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Composting toilets a misnomer: Excessive ammonia from urine inhibits microbial activity yet is insufficient in sanitizing the end-product

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ABSTRACT

End-product from 16 public mixed latrine style composting toilets (CTs) at 12 sites between 50 and 2100 m.a.s.l. in Western North America was tested in order to evaluate the effect of composting variables (TS%, NH_3–N, temperature, and material age) on compost quality and hygiene (VS%, *Escherichia coli*, NO_3^-–N, and pH). Principal component analysis indicated that TS%, temperature, and material age equally contributed to reduction in VS%. NH_3–N had the greatest effect on NO_3^-–N, *E. coli*, and pH. Nitrification was significantly inhibited above 386 mg/kg NH_3–N, but no such limit was found for *E. coli*, despite a significant ($p = 0.016$) but weak ($r^2 = 0.11$) negative relationship. It may be possible to amplify the sanitizing effect of ammonia and overcome pathogen resistance due to low temperatures and re-contamination (caused by poor design) with generous dosing of urea and ash. However, even sanitized, the fertilization effect of discharged material on the natural environment may not be desired or permitted in parks or protected areas where many CTs were found. To this end, operators of CTs need to evaluate their primary management objectives and ensure congruency with proven system capabilities.

1. Introduction

Composting is the managed aerobic decomposition of organic waste into stable, mature, and sanitized end-product low in contaminants and foreign matter, which would not cause deleterious environmental impacts if land applied (Haug 1993, Wichuk and McCartney, 2010). In order to develop end-product material that meets this definition and passes relevant jurisdictional standards, feedstocks are conditioned and the process managed to induce a rapid temperature rise, which stimulates thermophilic microbial consumption of organic matter. To sustain thermophilic composting, organic matter must have an appropriate ratio of biodegradable carbon and nitrogen (~30/1) (Kayhanian and Tchabanoglous, 1992) despite the consumption of carbon, oxygen and water; all of which must be continuously available or replenished through forced aeration, periodic mixing and watering in order to prevent process inhibition and premature cooling (Haug 1993). Temperatures are expected to reach 55 °C or more for three days to three weeks (depending on which composting process is used) to kill and adequately sanitize pathogens (CCME, 2005; British Columbia Regulation 198, 2007). The World Health Organization (WHO) guideline recommend that composting of toilet waste should be performed at 50 °C or higher one week to month followed by two to four months curing time (WHO, 2006). Once the majority of rapidly degradable organic matter has been consumed the rate of oxidation drops, heat production slows, and the curing phase begins. This phase is less actively managed and is characterized by mesophilic microorganisms such as fungi and bacteria including nitrifiers, which convert remaining ammonium to nitrate, an essential process in the production of mature compost.

Composting toilets (CTs) are used in North America for the decentralized, waterless, treatment of human waste despite the WHO (2006) recommendation that the difficult process of fecal matter composting be conducted off-site as a centralized secondary treatment. CTs are commonly perceived and advertised as being capable of producing 'compost' onsite, a notion, which may be

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traceable to product literature. As a result, the disposal/land application of untested end-products into public park environments is prevalent. The objectives of nutrient reclamation and organic matter re-use add complexity to the primary objective of material sanitation, which is itself difficult to accomplish (Cilimburg et al. 2000). Numerous composting toilet studies indicate a failure to produce sanitized material let alone stable and mature compost low in foreign matter as defined above due to a variety of causes including: poor design, overuse, insufficient maintenance, low temperatures, anaerobic conditions, and excessive urine (Matthews, 2000; Redlinger et al. 2001; Holmqvist and Stenstrom, 2002; WHO, 2006; Tønner-Klank et al. 2007; Jensen et al. 2009; Hill and Baldwin, 2012). Land application of 'compost' failing to meet standards can result in pathogen transmission, eutrophication of aquatic ecosystems, and phytotoxic impacts (Wichuk and McCartney, 2010) and should be removed to appropriate treatment facilities according to most regulations pertaining to public operators on publically accessible land in North America (WSDOH, 2007). This can be labor intensive, offensive, expensive, and dangerous and at remote sites (Hill and Henry, in press; Hill and Baldwin, 2012).

The following explanatory factors have been explored in narrow CT field studies and laboratory experiments: operations and usage by Matthews (2000) and MWH (2003); moisture content by Zavala and Funamizu (2005), Tønner-Klank et al. (2007), Redlinger et al. (2001) who determined that 40%TS was optimal, below which anaerobic conditions developed and sustained pathogens; thermodynamics and temperature by Chapman (1993), Holmqvist and Stenstrom (2002), and Zavala et al. (2004) who reported that most in-field CTs operated at or near ambient air temperatures; storage time by Gibbs et al. (1997), Guardabassi et al. (2003), Vinneråas (2007), Jensen et al. (2009), and Sherpa et al. (2009), all of whom found storage time alone unreliable in the destruction of pathogens; and feedstock conditions by Chapman (1993), Vinneråas et al. (2003), Tønner-Klank et al. (2007), Niwagaba et al. (2009) each of whom showed that the addition of food-waste or diversion of urine can improve decomposition. But as far as the authors know, a comprehensive exploration of root causes of failure from in-field public, mixed latrine style microbial composting toilets (MLMCs) has not been conducted in North America.

We apply multivariate statistics to a comprehensive data set of end-product quality and process variables from public and in-field MLMCs in Western North America in order to evaluate underlying causes of variability and those most impactful on compost quality. By isolating consistent root causes of system failure: the management of in-situ systems could altered for improved sanitation and end-product quality; the most appropriate new sites can be chosen for systems currently on the market; and adaptations and advancements in product designs can be stimulated.

Based on the literature and variables measured in our study the following key variables (and their impact on the compost process) were chosen: TS% (moisture and ability to deliver oxygen); material age (residence time within treatment system); ambient site temperature (rate of biochemical reaction); and ammonia concentration (urine content). Compost quality was indexed by pH (general quality), nitrate (maturity), VS% (stability), and *Escherichia coli* (pathogen content).

2. Methods

2.1. Sites

Agencies operating public mixed latrine style composting toilets in Washington, USA, British Columbia, Alberta, and Northwest Territories, Canada were contacted requesting permission to extract samples of end-product for analysis. All those granting permission were visited. Twelve sites, with 16 chambers in total, were visited between 2009 and 2011. Nine were found in remote national, provincial, and regional park sites. Two were found in public buildings; only one site was housed within a heated utility room. All toilets sampled were commercial units, sized and installed professionally. Despite some differences in tank size all systems were used and maintained in a similar fashion by each agency according to operational manuals provided at the time of purchase. Sites were found scattered between 50 m and 2100 m elevation and between 46°N and 61°N. The sites received 500–45,000 uses per year per toilet with a concentration of usage in summer months and minimal usage in the winter months except at the toilet within the public building where usage was more consistent throughout the year. A summary of site characteristics can be found in Hill and Baldwin (2012).

2.2. Collection and maintenance

Both fecal matter and urine are collected through a single toilet hole. Pine shavings or peat moss bulking agent (40–200 ml) were added each use along with toilet paper. Site operators performed weekly and monthly maintenance according to the CT manufacturers' instruction manuals. During maintenance additional bulking agent was added if the pile was too wet, a judgment likely to differ considerably by operator. When a chamber filled up, end-product was removed from the bottom. A description of compost toilet chamber design and characteristics can be found in Hill and Baldwin (2012).

New chambers were started 2/3–3/4 full with bulking agent. Depending on use, chamber size, and operational procedures, this bulking agent will dominate the material removed for 1–8 years before true 'end-product' (fecal matter, trash, 'compost') could be observed.

2.3. Samples

Only samples from the oldest end-product in each chamber ware investigated. The material sampled was deemed 'finished' end-product as all material was older than six months and as old as eight years, which is in accordance with NSF/ANSI Standard 41 where testing of end-product is made after six months of system operation. Not all samples were tested for the complete suite of chemical analyses, resulting in minor variations in sample size by assay.

Two to five replicate grab samples were extracted from each compost chamber during 21 site-chamber visits, with a gloved hand from the oldest sections of the material pile according to NSF/ANSI Standard 41 (2011). Samples were extracted from the grab sample directly with a sterile 200 ml glass sample jar. Samples were placed into sterile glass jars in a cooler with ice packs for overnight transport by courier to the commercial laboratory for analysis. In the majority of cases samples were received by the laboratory within 48 h of sampling and a minority in 72 h.

2.4. Biochemical analyses

Benchmark Labs in Calgary, Alberta, an ISO 17025 accredited Lab, analyzed solid end-product samples according to Table 1 (Table 2).

2.5. Statistics

JMP version 8 (SAS 2009) was used to: perform Principal Component Analyses, univariate ANOVA tests when assumptions

Table 1
Parameters tested by Benchmark Laboratories, Calgary, in the evaluation of compost end-product stability, maturity, hygiene, and general quality.

Parameter	Test name/Description	Indicative of:	Units
Annual average ambient site temperature (temperature)	Extrapolated from nearest climate station and adjusted for elevation (1 °C/100 m)	Pile temperatures which seldom rise above ambient temperatures	°C
Percent total solids (TS%)	APHA Method 3540B	Moisture content Diffusion of oxygen	%
Material age	Estimated through interviews with operational staff	Residence time	Years
Uncharged free ammonia (NH_3)	Cold-water-shake 1:2 sample:water, followed by measurement with a Thermo Scientific Orion high performance ammonia ion electrode at 20 °C according to manual instructions for free ammonia concentrations of >1 ppm	Urine	mg/kg (ds)
pH	Cold water shake 1:2 sample:water, followed by measurement with VWR symphony pH probe at 25 °C	General quality	–
E. coli	Cold water shake extraction followed by USEPA Approved Method 1604, with only *E. coli* reported by membrane filtration using a simultaneous detection technique	Pathogen content	CFU/g (ds)
Nitrate (NO_3)	APHA Method 4110A	Maturity	mg/kg (ds)
Volatile solids (VS%)	APHA Method 2540	Stability	%

validated; Wilcoxon non-parametric statistics when parametric assumptions were not met; and for all graphical displays. Means and standard deviations are reported in text and in graphical displays. When graphically displayed, *E. coli* was $\log_{10}$ transformed and fitted lines were similarly $\log_{10}$ transformed along the *E. coli* axis. One outlier was removed from the ANOVA of toilet placement by *E. coli* in the trail group.

3. Results

Samples from brands B and D were younger (average 0.5 and 1 yr respectively) than brands A and C (averaging 4.0 and 2.6 yr, respectively) ($p < 0.05$), but this was expected due to the smaller size of brands B & D (<0.5 m^3) than A & C (>2 m^3). Material was also wetter in brands B & D than brand A, but the magnitude of the difference was not large enough to have induced a functional difference as all would be considered wet (i.e. TS equal to 22%, 16%, and 26% respectively). Brand D (−2 °C) was installed in a significantly lower ambient temperature environment than brands A, B & C (4, 7, 10 °C). However, despite this, no difference was found between brands in end-product quality (pH, VS%, NO_3^-–N, *E. coli)* or NH_3–N ($p > 0.05$). Samples were grouped together by brand. A summary of parameter values by site can be found in the supplemental material and in Hill and Baldwin (2012).

Fig. 1A displays an ordination of the controlling variables and the compost quality variables together. The relationships amongst the controlling variables are preserved in comparison to when the controlling plots are ordinated on their own (see Supplemental material). Eigenvalues 1 through 7 are statistically significant (all p-values < 0.019) and together explained 99% of the variation in the data set. The first two eigenvalues explained 56% of the variation in the data. Material age and temperature can be seen acting positively together and having similar correlation (0.4) and each negatively correlating to VS% (−0.3 and −0.5 respectively). Samples are marked by toilet placement as trail toilet samples cluster away from camp and commercial sites positively by Component 1 (Fig. 1B). The majority of samples in the lower left quadrant are <3 yrs old and in the upper right corner are >5 yrs old (Fig. 1B). Of the four explanatory variables, NH_3–N had the highest univariate correlations with pH and *E. coli* and the second highest correlation to NO_3^-–N after temperature.

Univariate relationships between NO_3^-–N and TS%, VS%, NH_3–N, pH, temperature, and material age were inspected and tested with linear regressions. Significant negative correlations were found between NO_3^-–N and NH_3–N ($p = 0.0054$) (Fig. 2A), and temperature ($p = 0.032$) (not shown) but neither relationship were well described by a linear function. High NO_3^-–N was observed only when NH_3–N concentrations were low suggesting a possible threshold inhibiting nitrification as discussed in Section 4. Significantly more nitrate was found in samples where ambient temperatures were >8 °C (3589 ± 3807 mg/kg, $n = 7$, $T_{avg} = 13.0$ °C) than in samples <8 °C (547 ± 1943 mg/kg, $n = 40$, $T_{avg} = 2.4$ °C) ($p = 0.0022$). The only significant and meaningful correlation was found only between NO_3^-–N and pH ($p < 0.0001$) (Fig. 2B).

Toilets placed at trail-head locations had significantly higher nitrate production (5058 ± 3326 mg/kg) than commercial (4.56 ± 9.36 mg/kg) and camp (251 ± 1394 mg/kg) locations (Wilcoxon $p < 0.0001$) and lower ammonia content (195 ± 254 mg/kg) than campground locations (951 ± 748) toilets (Wilcoxon $p = 0.0011$) (refer to the Figures in the Supp. Info.).

Significant negative relationships were found between numbers of *E. coli* and NH_3–N, TS%, and pH but not with annual average site temperature, VS%, or material age (Table 3). All were weak fits having r^2 values below 0.2 (refer to the Figures in the Supp. Info.).

Table 2
Correlations between NO_3^-–N (a compost maturity index) and TS%, VS%, NH_3–N, pH, temperature (°C), and material age (years), in end-product samples from public mixed latrine style composting toilets.

NO_3 (mg/kg)	TS%	VS%	NH_3	pH	Temperature (°C)	Age (years)
Correlation	−0.12	−0.10	−0.40	−0.63	0.31	0.17
p value	0.43	0.49	0.0054	<0.0001	0.032	0.26

4. Discussion

Despite minor differences in brand, the basic design, operation, feedstock, and amendments utilized by all brands were similar (Hill and Baldwin, 2012). All funneled urine and feces into the same chamber, recommended the addition of carbonaceous bulking agent such as pine shavings (majority) or peat moss (one location), and required frequent mixing by shovel, rake, or a built-in mixing device. It may be possible to tease out significant difference with larger sample sizes, but the relevant magnitude of the results are unlikely to be meaningful in comparison to relevant standards as discussed by Hill and Baldwin (2012) where all MLMC sites tested failed NSF/ANSI Standard 41.

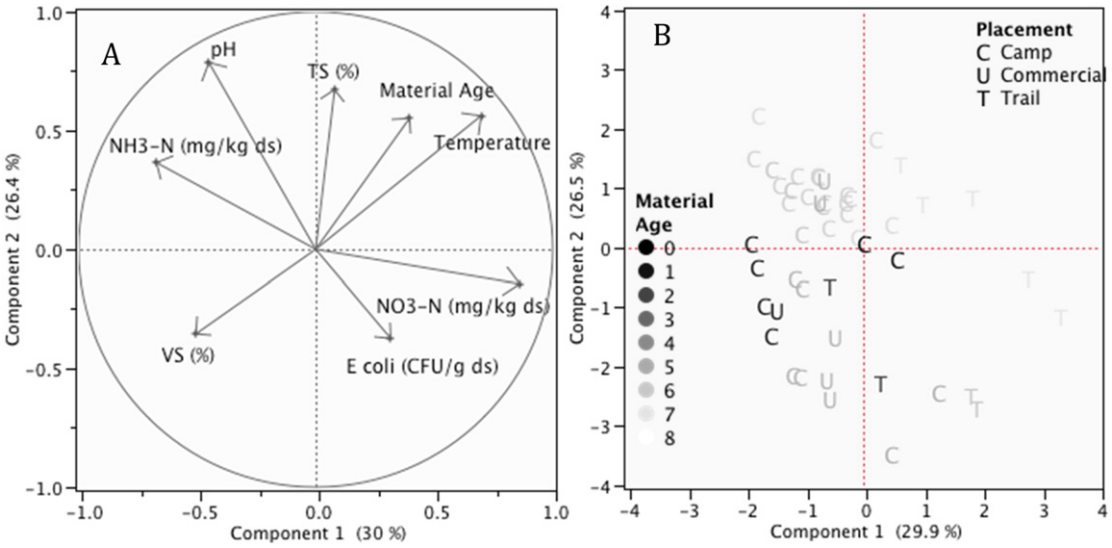

Fig. 1. PCA plots of the dominant variables together with compost quality variables in public mixed latrine style composting toilets (A) and samples plot location relative to PCA components (B) marked by material age and toilet placement as per legend.

To obtain highest stability and most degraded end-product, samples should land in upper right quadrant of Fig. 2B; this zone is opposite from the VS% vector (lowest VS%) and inline with the three main controlling variables (temperature, age, and TS%). These relationships were expected; greater decomposition can occur when organic matter is processed at a higher temperature, for longer periods of time, in the absence of anaerobic conditions (Haug 1993).

Thermophilic composting for days to weeks at temperatures of 55 °C or greater generally ensures simultaneous pathogen destruction and stabilization, which leads to the development of mature compost (Haug 1993). However, these temperatures are rarely attained in in-field public CTs (Chapman, 1993; Guardabassi et al. 2003; Jenkins, 2005) or in urine diverting CTs (Peasey, 2000; Hurtado, 2005) and were also not measured at any locations here. Even when thermophilic temperatures have been attained, often through diversion of urine, inclusion of readily degradable food waste, auxiliary heat, or pH adjustment, heterogeneity in the material can leave pockets of insufficiently heated material which could potentially harbor pathogens (Guardabassi et al. 2003; Vinnerås, 2007; Niwagaba et al. 2009). Hot air panels were the only functional auxiliary heaters found at the remote sites in this study, but it is possible that the small amounts of heat energy introduced are lost from the pile through evaporation of water into the dry air (Chapman, 1993). Thermophilic temperature treatment is an integral component of most definitions of the composting process; therefore it could be said that the use of the term "composting" in the MLMCs examined in this study is inappropriate. *Composting* toilets are commonly referred to as *dry* toilets in Europe due to the absence of flush water addition; adopting this term in North America would help minimize any misunderstanding around the capabilities of these waste management systems. Despite the inability of CTs to attain adequate temperatures for sanitation and thermophilic microbial communities, there remains a positive effect of temperature on VS% reduction, reflecting the increase in biochemical reaction rates, which result from elevated temperature.

A downside of adding dry bulking agents to absorb moisture and elevate TS% is reduction of the volume fraction of fecal material processed resulting in lower residence times within the composting chamber. Despite many samples comprised mostly of bulking agent, the mean TS% (24%) was still much lower than optimal (40%) (Redlinger et al. 2001; Zavala and Funamizu, 2005), suggesting that the addition of bulking agent is largely ineffective at elevating TS% to optimal levels. Water content in the pile could theoretically be reduced by placing the toilet in a location where it receives limited urine, such as on a trail, rather than in a campground. However,

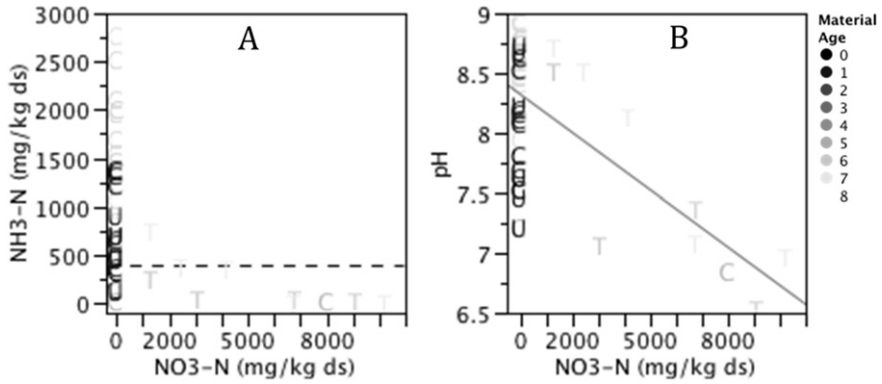

Fig. 2. Nitrate concentration from public mixed latrine style composting toilet end-product plotted against NH_3–N (A) and pH (B). The dotted line in A is at 386 mg/kg (ds) NH_3–N; the proposed upper limit below which significantly more NO_3^-–N is found than above ($p < 0.05$).

Table 3
E. coli (CFU/g) in end-product from public latrine style composting toilets linearly regressed with ammonia, pH, and TS%.

E. coli (CFU/g) regressions	*p* value	r^2	Regression equation	*n*
NH_3 (mmol/l)	0.015	0.12	=20.2 − 1.5*Log (*E. coli*)	47
pH	0.0096	0.14	=8.8 − 0.1*Log (*E. coli*)	47
TS%	0.03	0.089	=29.0 − 0.6*Log (*E. coli*)	53

there was no difference in TS% by toilet placement ($p = 0.26$) indicating that water content is not a function of placement regardless of any potential differences in use or urine use.

Urine diversion has also been discussed as an important modification to CTs in reducing excessive moisture and smell, encouraging decomposition, and for easier nutrient recapture (Chapman, 1993; MWH, 2003; Jönsson and Vinnerås, 2007; Niwagaba et al., 2009; Nordin et al. 2009a). With less urine the ammonia concentration and pH should drop, which, according to Fig. 1A, would result in higher nitrate. Nitrate is produced by obligate aerobic lithotrophic nitrifying bacteria, which are out-competed for ammonium by heterotrophs (Haug 1993). Nitrification of ammonium to nitrite by *Nitrosomonas* spp. can also be inhibited by free ammonia starting as low as 16 mg/l NH_3–N with complete inhibition at 150 mg/l NH_3–N (Anthonisen et al. 1976; Vadivelu et al. 2007). PCA vectors in Fig. 1 and relationships plotted in Fig. 2 suggest nitrification is most affected by pH and ammonia not just lack of oxygen (low TS%) or by high VS% (out-competition). The upper limit beyond which nitrification should not occur (150 mg/l NH_3–N, Anthonisen et al. 1976) was multiplied by the average moisture content of CT samples (72%) and divided by the average TS % (28%), obtaining an upper limit of 386 mg/kg which was plotted on Fig. 2A. The data set was divided at this threshold, 386 mg/kg < NH_3–N < 386 mg/kg; significantly more nitrate was found below this upper limit (2945 ± 3653 mg/kg) than above it (57 ± 262 mg/kg) ($p = 0.0056$) indicating that this threshold functions as an important determinant in CT end-product maturity ($p = 0.056$). The strong negative correlation between NO_3^-–N and pH (Fig. 2B) is due to an increase in acidity that accompanies nitrification and conversely to an increase in alkalinity that accompanies urea hydrolysis and the production of ammonia (Anthonisen et al. 1976).

The single sample containing considerable amounts of both nitrate (1341 mg/kg) and ammonia despite the ammonia concentration being above the proposed threshold (747 mg/kg) indicates that this nutrient snap-shot approach is insufficient to fully explain the process dynamics; seasonality of toilet use environment may add additional layers of complexity. It is conceivable that both nitrate and ammonia could co-occur in samples where seasonal toilet use introduced waves of fresh urine inducing periodically high ammonia concentrations to layers deep in the pile which had previously sustained nitrification. In order for ammonia to dissipate through leaching or by heterotrophic uptake and for nitrification to occur the low use season, the pile would need to not be inhibited by temperature or saturated conditions. The sample with both ammonia and nitrate came from a trailside toilet in a desert environment with a mean annual air temperature of 9 °C that experienced little to no use during the off-peak period in winter (wet) or mid summer (hot) and high use during spring and fall (warm & dry). The relatively short high-use periods could bring ammonia deep into the pile, but due to high temperatures in the summer and mild temperatures in the winter, this ammonia could leach, volatilize or be assimilated by heterotrophs and allow nitrification to proceed. Of the 11 samples containing less than 386 mg/kg NH_3–N yet little to no NO_3^-–N, a variety of explanations are likely: 4/11 samples were from high elevation/high latitude, low-use campgrounds/ranger cabins receiving little nitrogen and inhibited by temperature much of the year; 1 sample was from a system saturated enough to produce bubbling anaerobic gasses; and the remaining 6 samples from higher use high elevation campgrounds may result from the heterogenous nature of these small systems where leachate may take preferential pathways and not deliver ammonia to all areas of the pile but where temperature may inhibit nitrification.

Temperature was found to be a significant variable affecting nitrate production where samples averaging 10 °C warmer (13 °C as compared to 2.4 °C) produced an order of magnitude more nitrate (3589 ± 3807 mg/kg as compared to 547 ± 1943). This was expected; the relationship between temperature and microbial activity is widely accepted where every 10 °C rise in temperature doubles the rate of biochemical reactions.

Fig. 2A identifies trail toilet placements as having low NH_3–N, and high NO_3^-–N. Significant Wilcoxon sign rank tests confirm trail toilets have less NH_3–N than camp toilets ($p = 0.0023$) and more NO_3^-–N than both camp ($p < 0.0001$) and commercial ($p = 0.0022$) toilets (see Supplemental material). It is conceivable that trail toilets receive lower urination use compared to toilets at end point destinations such as campgrounds because people may be more apt to urinate without the privacy of a toilet facility when hiking and spread out at low population densities and unlikely to be seeing doing so. However, at campsites or in urban settings, higher population densities increase the chance of being seen urinating without the privacy of a toilet and may result in higher use per visitor. Visitors are also likely to spend more time at campgrounds than at trail-side toilets which may be placed by look-out or picnic spots. Trail placement toilets were also the main placement group in the upper right 'stable' quadrant in Fig. 1B, suggesting that of all MLMC sites, trail placements produce the highest quality end-product (higher TS%, lower VS%, lower *E. coli*, and higher NO_3^-–N) (Fig. 1). There was no significant difference in *E. coli* content by toilet placement.

In general, it appears that sanitation and maturity in MLMC end-product are mutually exclusive; low urine additions may enable nitrification from NH_4^+ to NO_3^-–N but the lack of NH_3–N toxicity can result in higher *E. coli* (Fig. 1A) (Allievi et al. 1994; Mendez et al. 2004; Nordin et al. 2009a). When mutually exclusive, the choice between producing mature or pathogen free end-product in MLMCs should be clear. MLMCs require constant maintenance placing operators in close contact with both fresh feces and end-product; best efforts should be made to minimize pathogen content and reduce the risk of transmission (Vinnerås, 2007).

The following six variables can affect pathogen destruction in fecal matter: temperature, TS%, time, out-competition by non-pathogenic microorganisms, pH, and NH_3–N (Redlinger et al. 2001; Vinnerås, 2007; Nordin et al. 2009a,b). Fig. 1A suggests that higher NH_3–N and pH are most responsible for decrease in *E. coli* numbers, both of which produced significant regression lines with *E. coli* ($p = 0.016$ and $p = 0.0038$ respectively) (see Supp. Figures) and can similarly disrupt cellular activity and destroy pathogens (and other beneficial microorganisms involved in material decomposition and stabilization including fungi and invertebrates) (Warren, 1962; Burge et al. 1983). Other studies (Magri et al., in press) show that addition of wood shavings can increase the bacterial survival in source separated faeces compared to plain storage. pH adjustment is a common method for pathogen reduction in biosolids (WHO, 2006) and has a stronger negative correlation with *E. coli* here ($r^2 = 0.17$) but NH_3–N addition can also used for pathogen control (Mendez et al. 2004) and may be more appropriate for remote CT applications despite a poorer fit ($r^2 = 0.11$). Urea does not increase the weight or salinity as much as

lime, urea is more stabile than lime, and is more effective in reducing pathogens including helminth ova in heterogenous fecal matter (Allievi et al. 1994; Moe and Izurieta, 2003; Mendez et al. 2004; Nordin et al. 2009b). The relationship between TS% and *E. coli* was also found significant and acting in the expected direction where drier samples contained less *E. coli*, but the probability and fit were the weakest (0.050 and 0.07) of the three significant variables and TS% in all samples was lower than optimal (Redlinger et al. 2001; Zavala and Funamizu, 2005).

Pathogen reduction of urea amended fecal matter has been demonstrated at a variety of scales on a variety of pathogens; the following studies examined pathogen die-off at ambient temperatures with the following treatment–time–temperatures; a 2% urea solution created a 235 mmol/l NH_3–N solution at 14 °C causing 6$\log_{10}$ reduction in *Enterococcus* spp. after 10 months and in *Salmonella* spp. (presumably similar to *E. coli*) after <2 weeks (Nordin et al. 2009a); a 1–2% urea solution and 1% urea solution with ash created 72–440 mmol/l and 130–230 mmol/l NH_3–N solutions resulting in maximum t_{99} times of 6–60 days for *Ascaris suum* at 34 °C and 24 °C respectively (Nordin et al. 2009b). McKinley et al. (2012) found 99% reduction in *Ascaris* after 19 weeks in excrement amended with fresh urine and ash which produced between NH_3–N concentrations ranging between 500 and 1500 mg/L and pH 10–12 at 20 °C.

Despite the long residence times in CTs studied here (1–8 years) and significant regression of NH_3–N with *E. coli*, natural ammonia averaging 709 ± 687 mg/kg (ds), (178 ± 185 mg/kg (wet), 10.5 ± 10.8 mmol/kg (wet)) and ranging from 0 to 2767 mg/kg (ds) (0–807 mg/kg (wet), 0–47 mmol/kg (wet)) appears unreliable in the elimination of *E. coli*. Considerable numbers of samples, even those found in systems at room temperature for many years, contained >1000 CFU *E. coli*/g which is the guideline value set by the WHO (2006) to ensure a health target of $<10^6$ DALY (a level which they indicate should be attained by 1.5–2yr storage at 2–20 °C). If *E. coli*, are this abundant, and should leachate be primarily responsible for high pathogen counts through re-contamination (rather than survival or regrowth of bacteria) it can be assumed that more resistant pathogens such as viruses and parasitic worms would also be present. Free-living parasitic nematodes of the genus *Diploscapter* and *Rhabditis* were identified in material by Benchmark Laboratories using a dichotomous key within in one of the smaller CTs and in a dump pile containing material >6 yrs old from one of the larger CTs highlighting the reality of this risk.

An obvious design flaw could have contributed to high *E. coli* counts found here. All systems sampled were built to function as continuous reactors receiving constant inputs and requiring periodic end-product extraction. Post-treatment batch processing was not being conducted in any case here (although this is common practice at some sites). As a result, liquids added at the top of the fecal deposit zone have the potential to percolate through the entire pile and re-contaminate older material, effectively reducing treatment time from the 1.5–2 years necessary for sanitation (WHO, 2006) to the time taken for blackwater to percolate through the reactor to end-product, which could be as little as days–weeks.

In order to minimize re-contamination of end-products with pathogens from raw excrement, urine could be diverted reducing leachate and/or end-product could be isolated from the collection zone and stored. More rapid, consistent, and thorough pathogen destruction could be accomplished by adding ash and urea to fecal matter along with the bulking agent to elevate ammonia concentrations well above the range necessary for pathogen destruction at 24 °C due to low ambient temperatures at most CT sites (Vinnerås, 2007). Between 2 and 20 °C, the WHO (2006) recommends 1.5–2 years storage time (without re-contamination risk) for adequate sanitation.

Vinnerås et al. (2009) estimated an NH_3–N concentration of 75 mmol/l as necessary to reduce *A. suum* in human fecal matter by 6$\log_{10}$ units at 34 °C in 28 days; the concentrations necessary to achieve the same result at 24 °C was ~10 times greater (610 mmol/l). Two percent urea by wet weight added to source separated fecal matter at 14 °C resulted in 134–235 mmol/l NH_3–N (Nordin et al. 2009a). At the low temperatures found at the sites evaluated in present study, mainly below 10 °C, the expected reduction is considerably slower. The ammonia/ash amendment, according to the recommendations by Nordin et al. (2009a), would cost ~$0.03 per toilet use based on current prices of fertilizer purchased in 10 kg bags and 100–300 ml ash per use (WHO, 2006) sourced at no cost. It would be possible to monitor this process by testing ammonia concentration, pH, and temperature; from these a desired residence time could be assigned (Nordin et al. 2009a,b) to achieve the health based targets set by the WHO (2006) and then verified at in-field systems.

The resulting sanitized material will likely have high ammonia and nitrogen content intended for application into highly productive land capable of receiving this immature and unstable fertilizer (McKinley et al. 2012). Most sites in this study were not in need of fertilization; on the contrary, most were in protected parks where human activities are managed to reduce impacts on the natural environment.

Source separating vermicomposting toilets (SSVCs) are an alternative remote public toilet system commercially available in Europe. By diverting urine directly to infiltration fields, nutrients in urine are dispersed into active soil layers and the fertilization effect of using urea as a sanitation agent is avoided. Vermicomposting of urine diverted and un-amended fecal matter is thus enabled and dramatically reduces O&M costs compared to current composting (dry) toilet systems (Hill and Baldwin, 2012). Solid end-products (a mixture of vermicompost, trash, and sanitary products) are eventually extracted and disposed off-site after considerable volume and volatile solids reduction. While nutrient recovery from urine and organic matter re-use from feces is conceptually interesting, the practicality and proven functionality of SSVCs should inspire a reanalysis of MLMC capabilities by current operators and redesign by manufacturers, especially when the intended site is in a remote park or protected area.

5. Conclusion

Temperature, moisture content, and material age act together as expected, in the aerobic decomposition of fecal matter. However, agencies should clearly establish whether their primary objective in purchasing a CT is waste management or nutrient recovery as end-product sanitation and maturity appear exclusively attainable and controlled by urine (through toxic effects of ammonia and pH on microorganisms including nitrifying bacteria and *E. coli*).

The amendment process requires a constant supply of feedstock conditioning, monitoring, and management, which would not likely be cost effective should material eventually be removed for further treatment, disposal, or re-use elsewhere.

The process in the toilet chamber is defined by slow degradation that in many cases are hampered by the high ammonia content from the urine. Therefore re-labeling 'composting' toilets as 'dry' toilets may help eliminate some of the confusion around system capability. Source separating vermicomposting toilets, designed specifically for remote site waste management (not nutrient recovery), utilize urine diversion and vermicomposting to reduce O&M costs and risks. While high quality end-products emanating from SSVCs demonstrate a model for nutrient recovery and organic matter re-use, they are disposed off-site, exemplifying the fundamental priorities in remote site human waste management.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2012.12.046.

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