[illegible]

[illegible]

[illegible]

Produced by:

Dr. Kenneth M. Brooks
Aquatic Environmental Sciences
644 Old Eaglemount Road
Port Townsend, WA 98368

March 13, 2012

[illegible]

[illegible]

[illegible]

The following comments are offered in an effort to help the authors focus their analysis of the considerable data generated in their efforts on questions appropriate to the protocols actually used in the various studies. The motivation behind Townsend and Solo-Gabriele (2001 draft) is provided in Section 1.2 where the authors state:

"In order to better evaluate the potential impacts of the alternative wood treatment preservatives on the environment it was thus important to evaluate the relative chemical composition and aquatic toxicity of leachates from the alternative chemical treated wood versus that of CCA-treated wood. . . . . This study was designed to compare both leaching and aquatic toxicity of ACQ-, CBA-, CC-, CCA-, and CDDC-treated wood with each other under a consistent set of experimental conditions."

These are appropriate questions that could form the basis of useful studies. However, the statement of purpose fails to specifically identify the environment of concern and therefore asks an incomplete question. The questions being asked could apply to landfill leachates, to upland sites, to groundwater, or to open bodies of surface water. Appropriate protocols applicable to the study of landfill leachates would likely involve low pH, anaerobic conditions with high sulfide levels, and treated wood removed from service after 15 to 20 years. If the questions focus on surface water effects, appropriate material for analysis would have been wood treated with approved preservatives to retentions specified in American Wood-Preservers' Association (AWPA) standards for freshwater use (C2, C3) or marine use (C3, C18). Furthermore, to address aquatic risks, the test conditions should have mimicked those typically found in surface waters (commodity size samples of newly treated wood in water with a hardness of 50 to 200 mg/L as $CaCO_3$; pH between 6.5 and 8.0; dissolved oxygen >5 mg/l and ambient temperature of 15 to 20 ^{o}C).

In short, the first significant flaw in this study is a failure to adequately define the question being asked. The protocols suggest the questions focused on landfilling of newly treated wood, while the author's interpretation covers a range of subjects. Because the Townsend *et al.* (2001) study includes toxicity testing using aquatic invertebrates, one must assume that the primary question referred to pressure treated wood used in contact with surface waters. However, the study design was not appropriate to this question. This statement is made unequivocally for the following reasons.

1. Inappropriate preservatives and/or retentions were evaluated. Pressure treated wood preservatives and retentions are specified in AWPA (2001). Preservatives and the retentions approved by AWPA Standard (C2) in pounds per cubic foot (*pcf*) for preservation of southern yellow pine used in freshwater and saltwater are compared with the samples used by Townsend *et al.* (2001) in Table (1).

Table 1. Comparison of AWPA approved uses and retentions for lumber timber and ties (Standard C2) used in aquatic environments with retentions and uses examined by Townsend *et al.* (2001). All values are in pounds per cubic foot (*pcf*). Actual assayed retentions are provided in parentheses.

Preservative	AWPA (2001) Approved Retentions		Retentions used by Townsend *et al.* (2001)
	Freshwater	Seawater	
Alkaline Copper Quaternary (ACQ-D)	0.40	Not approved	0.250 (0.285)
Copper Boron Azole (CBA-A)	0.408	Not approved	0.204 (0.194)
Copper Citrate (CC)	0.40	2.50	0.250 (0.208)
Chromated Copper Arsenate (CCA-C)	0.40	2.50	0.250 (0.263)
Copper Dimethyldithiocarbamate (CDDC)	0.20 as copper	Not approved	0.10 (0.131) copper 0.25 – 0.50 (0.419) CDDC

The retentions used by Townsend *et al.* (2001) were lower by factors of 1.6 (ACQ-D) to 10.0 (CCA-C in seawater) than those specified by AWPA Standards. It should be noted that only CCA-C, ACA and ACZA are approved for treating piling (Standard C3) in either freshwater or marine environments. The studies examination of ACQ, CBA and CDDC in seawater is questionable because these preservatives are not approved for any marine uses. Brooks (1996a, 1997a, 1997b, 2002) has shown that metal loss rates from pressure treated wood are a function of preservative retention and the examination of wood at other than AWPA approved retentions will give misleading results. If Townsend *et al.* (2001) wanted to evaluate aquatic risks, they should have obtained wood treated for "Soil and Fresh Water Use" rather than for "Above Ground" use (see AWPA 2001).

The bottom line is that Townsend *et al.* (2001) examined wood that was pressure treated with unapproved preservatives and/or inappropriate retentions for determining absolute or relative metal loss rates in aquatic environments. If the questions being asked were focused on landfill leachates and potential groundwater effects, then the use of newly treated wood was also inappropriate. In this latter case, the authors should have used pressure treated wood representative of the majority of the construction and demolition waste stream – i.e., wood taken out of service after at least 15 to 20 years of use. Lastly, the CDDC preservative recommended as comparable to CCA by Townsend *et al.* (2001) is no longer manufactured due to high costs and was never approved for use in marine environments.

2. Static tests are inappropriate for determining metal loss rates to aquatic environments. Brooks (2002) conducted a static leaching study on sections of CCA-C treated piling treated to 2.5 *pcf* that had been removed from service after being immersed in an east coast estuary for 16 months. The leachate's copper concentration was 651.8 ± 166 µg Cu/l (N = 20; mean ± 95% confidence) after 17 days of static immersion. The copper concentration increased to only 677 ± 302 µg Cu/L at the end of 34 days. The increase during the second half of the study was small and not statistically significant. This was unexpected because copper losses following 16 months of immersion are not expected to change as a function of time. Discussions with Dr. Eric Crecelius at Battelle Marine Sciences Laboratory supported Dr. Brooks' hypothesis that copper in the leachate had likely reached an equilibrium concentration with the unbound copper in solution in the wood. Dr. John Simonsen at Oregon State University (work in progress) has observed increasing PAH losses from creosote treated wood as a function of increasing current speeds in their dynamic leaching study.

My point is that static leaching tests, resulting in high leachate concentrations of preservative, likely underestimate loss rates from treated wood in aquatic environments. Townsend *et al.* (2001) reported copper concentrations in their leachate ranging from 3,300 to 10,000 µg/l. These values greatly exceed those observed by Brooks (2002) in his static leaching study, suggesting that further losses from the treated wood examined by Townsend *et al.* (2001) were also inhibited.

3. The pulverized test samples were not representative of wood actually used to construct projects in aquatic environments. Townsend *et al.* (2001) pulverized all of the wood samples to a < 3 mm particle size. The authors noted that the US EPA Toxicity Characteristic Leaching Procedure (US EPA, 1996) specifies a larger sieve opening of 9.5 mm. Fixation of CCA treated wood is a complex process in which the metals are bound by a number of mechanisms such as ion exchange, hydrogen bonding, etc. (Jin and Archer, 1991). Bull (2001) described the products of CCA-C fixation as being “dominated by chromium (III) arsenate, chromium (III) hydroxide, and copper (II) – wood carboxylate complexes.” Pizzi (1982) also demonstrated a variety of binding mechanisms for metal-chromium complexes with lignin and cellulose and of copper with guaiacol or glucose. He also noted that $CuSO_4$ is physically adsorbed within the wood structure, but is soluble and more highly leachable than other forms of bound copper.

Assuming that the copper is in solution in saturated wood immersed in aquatic environments, the question becomes one of how does the copper leave the wood? Water passage is facilitated through longitudinal tracheids and vessels; it diffuses much more slowly in a lateral direction. Therefore, water containing copper, chromium and/or arsenic in solution is more mobile along the longitudinal axis of a piece of wood than in a radial direction. The point is that metal losses from end-grain are expected to be much larger than losses across the face grain. In practice, a piling has one end (end grain) buried in sediments and the other end exposed to air. Essentially all of the leaching occurs across the less permeable face grain. Lebow (1996) discussed the problems associated with leaching treated wood having a high ratio of end-grain to face grain – including the 19-mm cubes specified in AWPA (1994).

An immersed piece of 2” x 6” lumber that is 12 feet long has an end grain to face grain ratio of 0.004 indicating that essentially all of the leaching is across the less permeable face grain. This is why recent studies to determine metal loss rates from treated wood into aquatic environments generally seal the end-grain with two coats of neoprene rubber (Lebow *et al.* 1999) or resin (Brooks 2002). The importance of this is emphasized by the findings of Cockcroft *et al.* (1978) and Wilson (1971) who noted that high density, lower permeability, wood species tend to be more resistant to leaching than highly permeable species.

The TCLP test used by Townsend *et al.* (2001) greatly enhances the availability of the internal cell walls and soluble metals creating a situation where end-grain exposure is approximately equal to that of the face grain. This unrealistically exacerbates metal loss rates by creating conditions that are not found in the real world. The TCLP test is not appropriate for determining losses of metals to aquatic environments or to landfills from treated wood. Furthermore, it is unclear why Townsend *et al.* (2001) further jeopardized the applicability of their results to the real world by reducing the maximum particle size from 9.5 mm (required by TCLP) to 3.0 mm.

4. If the questions being asked applied to surface waters, then inappropriate diluents were used. Townsend *et al.* (2001) leached pulverized wood in deionized water, buffered organic acid, inorganic acid, and synthetic seawater – none of which are representative of either fresh or marine water.

- **Deionized water**. Untreated pulverized wood contains acid resins, which can significantly reduce diluent water pH in the absence of adequate buffering (Cooper 1991). Further evidence of this is seen in the final pH of 4.49 observed by Townsend *et al.* (2001) for untreated and pulverized SYP leached in deionized water. The result was that the ending pH values observed in the SPLP, TCLP and DI tests were unrealistically low for determining metal loss rates to real surface waters. The mean ending pH values, by preservative and leachate solution, are provided in Table (2).

Natural surface or ground water contains organic and inorganic compounds that buffer the pH – particularly at typical hardness values of 50 to 200 mg/l as $CaCO_3$. Brooks (unpublished) has not observed decreases in pH when conducting flow-through tests on commodity size products using well water (pH = 6.8 to 7.0). Surface waters of the U.S. generally have pH values ranging between 6.5 and 8.0. The U.S. EPA Water Quality Criteria (WQC) for pH is 6.0 to 9.0 and the Florida pH WQC (FAC 17-320.530) for Class III water is 6.0 to 8.5. Granted, water passing through wetlands may contain high amounts of humic acids and have reduced pH – such as found in Pine Barrens streams in the Atlantic coastal plain. These "black waters" have highly altered biotic communities with few crustaceans and fish (www.epa.gov/airmarkets/acidrain/effects/surfacewater.html). The point being that there are exceptions to a lower limit of pH 6.0 – but these are exceptions and they are not typical of most surface waters in North America. All of the pH values at the end of the tests conducted by Townsend *et al.* (2001) were less than the Florida or U.S. EPA WQC. This has significant implications for both the metal loss rates, which are known to increase from CCA-C and ACQ-B at reduced pH, and for bioassays using *Ceriodaphnia dubia*.

Townsend *et al.* (2001) attempted to demonstrate that the aquatic risks associated with alternative preservatives were not significantly different from those associated with CCA-C. In part, their conclusions are not valid because 1) the final leachate pH for CCA-C (Table 2) was lower than for all of the other preservatives – biasing the metal loss rates against CCA-C and 2) the preservatives and retentions used (see Table 1) are not approved by AWPA for use in either freshwater or seawater.

Table 2. Mean and range of leachate pH values at the end of the leaching tests for five preservatives in three freshwater leachate solutions.

	Freshwater Leachate Solutions			
	Deionized Water	TCLP	SPLP	Average
CCA-C	4.64	4.94	4.50	4.69
ACQ	5.78	5.12	5.25	5.38
CBA	5.83	4.95	5.53	5.43
CC	5.83	5.12	5.82	5.59
CDDC	5.92	4.95	5.81	5.56

If Townsend *et al.* (2001) were interested in determining metal losses to aquatic environments and assessing biological risks, then the tests should have been conducted in surface water with a hardness of ca. 50 to 200 mg/l and pH in the range of 6.0 to 8.5. The use of deionized water, TCLP and SPLP diluents resulted in unrealistically low pH. This biased both the metal loss rates-, and, as will be seen in a later section, the bioassay results.

Because different preservatives respond differently to pH, the unrealistically low pH also invalidates the relative metal loss rates. For instance, Table (3) provides predicted CCA-C and ACQ-B copper loss rates as a function of pH. The last column is the predicted copper loss from each preservative at the pH recorded in the Townsend *et al.* (2001) tests. The ratios of ACQ-B to CCA-C copper loss rates are also provided. Observe that the ratio of copper losses from ACQ-B:CCA-C increases as the pH increases. That is because copper loss from CCA-C is more sensitive to pH than is the loss from ACQ-B. Also note that because Townsend *et al.* (2001) conducted static tests in diluents with little or no buffering capacity, the ending pH value for CCA-C was lower (4.69) than it was for ACQ-B (5.38) or any of the other preservatives for that matter. If these tests had been conducted in flow-through chambers using naturally occurring surface water, with a hardness of even 50 mg/l, then the pH values would have remained stable in the vicinity of their starting points.

The protocols used by Townsend *et al.* (2002) did not lead to reasonable estimates of either the relative or absolute risk posed to aquatic environments by any of the preservatives. It is likely that the relative risks associated with copper released from ACQ-B are actually 26% higher (100*[(9.46/7.50) – 1] than those associated with copper loss from CCA-C that were estimated by Townsend *et al.* (2001). Metal loss rates as a function of pH are not yet available for the other preservatives and it is not possible to even estimate the magnitude of the errors introduced by these inappropriate protocols.

Table 3. Copper loss rates from CCA-C and ACQ-B wood pressure treated to a retention of 0.25 *pcf* as a function of ambient pH.

	Diluent pH				
	4.5	5.0	6.0	7.0	At test pH
ACQ-B	57.06	50.65	39.90	31.43	46.26
CCA-C	6.50	5.68	4.34	3.32	6.17
Ratio ACQ-B:CCA-C	8.78	8.91	9.19	9.46	7.50

- **TCLP (Toxicity Characteristic Leaching Procedure).** The Toxicity Characteristic Leaching Procedure (TCLP) relies on a buffered organic acid solution to extract chemicals from solid wastes. Warner and Solomon (1990) used a citric acid – sodium hydroxide buffer system to assess copper losses from CCA-C treated wood as a function of pH. They reported copper losses equaling 112% of the amount of copper retained during the CCA-C preservation process at a leachate pH of 3.5 and 91.62% at pH = 5.5. This assertion did not make sense because such rapid copper depletion during 40 days of immersion would leave the wood unprotected and vulnerable to fungal decay. In contrast, CCA-C treated wood is known to last for decades in water where the pH is in the range 5.5 to 6.0. Numerous authors have subsequently examined this issue and all have discounted the results of Warner and Solomon (1990). For instance, Cooper (1991) found copper losses of 2.1 to 9.6 percent in equimolar sulfuric acid – nitric acid adjusted water with pH values between 3.5 and 5.5. He determined that the unrealistic losses reported by Solomon and Warner (1990) were a result of the sodium hydroxide – citric acid buffer used in their studies and concluded that, "It is evident that the SHCA solution has a strong potential to remove the CCA constituents even at pH 7.0. These results show that the extraordinary high copper losses from CCA-treated lumber in the study by Warner *et al.* were caused by the use of the NaOH/citric acid buffered leaching solution."

Kim and Kim (1993) examined metal loss from CCA-C treated wood in freshwater with pH adjusted to 2.0, 3.0 and 4.0 by adding seven parts sulfuric to three parts nitric acid. These authors reported 47% copper loss at pH 2.0, 16.6% at pH 3.0, 0.9% at pH 4.0 and 0.2% at pH 6.8. They concluded that, "Based on the results of this study, it might be concluded that losses of CCA components at pH 4.0 or above were not great enough to cause public concern about environmental problems and reduction of biological efficacy in service."

Brooks (1998) analyzed metal losses as a function of retention, leachate temperature, pH salinity and time from CCA-C treated wood using data from Van Eetvelde *et al.* (1995), Lebow *et al.* (1995), Brooks (unpublished) and Putt (1993). Brooks (1998) demonstrated that metal loss rates from ACQ-B treated wood also increase with decreasing pH. The algorithms describing copper loss dependence on pH are provided in equations (1) and (2). Interestingly the exponents for CCA (-0.284*pH) and ACQ-B (-0.239*pH) demonstrate that CCA-C is slightly more sensitive to reduced pH but the two are similar.

Equation (1) Cu $\text{Loss}_{\text{CCA-C}}$ as a function of pH = $-0.031*\text{pH} + 0.65*\ \text{exp}^{-0.284*\text{pH}}$

Equation (2) Cu $\text{Loss}_{\text{ACQ-B}}$ as a function of pH = $265.14*\text{exp}^{-0.239*\text{pH}}$

The use of an organic (acetic) acid likely unrealistically exacerbates metal losses from pressure treated wood. The TCLP procedure is designed to evaluate the loss of toxic compounds from waste in landfills. It is not an appropriate procedure for evaluating metal loss from pressure treated wood to surface or ground waters.

- **SPLP (Synthetic Precipitation Leaching Procedure).** Shieh (2001) reviewed the use of TCLP and SPLP for evaluating municipal solid waste (MSW) combustor ash and waste tires. SPLP contains inorganic acids (60% sulfuric and 40% nitric) simulating acid rain with pH = 4.2. This test would be appropriate for assessing metal loss from pressure treated wood used in overhead structures in areas subject to acid rain. However, the stormwater runoff from those overhead structures falls either on the ground, where most soils buffer the pH - and bind many metals, or it falls into water with a natural buffering capacity that reduces the pH. In order to evaluate aquatic risks, one must consider the transport and fate of contaminants – not just their release from overhead structures. In most areas, the pH of surface water is much higher than that of rainwater. In nearly every case, rainwater is buffered by soils or surface waters and it is inappropriate to assume that biological communities are subject to the conditions existing in rainwater or in stormwater from decks, piers, etc. The results of the SPLP tests conducted in Townsend (2001) provide a useful first step in quantify metal loss from pressure treated wood subjected to acid rain. However, to adequately assess environmental effects, the transport and fate of the source contaminants must be taken into account.

Sieh (2001) used AAS to evaluate arsenic compounds in MSW ash. Interestingly, the arsenic and chromium averages in Florida MSW ash averaged 29.7 and 190.0 mg/kg – within the range of metal concentrations found in pristine North American soils. Even more interestingly, Sieh (2001) also noted that his and other TCLP test results for Florida MSW combustor ash were all well below the regulatory limit of 5 mg/L for arsenic or chromium. The same statement was true for TCLP concentrations of arsenic (range = 0.007 to 0.010 mg As/L) or chromium (range = 0.071 to 0.172 mg Cr/L) in a seven-year study by Wiles and Shepherd (1999). The FDEP (1999) found arsenic concentrations ranging from 0.020 to 0.500 mg/L and chromium concentrations ranging from 0.012 to 0.500 mg/L in TCLP tests for Florida MSW combustor ash. All of the values were much less than the regulatory limit of 5.0 mg/L for either compound.

Despite assertions by Dr. Solo-Gabriele and Dr. Townsend that CCA-C treated wood presents a threat to the non-hazardous designation of ash from Florida's MSW combustion program, Dr. Shieh and the references he cites suggest that arsenic and chromium concentrations in the ash have not been a threat to the ash's designation using either TCLP or SPLP testing procedures. Lead appeared to be a far greater problem in terms of exceeding regulatory thresholds for hazardous waste designation than any of the other tested compounds.

- **Synthetic Seawater.** Synthetic seawater lacks the organic constituents of natural seawater than bind metals. Having said that, it is considered an acceptable substitute for seawater for the purposes of determining metal loss from pressure treated wood. However, only CCA-C has AWPA (2001) approval for use in seawater. None of the alternative preservatives assayed by Townsend *et al.* (2001) are approved for marine use. Therefore, the comparisons have no practical application.

5. The aquatic toxicity tests conducted by Townsend *et al.* (2001) ignored the adverse biological effects associated with the unrealistic low pH resulting from the release of natural resin acids from ground wood in their leaching tests. The bioassays also ignored the low diluent hardness and alkalinity in all but the seawater tests. Paragraph 9.1.9 of EPA/600/4-90/027 (1991) notes that:

> "Mortality due to pH alone may occur if the pH of the sample falls outside the range of 6.0 – 9.0. Thus, the presence of other forms of toxicity (metals and organics) in the sample may be masked by the toxic effects of low or high pH. The question about the presence of other toxicants can be answered only by performing two parallel tests, one with an adjusted pH, and one without an adjusted pH. Freshwater samples are adjusted to pH 7.0, and marine samples are adjusted to pH 8.0, by adding 1N NaOH or 1N HCl dropwise, as required, being careful to avoid overadjustment."

Environment Canada's Pacific Environmental Science Center (PESC) Aquatic Toxicology Section routinely conducts a broad suite of fresh and saltwater bioassays. Their protocol for *Selenastrum capricornutum* (SOP ID.: SELAN20.SOP dated August, 1999) reflects the EPA caution regarding pH, but limits the range to 6.5 to 8.5. In all cases, bioassay protocols require recording and reporting of the pH at the beginning of a test, daily and at the test's termination in the control and all test concentrations (EPA/600/4-90/027). In this case, because the protocols caused artificially low pH in unbuffered deionized water, TCLP and SPLP solutions, a discussion of pH in the bioassays is considered critical and its absence is a major flaw in this part of the study.

Low water hardness and/or alkalinity can also stress bioassay test animals. The EPA criterion for alkalinity is 20 mg/l or more as $CaCO_3$ for freshwater aquatic life except where natural concentrations are less. The PESC bioassay protocol (SOP ID.: EC50DA10.SOP) for *Daphnia magna* requires that (Section 1.3.9), "If the hardness is less than 25 mg/L $CaCO_3$, it must be adjusted to a hardness of greater than 25 mg/L $CaCO_3$ (preferably between 40 – 50) according to stipulations recorded in the Biological Test Method (Reference Method EPS 1/RM/14 July 1990, amended May 96)." EPA/600/4-90/027, paragraph 10.2.1.1 recommends that alkalinity and total hardness be measured in the control and highest effluent concentrations at the beginning of the test and at test solution renewal. Lastly, the toxicity of copper is associated primarily with unbound cupric ions and perhaps CuOH. This is reflected in the U.S. EPA water quality criterion for copper in freshwater, which is dependent on water hardness. At a hardness of 0.5 mg/L (as $CaCO_3$), the EPA chronic copper WQC is 0.36 μg Cu/L – less than micronutrient requirements of most fauna and flora! In contrast, the chronic standard in moderately hard water (75 mg/L as $CaCO_3$) is 7.97 μg Cu/L.

Townsend *et al.* (2001) should have recognized the importance of hardness and alkalinity – especially in the *Ceriodaphnia* and *Selenastrum* tests and the stress associated with leachate final pH values in the range 4.42 to 5.83. Lower pH values were associated with untreated wood and CCA-C than with ACQ, CBA, CC or CDDC. It should be emphasized that the low pH and lack of hardness is the result of the choice of leachates (DI, SPLP and TCLP) together with the use of ground wood and that these conditions would not be found in real surface waters. Failure to report bioassay pH, hardness and alkalinity, and failure to discuss the implications of these factors on the bioassays invalidates the daphnid and algae results. A reasonable interpretation of this bioassay data with respect to the toxicity of treated SYP would have compared the bioassay endpoints in the untreated wood bioassays against those for treated wood. However, the results for untreated wood (Table IV.2) cannot be used as controls because none of the results meet EPA protocol requirements for control survival. For instance, EPA/600/4-90/027F (1993) requires 90% survival of *Ceriodaphnia dubia* in controls. The LC_{50} for this test organism reported in Townsend *et al.* (2001) occurred at between 22.3 and 34.4 percent of the untreated leachates. Under any circumstances, the study should have included a diluent control to assess the affects associated with using deionized water (instead of the recommended moderately hard water) or TCLP and SPLP fluids. This was not accomplished, making any interpretation of the bioassay results problematic.

In Section IV.2, Townsend *et al.* (2001) compared their bioassay results between treatments. Stress is cumulative in aquatic organisms, but the authors fail to discuss the stress associated with reduced pH and hardness. This is very evident in the results presented in Table IV.2., which reports EC_{50}, IC_{50}, and LC_{50} at between 6.2% and 66.4% of the **untreated** wood leachate for *Selenastrum* and low values of 22.3 to 34.4% for *Ceriodaphnia.* In the real world, large organic debris, in the form of logs, is placed in streams to improve habitat value and dissipate energy. The results of Townsend *et al.* (2001) would suggest this commonly employed habitat enhancement strategy is inappropriate, when in fact the bioassay results have no real world meaning because of the inappropriate protocols.

Since copper toxicity is a function of water hardness and Townsend *et al.* (2001) did not report hardness, it is difficult to understand the basis for the LC_{50}, IC_{50} and EC_{50} values used to construct Figure IV.6.

6. The minimum detection limits (MDL) were far too high for meaningful biological risk assessments. The minimum detection limits for arsenic (98 µg As/L), copper (74 µg Cu/L) and chromium (54 µg Cr/L) far exceed the 0.05 to 1.0 µg/L detection limits for arsenic or copper recommended by PSEP (1996) for freshwater. The recommended detection limit for ambient monitoring of chromium is 0.01 to 1.0 µg/L. For toxicity monitoring the recommended detection limits in PSEP (1996) are 10 µg/L for arsenic and 2.0 µg/L for copper or chromium. In PSEP (1996), EPA states that the upper range (1.0 µg/L) is, "The maximum recommendation based on routinely available GFAA analyses." It is uncertain why the copper detection limits reported by Townsend *et al.* (2001) were at least an order of magnitude higher than these recommendations or the U.S. EPA or Florida water quality criteria.

7. The mass percent of metals leached from CCA-C and CC treated wood reported by Townsend *et al.* (2001) demonstrates the unrealistic test conditions used in their research. Pressure treatment of wood in accordance with AWPA Standards provides protection for decades in real-world environments. The summary provided by Townsend *et al.* (2001) in Table III.10 indicates that an average of 6.15% of the copper was lost from CCA in 18 ± 2 hours of leaching in deionized water. That suggests that only five percent of the copper would remain in CCA-C leached in deionized water at the end of 33.4 days. A similar analysis indicates that only 5 percent of the copper would remain in CC preserved wood leached in deionized water at the end of only 4.42 days! These results obviously don't make sense in the real world. The reason that these results cannot be applied to the real world is that diffusion of metals across the face grain is slow and surface water pH is rarely less than 6.0. For these same reasons, AWPA Standard Method E11-97 (AWPA 1999) is used in studies addressing the efficacy of pressure wood products, but the small cubes of wood with high end-grain exposure are not appropriate for assessing aquatic risks.

8. Protocols designed for evaluating hazards in the waste stream are not appropriate for assessing aquatic risks. As previously noted, the questions being asked by Townsend *et al.* (2001) were not clearly defined. The authors used protocols designed for evaluating solid waste. These protocols obviously do not apply to typical surface waters in North America. In this chapter, Townsend *et al.* (2001) attempted to extend their results using solid waste protocols to real world aquatic environments. This author will state unequivocally that this is inappropriate. The following specific comments are provided to support this assertion:

8.1. Hazardous Waste Designation. The authors state that, "The fact that arsenic exceeds the TC limit even for DI water demonstrates that arsenic does indeed readily leach from CCA-C treated wood." This statement is far too subjective to be included in a scientific discussion. Brooks (2002) analyzed the data from several studies on CCA-C treated wood and determined the arsenic loss rate provided in Equation (1). In fresh water, this algorithm predicts an arsenic loss rate of 0.074 $\mu g/cm^2$-day following 30 days of immersion. In terms of the real world, a ten-foot long 2 x 6 discarded in a C&D landfill contains 10,892 cm^2 of leaching surface area and it would contribute 0.0008 grams of arsenic to the landfill each day –

assuming the landfill was flooded and the wood was continually immersed. In fact, landfills are not typically flooded and the wood is not be continually immersed, so my own analysis represents an unrealistic worst case. The known arsenic loss rate is a number that can be used to evaluate environmental risks. Whether or not the loss of 0.0008 g As/cm^2-day from a ten foot long 2 x 6 implies that "arsenic does indeed **readily** leach from CCA-C treated wood" is a matter of opinion – not a statement of fact supported by science.

$$\text{Eq. (1) Arsenic loss } (\mu\text{g As/cm}^2\text{-day}) = 0.010 * \text{Salinity(ppt)} + 0.754 * \exp^{(-0.130 * \text{days of immersion})}$$

Townsend *et al.* (2001) goes on to state that, "For many heavy metal-containing wastes (e.g. lead bearing wastes), the amount of metal that leaches under TCLP conditions is much greater than SPLP or DI conditions. This is not the case with CCA-treated wood, and the argument that TCLP may be too aggressive to simulate reality does not apply." The fact is that CCA-C treated wood taken out of service and delivered to C&D landfills is not pulverized to pass through a 9.5 mm sieve as required by the TCLP protocol and it certainly is not further pulverized to the <3.0 mm particle size required by Townsend *et al.* (2001). When asked whether or not Florida landfills (including C & D landfills) are polluting ground water, most scientists would review the literature and then ask to see groundwater monitoring well data from C & D landfills.

8.2. Contamination of groundwater. In assessing the risks to groundwater from CCA-C treated wood in C&D landfills, Townsend *et al.* (2001) state that, "When evaluating land disposal of wastes, it is customary to compare the results from batch leaching tests, such as the SPLP test, directly to the Groundwater Cleanup Target Level (GWCTL)." The GWCTL is now assumed to be 10 µg As/L. This assumption ignores factors, such as binding by sulfides in anaerobic environments (see Di Toro *et al.* 1992 for a general discussion; or Parris 2002). It also ignores the attenuation of metal concentrations by soils. Gifford *et al.* (1997) examined leachate quality in 36 field lysimeters. Twelve contained CCA-C treated wood chips (retention = 6 kg CCA/m^3) that had been weathered for two months. The chips overlaid a 25 mm thick layer of washed sand and a 200 mm layer of sandy/silt soil. One lysimeter contained only CCA-C treated wood chips (similar to the Townsend study) and eleven lysimeters contained untreated wood chips. Additional controls included 11 lysimeters containing only sand and soil and one empty lysimeter. The lysimeters were monitored bimonthly for 18 months. The soils were described as sandy/silt tephra (78.2 to 80.5% sand, 15 to 17.5% silt and 4 to 6.8% clay). They had low organic content (7.7 – 13.3%) and low cation exchange capacity (18 – 25 me/100g). The authors considered this study a worst-case examination of CCA treated wood removed from service in residential applications – such as decks.

The maximum concentration of arsenic in leachate from the lysimeter containing only CCA-C treated wood chips ranged between 0.05 and 0.98 mg/L, with most values in the 0.60 to 0.80 mg/L range. Most importantly, Gifford *et al.* (1997) found that the concentration of all metals was reduced by a factor of 25 after passing through 25 mm of washed sand and 200 mm of sandy silt. The maximum concentrations of Cu, Cr or As found in treatments which contained a layer of soil under the treated wood chips were approximately 40 µg/L with most concentrations near 10 µg/L (the new drinking water standard for arsenic). The following points should be emphasized:

- Gifford *et al.* (1997) examined the potential for groundwater contamination associated with a C&D landfill that contained only chipped CCA treated wood. Most landfills contain larger pieces of wood with lower leaching surface area per volume of wood.

- Gifford *et al.* (1997) observed a 25 fold decrease in Cu, Cr and As concentrations after passing through only 7.9" of sandy soil. It is my opinion that it would be irresponsible to place C&D debris over groundwater with only 7.9" of soil separation.

- In all cases the leachate from a chipped CCA pressure treated wood landfill underlain by 7.9" of sandy soil met the old U.S. drinking water standard of 50 μg As/L. Their graph suggests that the leachate met the new U.S. drinking water standard of 10 μg As/L after passing through the soil in nearly every case.

If there were no movement of groundwater under the C&D landfill, then there would be no human or environmental health risks unless someone drilled a drinking water well through the actual landfill. Assuming that groundwater is moving, then the leachate would be diluted. Based on the information provided in Figure (2) of Gifford *et al.* (1997), any dilution of the leachate would have resulted in arsenic concentrations below the new drinking water standard.

The binding of copper, chromium and arsenic in soils associated with CCA-C treated wood has been described by Cooper (1991, 1994) and Bergholm (1985, 1990) and many others. Bergholm (1990) examined metal loss from CCA treated wood and adsorption in a variety of soils. He found that arsenic concentrations were dependent on organic content, proportion clay, and the concentrations of amorphous iron, calcium and manganese. Sandy soils had a capacity to adsorb about 100 mg As/kg soils and clay soils adsorbed more arsenic (700 mg/kg).

Unless Florida is placing C & D landfills in direct contact with potable groundwater, the assumption of Townsend *et al.* (2001) that metal concentrations observed in the SPLP test are characteristic of those reaching groundwater are unfounded. The lack of an adequate literature search and discussion is a major flaw in Townsend *et al.* (2001). Had this been accomplished, the authors could have included this discussion in their interpretation of the data.

The question remains – are C & D landfills in Florida contaminating groundwater with arsenic? No one has better access to landfill monitoring well data that the Florida Department of Environmental Protection (FDEP). Parris (2002) has responded to a FDEP memo written by Mr. William Hinkley in which he notes the following:

- FEDP erroneously **excluded** all data in which arsenic was not detected in groundwater - significantly biasing their analysis. As noted by Dr. Parris, there is no basis for excluding non-detects from data. In my own analyses, I use a value equal to half the detection limit for non-detects (see Goyette and Brooks, 1999).

- FEDP combined the data from *Detection wells* located in the disposal area with data from *Compliance wells* located down gradient from the disposal area. A comparison of arsenic concentrations in *Compliance well* water with that found in *Background wells* located up gradient from the landfill would allow for an assessment of the contribution of arsenic to groundwater flowing under C & D landfills.

- The data reported by Parris (2002) indicated that only 45 of 246 monitoring well samples had detectable arsenic in them. The mean for all *Background* well samples was 2.37 µg As/L. When non-detects were included in the combined monitoring data for *Detection* and *Compliance* wells, the mean arsenic concentration was 2.65 µg As/L. The mean *Detection* and *Compliance* well arsenic concentration was increased by 0.28 µg/L above the mean found in *Background* wells. Parris (2002) concluded that the increase was not statistically significant. In any case, the mean arsenic concentration in *Detection* and/or *Compliance* well samples was well below the new US EPA drinking water standard of 10 µg As/L.

This discussion is not intended as a conclusive rebuttal of the Townsend *et al.* (2001) statement that, "These results imply that treated wood products may pose a threat to groundwater." However, the available evidence strongly suggests otherwise. A defensible analysis in Townsend *et al.* (2001) would have begun with a thorough literature review, continue with an assessment of soils underlying unlined C & D landfills in Florida and the use of any one of a number of models available for predicting the potential for groundwater contamination. The evaluation of CCA-C's threat to groundwater under C&D landfills would have been greatly improved by an analysis of actual monitoring well metal concentrations as a function of the presence or absence of CCA-C treated wood. Townsend *et al.* (2001) does not include this information and their assertion of a threat to groundwater posed by CCA-C treated lumber at C & D facilities is not substantiated. This statement of inadequacy also applies to the relative risks presented in Tables V.1 and V.2 of Townsend *et al.* (2001).

8.3. Section V.2 - Evaluation of regulations applicable during in-service use of treated wood. Townsend *et al.* (2001) evaluated the risks inherent in the use of wood preservatives on the basis of Florida's Soil Cleanup Target Levels (SCTL). The author of this critique does not question Florida's right to invoke any environmental standards it wishes. However, to be credible, Townsend *et al.* (2001) should have reviewed both natural background arsenic concentration in North America and they should have reviewed similar SCTL standards invoked by other jurisdictions.

The United States Geological Service (USGS) has a wealth of information describing ambient concentrations of a broad range of elements found in North American soils. Mean arsenic concentration in North American soils are summarized in Table (4) for a few states. In addition, it should be noted that mean soil arsenic concentrations in the 49 conterminous states plus Alaska have a range of mean values from 3.8 to 13.3 mg As/kg soil. The maximum observed soil concentrations, by state, ranged from 5.9 in West Virginia to 750 mg/kg in Alaska. The median maximum arsenic concentration by state was 18.2 mg As/kg. Please note that these data do not include agricultural soils, which generally contained elevated arsenic concentrations, and which were listed separately by Dragun and Chiasson (date unknown). Florida's residential SCTL is lower by a factor of 9 than the mean arsenic concentration found in the conterminous U.S., and the industrial SCTL is less than the lowest mean value found in soils of any state – including the mean of 4.7 mg As/kg reported for Florida. However, as previously noted, the choice of standards is up to Floridians.

Table 4. Soil arsenic concentrations in North America (mg arsenic/kg dry soil). These values do not include agricultural soil measurements which were listed separately in the source document (Dragun and Chiasson).

Area	Mean arsenic	Range of values
Conterminous U.S.	7.2	<1.0 to 97.0
Eastern U.S.	7.4	<1.0 to 73.0
Western U.S.	7.0	<1.0 to 97.0
Arizona	9.8	2.0 to 97.0
California	6.6	0.3 to 69.0
Florida	4.7	<1.0 to 52.0
Minimum mean in Maryland	3.8	
Maximum mean in Tennessee	13.3	

Maximum concentration from Alaska = 750 mg/kg.

Florida's Residential Soil Cleanup Target Level = 0.8 mg As/kg soil.
Florida's Industrial Soil Cleanup Target Level = 3.7 mg As/kg soil.

How do Florida's residential and industrial soil cleanup target levels compare with similar benchmarks invoked by other jurisdictions? Table (5) lists residential and industrial cleanup benchmarks invoked by some other jurisdictions. My point in this discussion is not to evaluate Florida's SCTLs, but rather to encourage Townsend *et al.* (2001) to review this issue in a broader and more relevant context because their report will be read outside Florida and may be misleading.

Table 5. Summary of residential and industrial soil arsenic benchmarks invoked by various government bodies. All values are in mg arsenic/kg soil. References to these benchmarks are available upon request.

Jurisdiction	Residential	Industrial
New Zealand Health and Environmental Guidelines	30 to 100	300
British Columbia Ministry of Environment, Lands & Parks	30	50
Government of Quebec	30	50
Washington State	20	200
Pennsylvania	12	53
Florida	0.8	3.7

Brooks (2002) used data from a variety of sources to estimate the long-term average arsenic loss rate from CCA-C treated wood at 0.074 µg As/cm^2-day. The 8' x 8' deck, modeled by Townsend *et al.* (2001) would require approximately 25 pieces of 2" x 6" x 8' long CCA-C treated wood (including deck supports on two-foot centers and facia on three sides).

Assuming that the boards are wetted on all sides during rainfall, this structure would have approximately 217,186 cm^2 of leaching surface area and would contribute approximately 0.016 grams of arsenic to the soils during each 24 hours that it is wetted. Assuming that the deck is wetted 15% of the time over an average twenty-year life span, the structure would lose a total of 17.6 grams of arsenic. Assuming the deck drains onto a sandy-loam soil, the arsenic would be distributed in the upper 6 cm of soil with a density of ca. 2.56 g/cm^3. The predicted increase in soil arsenic over 20 years would be 17.6 µg As/713,492 g soil or 24.7 µg As/g dry soil. The actual depth to which arsenic is distributed in soils depends on the soil grain size distribution with shallower depths and therefore higher concentrations in fine-grained soils (clay) and deeper penetration and lower concentrations in sandy or gravely soils. Using the methods of Townsend *et al.* (2001) for assessing risk, 24.7 µg As/g dry soil predicted under decks is less than some residential standards in Table (5) and about twice the 12 µg As/g standard set in Pennsylvania and less than the industrial standard for all listed jurisdictions – excepting Florida. I am unaware of any published empirical evidence supporting the value of 187 µg As/g proposed in Table V.3 by Townsend *et al.* (2001).

This assessment is not intended as a comprehensive examination of this issue. The discussion is provided simply to point out the lack of meaningful and unbiased analysis in Townsend *et al.* (2001) in light of other published information. From the perspective of a westerner, where soil concentrations of arsenic greatly exceed the Florida residential SCTL, the information presented in Table V.4 is simply nonsense. Townsend *et al.* (2001) are telling me that it is dangerous to allow children to play in western dirt. More specifically, should I require my grandchildren to wear gloves when they play in the dirt on my farm? The results presented in Table V.3 are based on a Florida residential SCTL that is only 11% of the mean arsenic standard found in North America and on an a methodology that does not realistically model metal losses from treated wood in C & D landfills. The result is that the soil arsenic concentrations predicted by Townsend *et al.* (2001) are grossly inflated when compared to values available in other published reports.

8.4. Section V.2.b - Aquatic uses. The use of regulatory standards as benchmarks for assessing aquatic risks is considered appropriate. However, it should be pointed out that all of the preservative retentions examined by Townsend *et al.* (2001) are for above ground uses. The authors did not examine wood treated to AWPA standards for soil or freshwater use – or for marine uses. Furthermore, excepting CCA-C, none of the preservatives tested by Townsend *et al.* (2001) are approved for marine use (C3 for piling or C18 for marine construction). In light of this, Townsend *et al.* (2001) did not explain why they applied the old marine water quality standard (2.9 µg Cu/L) to aquatic uses. Freshwater copper standards are dependent on water hardness and perhaps the authors realized the problems in the entire biological evaluation created by their use of leaching fluids lacking any realistic hardness. At any rate, if their studies were intended to evaluating aquatic risks, they should have examined metal losses from commodity size products into freshwater that is characteristic of Florida surface waters. This would have required use of moderately hard water (~75 mg/L as $CaCO_3$) having a freshwater chronic copper standard of 7.97 µg Cu/L.

The risk factors developed in Table V.6. have no meaning unless Townsend *et al.* (2001) propose that CCA-C treated structures be constructed of pulverized wood in Florida surface waters characterized by deionized water or SPLP or TCLP fluids. It is certain that structures will not be constructed of pulverized wood and I could find no data suggesting that

Florida surface water has pH values in the range of 4.0 to 5.0 or that their waters have hardness values < 1 mg/kg as $CaCO_3$.

It should be emphasized once again that none of the alternatives to CCA-C examined by Townsend *et al.* (2001) are approved by AWPA (2001) for use in marine environments. The marine alternatives to CCA are creosote or ACA and ACZA treated to 40 kg/m^3 (2.5 pcf) in the treated zone. All of the data collected for preservatives other than CCA-C by Townsend (2001) have no value in determining risks in marine environments. Furthermore, I can see no way to salvage any useful information from that data.

The question regarding relative risks associated with alternative preservatives in freshwater is an important one. If Townsend *et al.* (2001) had used appropriate protocols for determining metal loss rates from CCA-C, ACQ and Copper Azole into freshwater, they would have provided useful and timely information – particularly for Copper Azole and Copper Citrate. Unfortunately, the use of pulverized wood and low pH leaching fluids unrealistically exacerbates metal losses from treated wood and the use of static testing inhibits those losses. How these antagonistic errors affect the outcome is unknown because the different preservatives respond somewhat differently to these factors. This author has published algorithms describing metal loss data from commodity size pieces of wood treated with CCA-C (Brooks 2002) and ACQ-B (Brooks 1998) at AWPA specified retentions for use in freshwater. I do not have leaching data for ACQ-B in freshwater at the above ground retention used by Townsend *et al.* (2001). Metal losses from CCA-C are a function of water temperature. Unfortunately, Townsend *et al.* (2001) did not report the temperature at which their leaching studies were conducted. A temperature of 17.5 °C will be assumed. Table (6) compares the results of modeling copper losses from CCA-C and ACQ-B using algorithms developed by Brooks with the actual results presented by Townsend *et al.* (2001) in their Table V.8 for data from the SPLP test.

Table 6. Comparison of metal loss rates predicted by Brooks (1998, 2002) using the pH, time and retention given for the SPLP tests and results reported in Townsend *et al.* (2001)

	Brooks CCA	Townsend *et al.* CCA	Brooks ACQ-B	Townsend *et al.* ACQ
pH	4.500	4.500	5.250	5.250
Time (days)	0.375	0.375	0.375	0.375
Retention (kg/m^3)	4.202	4.202	4.540	4.540
Temp (°C)	15.000	Unknown	NA	NA
Cu loss (µg Cu/L)	7.66	3.300	49.333	14.000

Table (6) indicates that the results of Townsend *et al.* (2001) were lower by a factor of 2.32 than the predictions of Brooks (2002) for CCA-C made using the SPLP test endpoints. Similarly, the results of Townsend *et al.* (2001) were lower by a factor of 3.52 than predictions from Brooks (1998) for ACQ. Townsend *et al.* (2001) guessed at the actual leaching surface area in their study by assuming that the pulverized wood was a sphere with diameter = 1.5 mm or half the maximum allowable size. The shape of the wood particles was not evaluated nor was there any corroborating evidence for this guess, such as a particle size distribution or even the measurement of a representative subsample of the particles. As presented, the size assumption is at best simply a guess.

The Brooks algorithms predict 6.4 times more copper lost from ACQ compared with CCA during the first 18 hours of immersion. However, it must be emphatically emphasized that SPLP conditions are not appropriate for assessing either metal loss rates or risks in real aquatic environments. Brooks (1998, 2002) provides simple computer models, based on metal loss rates from commodity size wood treated to AWPA specifications and leached at realistic pH and temperatures. The author asserts that these data provide a far more realistic assessment of the relative and absolute aquatic risks associated with the use of these two products. Table (7) provides the results of comparing southern yellow pine lumber treated with ACQ-B and CCA-C to retentions of 6.4 kg/m^3 as required by AWPA Standard (C2) for soil and freshwater use. The predictions are for the first day of immersion in freshwater with pH = 7.0, temperature = 17.5 ^{o}C, hardness = 75 mg/L as $CaCO_3$. The short-term copper loss rate for ACQ-B is 8.17 times higher than for CCA-C. However, it should be pointed out that the "first flush" of metals from either of these products lasts for less than a week. The predicted long-term metal loss rates from ACQ-B and CCA-C are provided in Table (8). Townsend *et al.* (2001) used the methodology of Brooks (1998) to conclude that copper concentrations around piling that lose copper at the rates determined in their SPLP tests would not increase water or sediment concentrations of copper to levels of concern. This last conclusion is most important for assessing aquatic risks associated with any activity – including the use of pressure treated wood. The fact that ACQ-B initially loses 8 times more copper than CCA-C and that long-term copper losses from ACQ-B are five times higher than for CCA-C is irrelevant if neither preservative causes significant stress in typical applications. Similarly, the increased risks associated with arsenic released from CCA-C treated may be irrelevant if the sum of the sources of arsenic to the water result in concentrations less than the drinking water standard of 10 μg/L. Extensive use of the Brooks' models have shown that there are applications where

Table 7. Comparison of short term copper loss rates predicted by Brooks (1998, 2002) for CCA-C and ACQ-B preserved wood immersed in freshwater with retention = 6.4 kg/m^3, pH = 7.0 and temperature = 17.5 ^{o}C.

	ACQ	CCA
Copper loss (μg Cu/cm^2-d)	20.27	2.48
Arsenic loss (μg As/cm^2-d)		0.66
Chromium loss (μg Cr/cm^2-d)		0.09
DDAC loss (μg DDAC/cm^2-d)	16.90	

Table 8. Comparison of long term copper loss rates predicted by Brooks (1998, 2002) for CCA-C and ACQ-B preserved wood immersed in freshwater with retention = 6.4 kg/m^3, pH = 7.0 and temperature = 17.5 ^{o}C.

	ACQ	CCA
Copper loss (μg Cu/cm^2-d)	2.02	0.40
Arsenic loss (μg As/cm^2-d)		0.07
Chromium loss (μg Cr/cm^2-d)		<0.00
DDAC loss (μg DDAC/cm^2-d)	0.00	

the predicted copper concentrations exceed U.S. EPA water quality criteria during the first few days of immersion. However, those projects usually involve extensive bulkheads located in poorly flushed residential canals. Similarly, the use of ACQ-B typically poses little or no risk to most aquatic environments. Small to moderate size projects constructed using wood preserved with either of these preservatives poses minimal risk to aquatic environments.

Table (9) provides water quality criteria, and predicted concentrations of copper, chromium, arsenic and DDAC within a few centimeters of a 9.5” x 9.5” post treated with CCA-C or ACQ-B to a retention of 6.4 kg/m^3 and immersed in water flowing at 0.5 cm/sec (a swamp or small pond). The US EPA water quality criteria for copper, chromium and arsenic are provided to assess the environmental risks. The DDAC benchmark was developed by Brooks (1998) for assessing the aquatic risks associated with this organic component of ACQ-B.

Table 9. Comparison of predicted water column concentrations of copper, chromium, arsenic and DDAC in fresh water flowing next to a 12” square post treated with CCA-C or ACQ-B and immersed in fresh water with pH = 7.0, hardness = 75 mg/L as $CaCO_3$ and temperature = 17.5 ^oC on the first day of immersion in water flowing at 0.5 cm/sec. All values are micrograms per liter (parts per billion).

	ACQ	CCA	Water quality criterion
Copper loss (µg Cu/cm^2-d)	1.47	0.18	7.97
Arsenic loss (µg As/cm^2-d)		0.05	190.00
Chromium loss (µg Cr/cm^2-d)		0.01	163.53
DDAC loss (µg DDAC/cm^2-d)	1.23		49.00[1]

[1] Benchmark adopted by Brooks (1998) for assessing aquatic risks associated with the use of ACQ-B pressure treated wood.

The long-term water column concentrations of metals have been shown to be so low as to be indistinguishable from background – even adjacent to the very large projects constructed in worst case environments reported by Brooks (2000a and 2000b). However, consistent with this discussion and the known metal loss rates from these two preservatives, Lebow (2000) and Brooks (2000b) did observe exceedances of the U.S. EPA copper water quality criterion immediately adjacent to the ACQ-B treated boardwalk for the first 162 days following construction in the Wildwood Wetland Boardwalk study. Similar exceedances were not observed at the CCA-C boardwalk for any of the metals at any time. It should be emphasized that no adverse effects were observed in the aquatic insect community associated with any of the treatments examined in Wildwood. All of the preservatives were acceptable from a biological point of view. However, as predicted by leaching studies and the models discussed above, the risks were higher for ACQ-B compared with CCA-C.

Summary. Different scientists approach studies in different ways. It is my belief that every good study begins with specific and carefully thought out questions. Carefully posed questions are followed with a thorough literature review to gain an understanding of the available information. That avoids mistakes that others have made. It eliminates the duplication of work already accomplished and it capitalizes on proven techniques. The third step is to design a defensible set of protocols that are specific to the questions being asked. Data are collected only after all of these preliminary steps are complete and after the study design has been reviewed by peers in search of flaws. I could not identify these necessary steps in the Townsend *et al.* (2001) report. The following comments summarize my concerns with Townsend *et al.* (2001).

- The data's interpretation suggests that the authors were interested in comparing the environmental risks to ground and surface water associated with available waterborne preservatives. But that is not specifically stated. If this was the goal, then why did the authors use wood that was not treated to retentions required by AWPA standards (C2, C3 or C18) and why did the authors test preservatives in saltwater that are not approved for that application? Lastly, the inclusion of a preservative (CDDC) that is no longer produced or marketed is confusing. The authors' recommendation of CDDC as an alternative to CCA-C is even more perplexing.

- In contrast, the leaching of pulverized wood in deionized water, SPLP and TCLP leaching solutions suggests that the authors were interested in what would happen if they landfilled pressure treated sawdust. The particle size of their pulverized wood was only one third (3 mm) that of the sieve size recommended for TCLP studies (9.5 mm). If the authors were truly interested in determining the loss of metal from pressure treated wood sent to C & D landfills, then they should have gone to a typical landfill, retrieved some of the wood placed there and examined metal losses from that wood to include wood of a size typically found in the landfill.

- In any case, it is unexplained why the authors conducted bioassays on the deionized water, SPLP and TCLP leachates, which had near zero hardness, and whose pH's were further reduced, by natural wood resins, to values that are stressful to nearly all aquatic organisms.

- If the authors were interested in comparing metal losses from different types of treated wood, then they should have insured that the leaching conditions were the same for all treatments. In this case the pH was consistently lower for CCA-C treated wood than for the other treatments. That confounds any comparison between treatments. Natural environments all have some buffering capacity and most surface water has a significant buffering capacity.

- The authors presented no background information demonstrating a familiarity with the existing literature. It is obvious in reading their report that they could have avoided the many mistakes made in designing and interpreting their studies if they had spent a few months reviewing and understanding the tens of thousands of pages of existing reports applicable to the questions being asked.

Following the collection of data in any well-done study, the authors compare their results with the existing literature. Sometimes these comparisons point out significant differences and inconsistencies that should present red flags to the researcher. Inconsistencies can mean that the research has uncovered something new and exciting. More frequently, the inconsistencies lead to the illumination of flaws in the study design, collection of data, or its interpretation. This is particularly true of laboratory studies that attempt to mimic the real world. Empirical evidence from the real world should always be given more credibility than evidence from laboratory studies or theory. In this case, a literature review would have revealed the significant adsorption capacity of even sandy soils overlying groundwater. The use of any of the numerous transport models would have revealed significant metal attenuation as landfill leachates percolate downward through soils. Lastly and most importantly, the authors should have examined ground water monitoring well data from Florida C & D landfills where arsenically treated wood has been deposited. The available monitoring well data from Florida landfills does not support the authors' assertion that arsenic from CCA-C treated wood is jeopardizing groundwater under these facilities. Granted, Florida's sandy soils may exacerbate this problem in some areas. However, as seen in this review, the lining of C & D landfills with even 8" of sandy soil containing 20 percent fines (silt and clay) could be expected to adsorb most of the arsenic in leachate percolating downward.

This critique is not intended as a thorough examination of all of the untended issues associated with Townsend *et al.* (2001). It is intended to point out the serious flaws in the study design and in the interpretation of their results. Those flaws have occurred because the authors failed to carefully define the questions they were asking; they failed to use appropriate protocols for determining affects in surface or ground water; and they failed to utilize a voluminous literature in designing and interpreting their results.

This author is always looking for good information to help understand and manage the environmental risks associated with the use of pressure treated wood. The metal loss data from the SPLP tests will be useful for evaluating copper loss from acid rain exposed wood and I recommend that information be published separately as a data report. Unfortunately, I was unable to find other information in Townsend *et al.* (2001) that is useful in understanding these preservatives in the real world.

For those reviewers interested in preservative loss rates from pressure treated wood, data are being currently being developed from commodity size pieces of wood at ambient pH (7.8) and hardness (50 mg/L) in dynamic test cylinders. The results for ACQ-D and Copper Azole will likely be available within one year.

References

Bergholm, J. 1985. Leaching of copper, chromium and arsenic from preserved wooden chips deposited in the soil. Swedish Wood Preservation Institute Report Number 154. 111 pp.

Bergholm, J. 1990. Studies on the mobility of arsenic, copper and chromium in CCA contaminated soils. IRG/WP/3571. International Research Group, Stockholm, Sweden

Brooks, K.M. 1996a. Evaluating the environmental risks associated with the use of chromated copper arsenate-treated wood products in aquatic environments. Estuaries Vol. 19, No. 2A, pp. 296-305.

Brooks, K.M. 1997a. Literature review and assessment of the environmental risks associated with the use of CCA-C treated wood products in aquatic environments. Western Wood Preservers Institute, 7017 NE Highway 99, Suite 108, Vancouver, WA 98665. 62 pp.

Brooks, K.M. 1997b. Literature review and assessment of the environmental risks associated with the use of ACZA treated wood products in aquatic environments. Western Wood Preservers Institute, 7017 NE Highway 99, Suite 108, Vancouver, WA 98665. 62 pp.

Brooks, K.M. 1998. Literature review and assessment of the environmental risks associated with the use of ACQ-B treated wood products in aquatic environments. Western Wood Preservers Institute, 7017 NE Highway 99, Suite 108, Vancouver, WA 98665. 95 pp.

Brooks, K.M. 1999. Recommendations to the National Marine Fisheries Service for the use of CCA-C, ACZA and creosote treated wood products in aquatic environments where threatened or endangered species occur. Prepared for the Western Wood Preservers Institute, 7017 NE Highway 99, Suite 108, Vancouver, WA 98665. 32 pp.

Brooks, K.M. 2000a. Environmental impact of preservative-treated wood in a wetland boardwalk: Part II. Environmental Effects. Research Paper FPL-RP-582. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 126 pp.

Brooks, K.M. 2000b. Assessment of the environmental effects associated with wooden bridges preserved with creosote, pentachlorophenol, or chromated copper arsenate. Research Paper FPL-RP-587. 100 pp.

Brooks, K.M. 2002. Characterizing the environmental response to pressure treated wood. Forest Products Society Meeting. February 11 – 13, 2002. Available on the FPS website.

Bull, D.C. 2001. The chemistry of chromated copper arsenate II. Preservative-wood interactions. Wood Science and Technology, Vol. 43, pp. 459 – 466.

Cooper, P.A. 1991. Leaching of wood preservatives from treated wood in service. Report prepared for Public Works Canada by Paul A. Cooper, University of Toronto, Canada, January 1991.

Cooper, P.A. 1994. Leaching of CCA: is it a problem? In *Environmental considerations in the manufacture, use and disposal of preservative-treated wood.* Forest Products Society, Madison, WI, USA, pp. 45-57.

Di Toro, D.M., J.D. Mahony, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. ankley. 1992. Acid Volatile Sulfide Predicts the Acute Toxicity of Cadmium and Nickel in Sediments. Environmental Science and Technology. Vol. 26, pp. 96-101.

Florida Department of Environmental Protection. 1999. solid Waste Management in Florida, July 1999. Division of Waste Management, Bureau of Solid and Hazardous Waste. Pp. 51-68.

Gifford, J.S., N.A. Marvin and P.H. Dare. 1997. Composition of Leachate From Field Lysimeters Containing CCA Treated Wood. Procedings of the American Wood-Preservers' Association, Vol. 93 pp. 292 – 306. April 27 – 29 Grandbury, Texas.

Goyette, D., and K.M. Brooks. 1999. Creosote evaluation: phase II. Sooke Basin study – baseline to 535 days post construction 1995 – 1996. Environment Canada, North Vancouver, British Columbia, Canada 568 pp.

Jin, L. and K. Archer. 1991. Copper Based Wood Preservatives: Observations on Fixation, Distribution and Performance. Pro. Am. Wood-Preservers' Assoc. pp. 169 – 184.

Lebow, S.T. 1996. Leaching of Wood Preservative Components and Their Mobility in the Environment – Summary of Pertinent Literature. General Technical Report FPL-GTR-93. Madison, WI: U.S. Department of Agriculture, Forest Service, Forest Products Laboratory. 36 pp.

Lebow, S.T., D.O. Foster and P.K. Lebow. 1999. Release of copper, chromium, and arsenic from treated southern pine exposed in seawater and freshwater. Forest Products Journal, Vol. 49, No. 7-8, pp. 80-89.

Parris, G.E. 2002. Comments on Disposal of CCA-(Arsenically-) Preserved Wood with Special Reference to the Situation in Florida. Prepared by George E. Parris, Director of Environmental and Regulatory Affairs, American Wood Preservers Institute. Washington D.C.

Pizzi, A. 1981. The chemistry and kinetic behavior of Cu-Cr-As/B wood preservatives. I. Fixation of chromium on wood. J. Polymer Science, Chem. Ed., Vol. 19. pp. 3093-3121.

U.S. Environmental Protection Agency. 1994. *Short-term Methods for Estimating the Chronic Toxicity of Effluents and Receiving Water to Freshwater Organisms.* Office of Research and Development, Washington D.C.

U.S. Environmental Protection Agency. 1996. *Test Methods for Evaluating Solid Waste, SW846, 3 red edition.* Office of Solid Waste and Emergency Response, Washington D.C.

Shieh, C.-S. 2001. Criteria of Selecting Toxicity Characteristic Leaching Procedure (TCLP) and Synthetic Precipitation Leaching Procedure (SPLP) Tests to Characterize Special Wastes. Florida Institute of Technology, 150 W. University Blvd., Melbourne, FL 32901. 20 pp.

Wiles, E. and P. Shepherd. 1999. Beneficial Use and Recycling of Municipal Waste Combustion Residue – A Comprehensive Resource Document, National Renewable Energy Laboratory, Colorado. 126 pp.