where aquifers discharge into surface waters and create ecological impairment.

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#### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.wasman.2011.11.016.

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