An Evaluation of Waterless Human Waste Management Systems at North American Public Remote Sites

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GEOG 699

September 16, 2013

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by

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A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF

DOCTOR OF PHILOSOPHY

in

THE FACULTY OF GRADUATE STUDIES

(Geography)

THE UNIVERSITY OF BRITISH COLUMBIA

(Vancouver)

January 2013

Abstract

The absence of sewers and roads at backcountry sites makes the management of human excreta offensive, intensive, and expensive. Proper management is essential in order to prevent deleterious pathogen and nutrient discharge. The dearth of resources, vague certification standards, absence of monitoring, and erroneous popular perception have caused misapplication of systems and mismanagement of end products. Elevated environmental impacts, human health risks, and management costs have resulted.

The diversion of urine from urinals and by urine diversion seats significantly reduced the mass of helicopter extracted excrement. However, until a more robust urine diversion system is developed that does not clog, only urine from urinals should be diverted.

Composting toilets failed to produce safe, stable, and mature end-product at all sites surveyed. They should be re-named *sawdust* toilets, following European nomenclature, to avoid further confusion. Performance was dramatically improved with urine diversion, elimination of bulking agent, and optimization for vermicomposting. Despite improving mineralization and reduction of volatile solids, operating costs, exposure risk, and *E. coli, Eisenia fetida* earthworms did not reduce *Ascaris suum* ova concentration or viability. Vermicomposting toilets, unlikely to produce residuals approved for unrestricted discharge, should be designed to minimize waste, costs, hazards, and environmental impacts. This approach is seemingly opposite from *sawdust* toilets, which at considerable cost, strive against unfavorable biochemistry and thermodynamics to produce 'compost' for onsite disposal despite precautionary federal regulations.

Solvita® test paddles, useful in the assessment of end-product, could be used with vermicomposting toilets, to ensure low ammonia is present in feedstock (values 4-5) and to ensure stability (values 7-8) prior to disposal.

Pit toilets, commonly excavated to depths greater than seasonal high ground water, carried the greatest risk of pollution. These were conceptually redesigned to prevent disease transmission and treat nutrients with septic fields. In order to reduce the risks of eutrophication and ammonia toxicity, fields should: be oversized by at least a factor of 10 based on daily urine output; maximize the depth of unsaturated soil with curtain drains where necessary; lie >60m from surface water; and where appropriate use natural wetlands such as moist, acidic, productive and phenotypically plastic graminoid meadows.

Preface

The vast majority of project fundraising, project planning, field data collection, experimental design, experimental data collection, analysis, and writing were conducted independently. The vast majority of biochemical analyses were contracted to Benchmark Labs, a commercial laboratory in Calgary specializing in compost analysis. Dr. Greg Henry (supervisor), Dr. Sue Baldwin, Dr. Anthony Lau, and Dr. Cecilia Lalander provided input and recommendations on sampling procedures, analyses, and experimental design. Dr. Baldwin directed and conducted the majority of the qPCR procedure utilized in Chapter 6. Body chapters of the thesis were written as journal articles and Chapters 2 through 6 were submitted to journals.

Chapter 2 was accepted in the *International Journal of Wilderness Management*:
Hill, G.B. and Henry, G.H.R. (in press) The application and performance of urine diversion to minimize waste management costs associated with remote wilderness toilets. *International Journal of Wilderness Management*.

Chapter 3 was accepted in the journal *Waste Management*:
Hill, G.B., Baldwin, S.A. (2012) Vermicomposting toilets, an alternative to latrine style microbial composting toilets, prove far superior in mass reduction, pathogen destruction, compost quality, and operational cost. Waste Management 32(10), 1811-1820.

Chapter 4 was accepted in the *Journal of Environmental Management:*
Hill, G.B., Baldwin, S.A., Vinnerås, B. (in press) Composting toilets a misnomer: Excessive ammonia from urine inhibits microbial activity yet is insufficient in sanitizing the end-product. Journal of Environmental Management, Ms. Ref. No.: JEMA-D-12-01268R1.

Chapter 5 was submitted for publication under the title "Evaluating Solvita® compost stability and maturity tests in the assessment of mixed latrine style composting toilet end product quality and safety" by Geoffrey B Hill[a1], Susan A. Baldwin, and Bjorn Vinnerås.

Chapter 6 was submitted for publication under the title "Vermicomposting effects on *Ascaris suum* ova" by Geoffrey B Hill, Cecilia Lalander, and Susan A Baldwin.

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Glossary of terms

BFO: Barrel-fly-out toilet system

PT: Pit toilet

UD: Urine-diversion

CT: Composting toilet

UDDT: Urine-diverting dehydrating toilet

UDVCT: Urine-diverting vermicomposting toilet

VT: Vault toilet

WPT: Wet pit toilet (pit acts as septic tank with rain water doing flushing)

DPT: Dry pit toilet (urine diverted solids collect in sealed pit liner)

Acknowledgements

Support of for this research project was provided by: the British Columbia Community Legacy Program; Natural Sciences and Engineering Research Council of Canada; British Columbia Parks; Parks Canada; US National Parks Service; Alberta Parks and Recreation; Back- country Energy Environment Solutions; Mountain Equipment COOP; the American Alpine Club; Metro Vancouver; and the Alma Mater Society UBC. Special thanks to Knut Kitching, Frederic Neau, Cecilia Lalander, Sue Baldwin, John Leffel and all the system operators who accompanied me to the sites. Final thanks to Gail Clevenger for putting up with smelly samples and field equipment.

Dedication

This thesis is dedicated to the belief that our society has (or will have in the future) the will and capacity to prioritize research and implementation of technically appropriate, operationally simple, and biologically robust 'waste' management systems in our most cherished parks and protected areas and in our urban cores. As per the old saying, "it is an ill bird that fouls its own nest" (Alsop 1666), flush water and sewers have made it easy to avoid polluting our own nests. However, where these are absent, even in our most cherished protected areas, considerable nest-fouling occurs. Building environmental consciousness has made nest-fouling less acceptable, especially in popular parks and protected areas, resulting in rising operating costs and a search for better systems. Ultimately, sustainability will be reached when the concept of waste has been eliminated, where our individual (and industrial/commercial) excreta are utilized effectively by surrounding ecosystems. While considerable headway towards this goal has been achieved and documented in this thesis at remote sites, it is not complete and must also be proven applicable to higher density settings to be valuable in the greater sustainability context.

Chapter 1. Introduction

All humans excrete waste. The average person in a developed country excretes 550 kg wet weight of urine (21 kg dw) and 51 kg of wet weight fecal matter (11 kg dw) per year consuming 8.9 kg of toilet paper (WHO 2006). Contained within this annual waste stream is 4550 g of nitrogen and 548 g of phosphorus, of which 4000 g nitrogen (88%) and 365 g phosphorus (67%) are found in urine (Vinnerås 2002). Pathogen concentrations in an infected person's fecal matter ranges from 10^{4-8} of protozoa cysts and helminth ova, 10^{4-9} of bacteria such as *Salmonella spp*., *Campylobacter spp*., and *Escherichia coli* and as high as 10^{11} for enteric viruses such as Hepatitis A, Rotavirus, and Norovirus (WHO 2006). The median infectious dose, also known as the concentration which causes infection in 50% of exposed organisms, of pathogens ranges as low as <1 for *Ascaris* sp. and as high as 23600 for *Salmonella*. Infectious diseases account for 25% of deaths worldwide (Murray & Lopez 1996). Diseases that were once locally contained are now found widely spread across the world (Smith *et al*. 2007) and as biological diversity decreases with latitude and altitude, the chance of pathogen transmission increases (Smith and Guégan 2010).

Toilets, flush-water, sewers, and centralized wastewater treatment plants together with advances in sanitation, hygiene and medical treatment have greatly reduced the transmission of pathogens in urban areas (Smith and Guégan 2010). However, there are many publically accessed remote sites in the developed world, which lack running water and sewers; as visitation and use of these locations increases, the management of human waste can become offensive, expensive, and intensive. The exploration, evaluation, and recommendations for waterless human waste treatment and management at remote sites in North America are the foci of this thesis. Impacts are exacerbated and more visible at heavily used remote sites, which tend to be popular and well known National Park or other designated high value ecological / conservation sites requiring an extra level of consideration for environmental values and ecological function.

Risks of mismanaged human waste include: visual impacts (open defecation); olfactory impacts (smell of fecal matter or ammonia from urine); contamination of drinking water, ground water, and recreation water with pathogens and/or nutrients; and disease transmission to toilet operators or to visitors through direct contact with fecal matter or indirect contact through a vector. Despite obvious transmission routes, these risks may be adequately managed by the treatment of drinking water at low use sites where surface burial of excrement (cat holing) is the primary method and pollution in non-point source (Cilimburg *et al.* 2000). However, at higher use sites with obvious point-source pollution, there is greater risk, as indicated by the Austrian case studied by Rauch and Becker (2000), where pathogen contamination of a downstream water supply was traced back to improperly managed fecal matter at a mountain hut.

The World Health Organization (WHO 2006) reviewed the risks associated with pathogens in fecal matter applied to soil with respect to intentional excreta reuse in developing countries, but in the developed world, there are very few realistic situations where nutrients in fecal matter at public remote sites need to be re-used as fertilizer due to the obvious risks and easy access to inexpensive, pathogen free fertilizer and soil amendments. This is especially so in conservation areas where even the unavoidable fertilization effects of climate change and industrial activity, such as increased nitrogen deposition, are undesired. As such, human waste at remote sites will be addressed from a waste management perspective (taking into account risks, costs, environmental impacts and regulations) rather than a theoretical nutrient re-use perspective. There is value in research on excreta nutrient reuse, but this should be done in a carefully managed way to minimize pollution and health risks or in a setting where nutrients are scarce and excreta reuse is necessary.

Risks are managed either by centralizing collection in toilet structures or by provision of personal collection bags and containers. Personal pack out programs require a unique set of conditions including: a limited number of highly regulated park access points; high cost of mechanized park access; compliant user group; and acceptance of material at landfill sites. A review of pack out programs is made by Robinson (2010) who found the greatest challenge lies in changing the culture of toilet use through education to encourage defecation into bags rather than into a toilet. Centralized toilets systems are generally entirely managed by the public agency / operator in which the toilet is located and are much more common at popular remote sites than pack-out programs. There is considerable interest emerging in developing pack-out programs at numerous mountain parks in the United States (Robinson, personal communications August 2011), however, the switch may be partially motivated by poor toilet system performance or the lack of suitable public toilet technologies. Effort placed on the widely applicable toilet systems may be more fruitful than deep exploration of pack-out programs due to the narrow conditions under which pack-out programs are likely to succeed (Robinson 2010).

A variety of systems are used in North America to manage waste at public sites, all of which mix urine and fecal matter together (in order of increasingly complex technology): pit toilets (PTs) (leach liquids to surrounding soil); vault toilets (VTs) (impermeable to surrounding soil); barrel (BFOs) (removable containment device); solar dehydrating toilets (DTs) (designed to evaporate liquid fraction); and composting toilets (CTs) (require bulking agent and designed to decompose material into compost). Composting toilets are referred to as dry toilets or sawdust toilets in Europe. Urine diverting dehydrating toilets (UDDTs) are common in developing countries but not found in North America presumably due to lower temperatures and less consistent insolation. UDDTs have been reviewed by McKinley *et al.* (2012a) and will not be covered here. Urine diverting vermicomposting toilets (UDVCTs) rely on urine diversion systems to render fecal matter into a attractive food-source for detritivorous earthworms and associated microbes; these toilets are manufactured in France and Wales and sold to the European market.

There is a clear need for a thorough review and empirical comparison between systems as there is limited relevant peer reviewed experimental and field study research on these systems separately and few comparisons between them. A small number of technical reports had been commissioned by public agencies on various waste management topics but most of these were proprietary and not publically available. Moreover, the conclusions drawn by experts, in many cases, are in opposition to commonly held lay perception of system performance and capability. The root causes of the disparity are not traced here, yet this exercise should be conducted to ensure supposedly ‘green’ technologies are thoroughly reviewed and tested by independent organizations and accredited professionals prior to being sold to private or public agencies (especially when involving human excreta). Regardless of their root, these erroneous perceptions have led to mismanagement of end-product from public CTs (Chapter 3, 4, 5) and PTs (Chapter 7) jeopardizing public health (Vassos personal communications January 2012) and environmental quality (Bolton personal communications March 2011).

The PT is one of the most common back-country toilets that range in design from a small hole dug into the ground upon which individuals squat over to defecate and urinate to large 3 m deep, 2 m wide holes excavated with heavy machinery (often into soil parent material) and cribbed with rot resistant wood upon which a removable toilet structure and riser is placed for individuals to sit upon while excreting waste (McCrumb personal communications June 2012). There is very little research on deep PT pathogen and nutrient transfer. Considerable work has been conducted on pathogen destruction and transmission in shallow scat holes (Temple 1982, Cilimburg *et al*. 2000) and septic fields (Moore *et al*. 2010), but these are both constructed in the surface layers of organic and mineral soils which experience greater microbial activity, plant activity, temperature fluctuation, and drainage. Results from these studies conclude that pathogens and nutrients require unsaturated conditions in order to be attenuated by a complex suite of primarily aerobic soil processes (Sherlock *et al*. 2002, Meile *et al*. 2010, Moore *et al*. 2010). Should pathogens be deposited into groundwater, transmission up to 1000 m or more and can be expected for enteric viruses with survival times recorded in years (Goodwin and Loso manuscript, Zyman and Sorber 1988, Cilimburg *et al*. 2000, Moore *et al*. 2010). Findings such as these likely influenced the development of codes: e.g. septic system effluent dispersal through septic fields which require 1-2 m of unsaturated, adequately draining soil (not too slow or fast) in order to prevent transmission to ground water. In some regions, such as Washington State (Washington State Department of Health 2007) and Australia and New Zealand (AS/NZS 2001) the same codes regulate PT construction. However, in some regions, PT construction is not regulated such as in Alberta (Alberta Regulation 2003), the only requirements are that the pit:

(i) is located and maintained so that no nuisance is created,

(ii) is maintained in a clean and sanitary condition and in good working order,

(iii) is protected so that vermin do not have access to the contents,

(iv) and in the case of an outdoor pit privy, the contents are covered with earth or other suitable material when the outdoor pit privy is abandoned or removed.

B.C. Parks, one of the main public operators of PTs in B.C., uses 'best practice' guidelines that place the bottom of pits 2.4 m deep into soil, a depth beyond the influence of active microbial and plant communities (B.C. Parks 2005). These 'best practices' are better labeled 'least expensive practices' as the British Geological Survey (ARGOSS 2001) states that where <5m of unsaturated soil exist below pit toilets, the risk of water contamination is significant. Exploratory field visits and conversations with park operators confirmed the prevalence of ground water in PTs. Even if pathogens and nutrients were traced back from public water supplies to public PTs by conducting expensive and challenging ground water studies there is little likelihood that the use of PTs would be abandoned for the following reasons: 1) ground water fluctuates and in order to determine safe depth levels ground water would need to be evaluated for a period of years; 2) PTs are very cheap to construct and operate because urine, the dominant fraction of waste (Vinnerås 2002), seeps out and the fecal matter and the remaining sludge is left *in situ* never having to be handled; and 3) there are no inexpensive alternatives to the PT. Instead of quantifying the extent and severity of the pollution leaching from PTs at remote backcountry sites, focus was placed on the redesign of the PT with residential septic tank and septic field designs and components in order to eliminate effluent discharge to ground water through urine diversion to surface soil and isolation of fecal mater in an impermeable container. This accomplishment may also make the residual solids (toilet paper and fecal matter) suitable feedstocks for vermicomposting, which could further reduce pathogens and excrement volume. The efficacy and applicability of this urine diversion retrofit is discussed in Chapter 2. The efficacy and application of vermicomposting human waste are discussed in Chapter 3 and 6. The PT redesign is presented in Chapter 7.

Vault toilets are not an option at remote sites lacking truck access because no safe means of pumping septic waste from the vault would would be possible. Accordingly, vaults were not evaluated directly in this thesis. However, the main criteria of the PT re-design could be adapted to a vault toilet with similar improvements (Bianco personal communications September 2012). Vault toilets will only be briefly touched upon in Chapter 7 where costs are compared as the vault toilet is the norm in front country sites and provides a good benchmark for capital cost, operating cost, and occupational exposure.

Excrement at many high alpine sites, including many popular sites in the Canadian Rockies, is collected in plastic barrels and flown by helicopter (BFO) to a staging area where a septic truck removes the sewage for eventual treatment at a waste water treatment plant (WWTP) or lagoon. This system has generally low environmental impacts but has high costs per use due to the expense of helicopter flight and high operator risk and the challenge involved in moving full 200 kg barrels of sewage from their location under the toilet to the helicopter long line or sling connection point. In some locations sewage was dumped onto non-visited mountainsides, even in National Parks, introducing considerable opportunity for pathogen transmission and eutrophication (Catto personal communications 2011). A variety of processes were tested at a remote, high use BFO site including: urine diversion with urinals and urine diversion seats; solar dehydration; 110V desiccation (the site had hydro power during the summer); and 110V incineration with an Elastec American Marine Smart Ash® incinerator and a custom-made burn cage. The reduction in mass and potential for complete incineration onsite are presented in Chapter 2. The benefits of urine diversion were most evident, but repetitive clogging of the urine diversion seat insert suggested that there was a need for a more robust urine diversion system suitable for public toilet systems as can be found in Europe (Chapter 3).

Composting toilets (CTs) are used in North America for the decentralized, waterless, treatment of human waste with the aim of producing “a safe, stable, useable material” (B.C. Parks 2005). Language similar to that in quotations can be found in almost every popular piece of non peer-reviewed writing on the subject of composting toilets. Seldom are any references made and if they are, they are often to Jenken’s (2005) “Humanure Composting”, which barely addresses public commercial composting toilet systems. The prevalence of this language may explain why CTs are commonly perceived as being capable of producing ‘compost’ onsite despite the WHO (2006) recommendation that the difficult process of fecal matter composting be conducted off-site as a centralized secondary treatment. As a result, the disposal / land application of untested end-products into public park environments has been observed and recorded in provincial and National Parks in Canada (Cieslak, Catto, Volp, Trewitt. personal communications 2010-2011).

The objectives of nutrient reclamation and organic matter reuse add complexity to the primary objective of material sanitation, which is, in its self, 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 (such as trash), and pathogens 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, Tonner-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 (B.C. OMRR 2007, WSDOH 2007). This can be labor intensive, offensive, expensive, and dangerous at remote sites. In order to quantify the severity of system failure and regional extent, end-product biochemical quality and hygiene were tested on samples extracted from dozens of CTs in Western North America; the results are summarized in Chapter 3.

It was discovered that 100% of CT sites failed to meet NSF Standard 41 (2011). Moreover, the NSF Standard 41, requiring only fecal coliform testing and total solids determination, was determined to be quite weak and potentially misleading due to its interpretation as approval for onsite discharge of residuals which is actually regulated by various levels of government from county to federal depending on the location. In-depth investigation was made in order to expose the complex biochemistry occurring within CTs. Through Principal Components Analysis, it was discovered that the primary objectives of composting (sanitation, decomposition, mineralization), were mutually exclusive within CTs due to design flaws.

Ammonia and high pH from urea (urine) hydrolysis was the main agent in sanitation, but at high enough concentrations to be effective, the microbial activity necessary for decomposition and mineralization of nutrients was greatly weakened. Conversely, at sites with low urine input, high temperature, aeration, and time could bring about decomposition and mineralization but failed to eliminate pathogens and/or provided conditions that were suitable for growth or sustenance of pathogen populations. It appeared that fundamentally new designs were needed which could processes the urine fraction and fecal fraction separately. These results are presented in Chapter 4.

Many thousands of dollars were spent shipping samples of end-product to a commercial laboratory for analysis in four categories: pathogens (*E. coli*); stability / decomposition (VS%, CO_2 evolution), maturity / mineralization (ammonia, nitrate); and general quality (pH). The uncertainty around what parameters to test, lack of guidelines, and expense of testing end-product likely contributes to the dearth of field data on the topic. We tested a handheld test kit designed to evaluate commercial / agricultural compost stability and maturity, called Solvita®, manufactured by Woods End Labs (Mt. Vernon, ME). It proved a valuable and insightful tool when used on CT end product and these results are included in Chapter 5.

Stimulated by the failure of CTs and a unproductive search for proven alternatives in North America, contact was made with a company producing the conceptually promising UDVCT (vermicomposting has been shown to stabilize, mature, and sanitize animal and human fecal matter at low temperatures in the absence of urine and ammonia (Yadav *et al*. 2010, Sinha *et al*. 2009)) and a field research trip to France and Switzerland was conducted. End-product from UDVCTs was compared to that from CTs and these novel findings are presented in Chapter 3. According to some studies, vermicomposting could destroy even the most resistant pathogens (Eastman *et al*. 2001) thus making the vermicomposting process a potential candidate for adoption into the suite of approved processes capable of producing soil amendment safe for unregulated application to public land from sewage and biosolid feedstocks. If supported and eventually approved, the land application of UDVCT end-product would save considerable transport and disposal costs and handling risks. In order to verify the capability of vermicomposting to destroy helminth ova (one of the most resistant and infective human pathogens found in fecal matter), a controlled bench scale experiment was conducted with fresh, urine diverted, human fecal matter spiked with *Ascaris suum*, a pig helminth commonly used instead of the similar human parasite *Ascaris lumbricoides*. These results clarified a debate which had existed in the literature for a decade and are presented in Chapter 6.

The final synthesis chapter, Chapter 7, steps back and compares system by cost and operational hazards, explores the ecosystem impacts of urine diversion, and presents a decision making tool to aid public agency operators select a toilet systems suitable to their North American remote site.

Chapter 2. Extending the capacity of excrement collection barrels at the Kain Hut, Bugaboo Provincial Park

2.1. Abstract

The diversion of urine away from fecal matter, prior to contact, has the potential to improve a wide variety of public toilet systems managed at remote wilderness sites. In order to evaluate the reduction in mass, cost, and impact associated with human waste management at the Kain Hut (2200masl), Bugaboo Provincial Park, BC, Canada, we designed and tested three alternative waste treatment systems, all of which involved the diversion of urine with urnals and urine diversion seats. By quantifying the mass of excreta deposited per toilet use, we were able to compare the baseline excreta mass collected per use in an unmodified barrel-fly-out toilet system with: that collected in a barrel toilet modified with urine-diversion (urinals and urine diversion seats); urine diversion with solar-dehydration, and urine diversion with 110V-evaporation. The residual material of the 110-V-evaporations system was incinerated with a Elastec Smart Ash® electric incinerator to further reduce the mass. ,Urine diversion significantly reduced human excreta by 60% potentially saving $108 per barrel when removed from Bugaboo Park with a Bell 407 helicopter. Urine diversion with 110V dehydration reduced the excreta mass by a significant additional 34% beyond UD (94% less than the baseline barrel-fly-out) but required constant high amperage power, which is unlikely to be available at most wilderness sites. Solar dehydration and incineration of desiccated material were not effective. Urine diversion seats clogged frequently; a robust public utility urine diversion system is required. More research is needed on the ecological impacts of locally discharged urine. Urine-diverted excrement becomes a viable feedstock for low-temperature vermicomposting treatment, which has recently proven to be one of the more effective waste management systems available for remote / wilderness sites.

2.2. Introduction

Parks Canada is aiming to increase annual wilderness visitation to 22.4 million visits in 2015 from 20.7 million visits in 2008 (Parks Canada 2011). Total waste volume and waste management costs increase directly with increased visitation. In wilderness areas experiencing low usage, human waste may be adequately managed by pack-out, cat-holeing (in areas with adequate soil structure; Cilimburg et al. 2000) or desiccation by smearing (dry/hot); Ells and Monz 2011). Under ideal conditions of low use and suitable environmental conditions these standard methods of disposal should have litte risk of ground or surface water pollution, pathogen transmission, or negative visitor experience (Cilimburg et al. 2000). However, should any of these criteria not exist, the risks associated with human waste outlined by Temple (1982), Cilimburg *et al*. (2000), Moore *et al*. (2010), and Banerjee (2011), should stimulate the implementation of waste management plans.

Human waste management in wilderness and especially alpine wilderness is very challenging. Remote sites frequently lack standard municipal infrastructure including road access, sewerage, electricity, and water supply. Without these basic services the removal or treatment of human waste can become an expensive, intensive, offensive and dangerous task. Additional challenges at alpine sites include: short summers, large diurnal fluctuations, frequent freeze-thaw events, extreme weather, shallow weak soils, limited vegetation, and challenging terrain (Weissenbacher *et al*. 2008). Nonetheless, the proper management of human waste is essential in order to prevent environmental contamination, ensure adequate user sanitation, and meet legal requirements.

There are two approaches to waste management programs in parks and wilderness areas, pack-out or provision of toilets. Pack-out involves the collection of fecal matter in bags, transport throughout the wilderness visit, and disposal at an approved septic waste disposal facility. Toilet provision involves the construction, maintenance, collection, and either on-site treatment and onsite disposal of end-products or transport for offsite treatment.

Effective pack-out programs have a specific set of criteria. These are reviewed by Robinson (2010) and White (2010). In all other wilderness areas, where annual visitation or intensity exceeds the loading rate manageable by open defecation and cat-holing (which can vary greatly site by site), toilets are generally provided. There are a variety of toilet systems used in North American remote wilderness areas including: pit toilets; barrel collection toilets (barrel-fly-out); composting toilets;; and dehydrating toilets. There is a wider selection of waste treatment technologies available in Europe as wilderness travel in Europe is supported and serviced by large networks of popular and high use huts, but many of theses require running water or power (Becker *et al*. 2007).

A variety of systems are utilized to manage the excrement deposited into wilderness toilets including: pit toilets, barrel-fly-out toilets, composting toilets, dehydrating toilets, and occasionally incinerating toilets. Human excrement is composed of urine and feces; the majority of which is urine. Urine, containing the majority of nutrients and much lower pathogen content than feces, could conceivably be treated onsite with minimal impacts by natural soil processes assuming leachate to groundwater was not allowed. Feces, having high organic matter content and pathogen content is much more difficult to treat onsite and in most cases, except where collected in pits, is removed for offsite treatment. The diversion of urine away from feces is commonly practiced in Scadanavian countries primarily in order to capture and reuse uncontaminated nutrients in urine. However there are a number of other beneficial uses of urine diversion, especially when applied to remote site waste management toilet systems.

Pit toilets are one of the least expensive toilet systems to build and operate as they function both as collection and on-site treatment by relying on natural soil to attenuate pathogens and nutrients (Gunn and Odell 1995). Despite research which indicates that >20m unsaturated soil must exist below a pit toilet in order to effectively remove viral pathogens from water 50m horizontally away (Moore *et al*. 2010), common practices either place the pit frequently into ground water (McCrumb personal communications 2012) or require only 1-2 m of vertical separation from seasonal high ground water (Gunn and Odell 1995). Horizontal separation to surface water is reported by Gunn and Odell (1995) to be 10-20 m depending on soil type, but with more recent concerns over enteric virus survival and transport, these distances may be as high as 1000-3000m depending on soil type and depth of unsaturated soil below the pit. Moore *et al*. (2010) provide an in-depth summary and calculation templates for separation distances and risk tolerance. In light of the likely impacts of pit toilets on water quality, they may no longer be a reasonable choice except where proof of vertical and horizontal separation from ground water and surface/well water is suitable for soil type and seasonal flux in water table. It may be possible to eliminate the pollution risk associated with pit toilets if urine is diverted away from the pit and the pit sealed with an impermeable liner. However, more information is needed on the efficiency of urine-diversion and on the reliability of urine diversion systems in wilderness locations with low O&M frequency.

North American mixed latrine style composting toilets propose to employ aerobic bacteria and microorganisms to decompose excrement to the point at which end-products are 'safe' for onsite discharge, making them an attractive alternative for pit toilet sites. However the body of literature on mixed latrine style composting toilets, especially from field studies, indicates that they are unreliable in the production of compost suitable for discharge into public park environment (Matthews 2000, Redlinger *et al*. 2001, Holmqvist and Stenstrom 2002, Guardabassi *et al*. 2003, Jenkins 2005, Jönsson and Vinnerås 2007, Tonner-Klank *et al* 2007, Jensen *et al*. 2009, Hill and Baldwin 2012, Hill *et al*. in press). Moreover, this style of composting toilet is expensive and hazardous to maintain as material must be removed annually to create space for new additions (Hill and Baldwin 2012). With the diversion of urine away from feces, the feces become a viable feedstock for invertebrate driven decomposition (vermicomposting) and the source of odor is eliminated (ammonia from urea) making them far superior in performance and hazard reduction (Hill and Baldwin 2012). However, there are currently no commercially available public-utility urine-diversion systems available in North America. Urine-diversion seats and urinals, commonly used in residential Scandinavia, require testing in a public environment to prove their worth.

In rare circumstances dehydrating toilets and incinerating toilets can be found, but there is limited data on these systems in North American wilderness environments and their ability needs to be evaluated prior to greater market uptake.

Alpine sites, generally not suitable for pit toilets (lack of soil) or composting toilets (too cold), are frequently serviced with barrel-fly-out collection toilets in Canada. Barrels are transported annually by helicopter for off-site treatment. However, the expense and intrusion of helicopters to regularly remove barrels from wilderness destinations is large and can cost thousands of dollars per year at high use sites (Hanson personal communications 2011). By diverting urine, which constitutes 75% of daily excreta mass per capita, away from the collection barrel into a shallow septic field or wetland, considerable expense, intrusion, and risk associated with helicopter removal of excreta could be minimized. The remaining fecal matter, high in moisture, could be further minimized through desiccation.

The performance of urine diversion by urine-diversion seat and urinal would be most effectively evaluated at a barrel-fly-out toilet site because of the simplicity in quantifying excrement collected in easily managed drums. In order to evaluate and enumerate the reduction in excreta associated with each mass reduction treatment we established each treatment at a high use backcountry wilderness site and periodically measured the change in mass collected per average toilet use under each toilet treatment system. Based on the reduction in mass, potential cost savings were estimated using available financial data.

2.3. Methods

The Conrad Kain Hut, Bugaboo Provincial Park, B.C., elevation 2100 m a.s.l., was chosen as a site to test three alternative waste treatment technologies. The hut sits 5 km from and 700 m above the trailhead, 45 km west of Brisco, B.C. Accommodating 40 overnight occupants, it is used principally in the summer by hikers, climbers and guides. It is one of the more popular destinations in the Canadian alpine and is serviced with propane for cooking and lighting. Not common for Canadian huts, it is serviced with a micro-hydro generator for heating and lighting. Water from above the hut is piped directly to plumbing in the hut for cooking and drinking. Greywater is gravity fed to a solids separating tank (when operational) or direct to disposal field in a natural sedge meadow overlying solid granitic parent material 30 m below the hut. There are three outdoor toilets, one close to the hut and two down a short flight of stairs. Prior to our experimental manipulations the toilet close to the hut was used as a urine-only-toilet; a mesh grate just below the toilet surface dissuaded fecal matter additions. Urine from the urine only toilet was diverted into the greywater disposal pipe. The hut and toilets sit upon a small bedrock knoll with unobstructed solar exposure until mid afternoon when Snowpatch Spire interrupts direct incoming solar radiation. This site was chosen for research as it was representative of other moderate-high use alpine sites, was guaranteed to have adequate visitation to accumulate necessary excrement for measurement, and provided attractive sanitation amenities including running water for handwashing and bathing important for researchers and assistants conducting this bio-hazardous research.

Three alternative toilet waste management systems were designed that could be retrofitted into any standard barrel-fly-out toilet (BFO). The simplest system was urine-diversion (UD), which included both a men's urinal and urine diverting seat from EcoVita (Bedford, MA) (Figure 2.1A and B). The second involved the urine-diversion system plus solar dehydration (UD12V). This system transfers incoming solar radiation to sensible heat inside a thin flat transparent panel; this hot air is then driven through ducting by a fan and photo-voltaic panel to the surface of the excrement pile. The 0.5 m^2 solar hot air panel, 100 cubic-feet-minute (CFM) fan, and 5 watt photo-voltaic panel were purchased from Clear Dome Solar (San Diego, CA). We designed our own solar dehydrating toilet system based on Arnold's (2010) design (Figure 2.1C). The third system combined urine-diversion with a 110V 800W heater and a 110V 110 CFM blower and exhaust fan inside an insulated chamber (UD110V) (Figure 2.1D). The toilet closest to the Kain Hut was chosen for UD110V due to its proximity to 110V outlets. The basement chamber at this toilet was insulated with 4 cm thick Styrofoam boards. Data for the UD110V treatment were collected during two sample periods, August 15-17 and September 3-5, during which time access to the other toilets was restricted so as to account for all toilet uses. BFO, UD, and UD12V treatment systems were established at the lower two toilets for 3-6 day periods according the following schedule: BFO - BFO August 14-18, BFO - UD12V August 18-20, UD - UD12V September 4-10, BFO - UD September 14-19. During these sample periods access to the UD110V upper toilet was restricted as much as possible without creating line-ups so as to maximize the use in the lower toilets and ensure no preference or bias was occurring in toilet selection (i.e.: upper toilet for urination, lower toilets for defecation). In addition, hut visitors were instructed to use all available toilets equally during their stay so as to ensure an even and unbiased distribution of toilet use (i.e.: potential preference for left vs. right).

In order to determine mass reduction performance with respect to the standard BFO, the number of door counts at 6-24 hour intervals were recorded over the course of the 3-6 day sample periods. The interval and period length depended on the intensity of hut visitation; the sampling intensity was increased with elevated visitation. We targeted 10-30 toilet uses per interval in order to maximize the number of intervals while minimizing differentiable mass change at the collection barrel below each toilet. Change in barrel mass was determined by weighing the collection barrel before and after each sampling interval with a veterinary pet scale. Door counters were EPC-MAG1 model made by Inter-Dimensional Technologies, Inc. (Hop Bottom, PA). A ten second delay function was employed in order to eliminate erroneous readings caused by wind or door closing errors. We subtracted the unit's final count from its initial and divided the difference by two in order to obtain the total toilet uses. Dividing the change in barrel mass by toilet use eliminated the effect of variable sampling interval length and established a robust quantifiable baseline in the assessment of remote site waste treatment performance. A simple mass balance equation was used to quantify performance. Temperature and humidity sensors connected to data loggers (HOBO U12, Onset Computer Corp.) were used to collect ambient and treatment system air temperature and relative humidity data. Windspeed at the outlet of the ducting above the barrel was measured with a Kestrel® 4500 (Nielsen Kellerman).

All three alternative treatment systems were tested twice. BFO was tested three times. Combined there were nine treatment runs conducted August and September 2010. Each run was divided into three to six sample periods. Measurements with less than five toilet uses per sampling period were not used in order to reduce variability.

Two complications of diverting urine away from the collection barrel were expected: disposal of remaining solids; and social acceptance. The collections barrel, post urine diversion, were expected to contain high total solids and require additional water at the time of collection by vacuum septic truck. By coordinating with the septic truck at the end of this experiment, it was possible to document the challenges associated with excrement extraction from barrel containers.

JMP 9 (SAS Institute) was used to analyze the data. For all tests, the alpha value was set at 0.05. One outlier was removed from the BFO treatment data set after it was discovered that a dysfunctional door latch caused an overestimation of toilet use. No other alterations or transformations were made or required for the data analysis.

2.4. Results

The installation of the urine diversion seat and urinal required one hour (Figure 2.1A, B). The solar hot air system was tested prior to installation on August 16th on an exposed meadow adjacent to the Kain Hut. The sky was cloudless and winds were calm over the course of the day. The solar hot air panel consistently raised the air temperature and reduced the relative humidity for eight and a half hours by an average of +10ºC and -14%, respectively, with a maximum heating of +15ºC and reduction in relative humidity of -19%. Wind speeds at the outlet of the vent varied from 0-3 m/s. The solar hot air system required eight hours to plan and install at the lower toilet site (Figure 2.1C). Over the course of two sample periods, spanning four days, the treatment consistently raised the air temperature and reduced the relative humidity for 6.8 hours per day. The hot air panel produced a maximum of 3 m/s air flow, heating of +7ºC, and drying of -18%.

UD110V system assembly and testing required 15 days. During a representative 20 hour sample interval the system increased the average basement temperature and reduced the relative humidity by an average of +24.7ºC and -44%, respectively, up to a maximum of +30.5 ºC and -63%. The system averaged an actual temperature of 31 ºC and 17% relative humidity.

Change in barrel mass per toilet use data were compared between sampling periods within treatment type with robust, rank sum, non-parametric Wilcoxon Kruskal-Wallis tests; none of the treatment runs were significantly different. Therefore, in order to increase sample sizes, we grouped treatment runs into treatment types (Figure 2.2). The relationship between mean change in excreta mass per toilet use by treatment type was significant ($p<0.0001$) with largest mass associated with BFO toilets (median = 0.27 kg/use, range = 0.11-0.37 kg/use), followed by UD (median = 0.11 kg/use, range = 0.05-0.15 kg/use), UD12V (median = 0.086 kg/use, range = 0.03-0.15 kg/use), and UD110V (median = 0.009 kg/use, range = 0-0.04 kg/use). There was no significant difference between UD and UD12V, but both were significantly different from BFO and UD110V.

2.5. Discussion

The median values of: urine mass/toilet use (feces mass/toilet use subtracted from excreta mass/event); feces mass/toilet use (UD mass/use); and excreta mass/toilet use (BFO mass/use); were found comparable to values from other locations (Table 1). Average urine output reported here (0.16 kg/use) was slightly lower than other studies (Table 1) but this could be explained by the remote location. All site visitors are required to ascend >1000 m in elevation to access the facility and the main activities include hiking and mountaineering, both of which are likely to induce dehydration. Wet fecal mass found here (0.11 kg/use) is considerably larger than the average wet fecal mass from Asian or European or North American reports which average between 0.02-0.10 kg/toilet use based on 5 excreta events per day (Table 1). The fecal mass we reported may also be slightly elevated due to the assumption that all matter collected in the UD treatment was fecal matter; it is likely that a small fraction of urine bypassed the urine diverting seat and urinal. If the efficiency of urine diversion was 90%, the fecal mass/toilet use would drop (0.10 kg/use) and the urine mass/use would increase (0.176 kg/use). It is also possible that backcountry toilets have a higher ratio of urination:defecation use compared to the frequency these occur naturally (5:1) as people are likely to urinate in the wilderness while they are hiking to the destination or may choose to walk away from the toilet to urinate instead of urinating in the toilet. The latter may be increase during high use times when the toilet has strong odors or a line-up.

The results indicate that with the addition of a urine diversion seat and urinal up to 0.16 kg of excreta per use can be eliminated from the barrel-fly-out system. This equates to a 60% reduction in barrel-fly-out mass. Equipped with urine diversion equipment, each barrel will hold 2.5 times as many excrement deposits as compared with standard all-in-one barrel collection systems, greatly reducing the total numbers of barrels filled at each site.

A urine diversion seat and plastic urinal costs less than $200CDN and is simple and quick to install. The urinal was easier to maintain than the urine diversion seat, which regularly became clogged with toilet paper and on one occasion was defecated into by a child, which required immediate cleaning to prevent unsanitary conditions. A low maintenance urine diversion system is not available in North America (Shiskowski 2009). There are two proven commercial urine diversion products in Europe: an inclined foot operated treadmill (Ecosphere Technologies (France)) and the adhesion and gutter systems (NatSol Ltd. (Wales)). Neither system could be found for sale in North America. Urine could be diverted into pre-existing greywater systems for dilution and to reduce the chance of struvite precipitation and potential constriction of flow. Sites without a preexisting greywater treatment system would need to design and construct a leach field according to local septic field codes to ensure sufficient soil surface area to attenuate nutrients and low levels of pathogens given estimate flows of urine (Steinfeld 2007). Septic field design considerations for waterless toilets are discussed in Chapter 7. Fertilization impacts and septic field monitoring are discussed in Chapter 7.

With non-significant differences between UD and UD12V, we are unable to conclude whether solar dehydration is a viable waste reduction treatment. Given labor and capital costs to set up and take down the dehydrating equipment and variable weather conditions that would reduce the efficacy of the dehydrating system we do not think dehydration through this type of retrofit is likely to be a reliable solution for these alpine areas. More effective commercial dehydration toilet systems maximize the surface area of fecal matter and the time it is exposed to a desiccating environment; the best example of this is the cloth bagging carousel systems developed by Ecosphere Technologies where a urine diverting treadmill moves urine-diverted fecal matter onto the surface of a rotating carousel (where cloth bags hang) ensuring subsequent fecal deposits do not cover up the most recent additions and even those buried can desiccate through the cloth fabric.

The UD110V treatment had the greatest reduction in mean excreta mass per toilet use but is the most inappropriate system for most wilderness toilets due to its reliance on micro hydro power and constant maintenance. This toilet also had the greatest degree of sampling error, being closest to the hut and likely used most frequently for quick urination trips at night and the lowest number of sample intervals. These factors suggest low confidence in the data from this treatment and questionable practicality of this waste management system. Instead, further research should be conducted on commercially available dehydrating toilet systems, which reportedly can dry material with only solar energy, to the point at which it can be burned onsite (Neau personal communications Jan. 2012).

Fecal matter must have <15% moisture content before it is easily burnable (Pretzsch personal communications August 2010); applying this to the average wet fecal deposit measure here of 110 g/toilet use, the estimated desiccated end-product would need to be <16.5 g/toilet use, which is slightly higher than the result obtained in the UD110V of 8.6 g/toilet use. However, we attempted to burn the end-product of the UD110V treatment onsite with a portable SmartAsh® cyclonic incinerator by Elastec without success, casting doubt on the ability to burn desiccated fecal matter. More research is needed to verify the claims that this material can be incinerated onsite. If validated, this treatment would result in the lowest mass/toilet use, management exposure, and off-site transport cost.

Diverting urine away from the collections barrel resulted in a much thicker residual material, which did not slosh when dragged out from under the toilet. This is an important aspect of waste management, as visitors are required to exchange full barrels for empty ones at many wilderness sites managed by the Alpine Club of Canada where hut custodians are not present. Full barrels of conventional excrement are predominantly urine and are more difficult and hazardous to handle but easier for the septic truck to evacuate. We documented the evacuation of the urine-diverted barrel, which required 4 times as long (20 minutes as opposed to a standard 5 minutes) and the addition of an equal volume of water. The success of this step was critical in proving the benefits of urine diversion in this context. Septic truck costs ($225/hr) are much lower than helicopter costs ($2000/hr) and many septic trucks carry water tanks.

Discharged urine will have a fertilization effect on the surrounding terrestrial system, but the impacts will be localized, generally favor grasses, sedges, and deciduous shrubs, and should not necessarily enhance invasive species (Bowman *et al*. 1995, Wang *et al*. 2010, Ells and Monz 2011). Competition for nutrients found in urine, by both microbes and plants, is strong and risk of leaching nitrogen into water bodies is low if sufficient unsaturated soil is available, even in alpine and Arctic soils (Brooks *et al*. 1996, Jones and Murphy 2007). These findings come from experimental studies simulating the effects of climate change, snow cover change, or land use change and while experimental fertilizer loading rates per area may be similar to those at a moderate use site with urine diversion into a leach field, there would be considerable value in documenting the changes resulting directly from the establishment of such a system to be sure there are no unexpected impacts. To the best of our knowledge, no such studies have yet been conducted.

Many wilderness destinations in the Canadian Rocky Mountains are used for winter travel where urine diversion would result in discharge to subsurface soil under a snowpack. Fortunately, nutrient uptake even occurs in winter in both Alpine and Arctic tundra under snowpacks (Bilbrough *et. al*. 2000, Schimel *et al*. 2004). There is some concern with frozen urine causing blockages in pipes or at the discharge point, but pilot projects have demonstrated this concern is limited when plumbing runs are short, piping is of appropriate diameter for flow but small enough to conserve heat in the urine (Neau personal communications Jan. 2012).

2.6. Conclusion

By periodically weighing the excrement collected in a 200 L drum under a toilet at a remote site in the Canadian Purcells we were able to evaluate the reduction in mass per toilet use associated with three cost saving retrofits: urine diversion, 12 volt dehydration, and 110V desiccation. Urine diversion through urinals and urine diverting seats was the most successful and reliable and can reduce barrel-fly-out mass by 60%, potentially saving up to $100 per barrel. At sites with very high transport costs, there may be value in further reducing mass by desiccating the remaining fecal matter, but our systems were unlikely to be adequate and we recommend more investigation into commercial urine diverting dehydrating systems. The residual thick sludge was successfully extracted by pump truck but required an equal volume of added water which contributed to the process taking four times longer than barrels without urine diversion. Due to the maintenance requirements associated with the urine diversion seat and the added expense and requirement for water, it is recommended that only urinals be used to divert urine until a robust and efficient toilet riser urine diversion system is made available on the North American market.

2.7. Tables

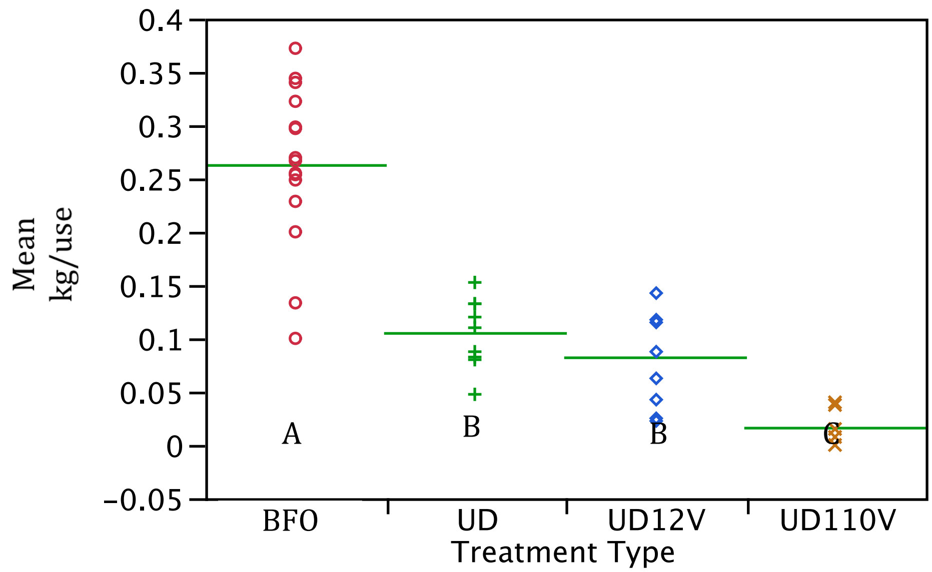
Table 2.1. Range/median urine, feces, and excreta wet mass per toilet use. Modified from Schouw *et al*. 2002, by dividing reported generation rate of urine, feces, and excreta per person per day by the average number of excreta events per person per day (5). * assumes a urine diversion efficiency of 100%.

Location	Urine range (g/toilet use)	Feces range (g/toilet use)	Excreta range (g/toilet use)	Reference
Vietnam	164-240	26-28	198-267	Polprasert *et al*. 1981 in Schouw *et al*. 2002
Developing nations	240	26-100	290-310	Feachem *et al*. 1983 in Schouw *et al*. 2002
Europe / N.A.	240	20-40	270	Feachem *et al*. 1983 in Schouw *et al*. 2002
Thailand	120-240	24-80	188-306	Schouw *et al*. 2002
Canada backcountry	160*	110*	270	This study

2.8. Figures

Fig. 2.1 Alternative toilet waste management treatments at Kain Hut, Bugaboo Park, B.C. Canada. A) Urine diversion toilet seat. B) Urinal with 1” braided drain pipe to collection barrel. C) Lower toilets with UD12V solar hot air panel (i) above a 5W PV panel (ii). D) Upper Kain Hut toilet insulated basement, 110V heater, and 110CFM exhaust fan positioned around a 200L plastic barrel with double garbage bags to collect solids.

Figure 2.2. Change in excreta mass per toilet use by treatment type (kg/use) (wet solids). BFO = the conventional system where all excreta are collected in a single barrel. UD = fraction of solid excrement collected (and weighed) with urine-diversion seats. UD12V = urine-diversion (UD) system plus addition of a solar hot air drying system. UD110V = urine diversion plus addition of a 110V heater and dehumidifier. Data are measurements from each 24 h interval over the summer. Significantly different treatments denoted with different letters (A, B, C) as determined by pair-wise comparisons using the Wilcoxon method (a=0.05).



Chapter 3. Vermicomposting toilets, an alternative to latrine style microbial composting toilets, prove far superior in mass reduction, pathogen destruction, compost quality, and operational cost

3.1. Abstract

Composting toilets aim to recycle excrement into safe, stable humus. Preceding this, low costs, low risks, and mass reduction should be ensured. Urine diverting vermicomposting toilets (UDVCTs) outperformed mixed latrine microbial composting toilets (CTs) in all categories. CTs: incurred ten times greater operational costs; created ten times more operator exposure; employed no proven pathogen reduction mechanism; produced solid end-products averaging 71,000±230,000CFU/g *E.coli* and 24±5% total solids which consistently failed NSF/ANSI Standard 41. CTs also: failed to reduce volatile solids in 'finished' compost compared to raw fecal matter; increased total contaminated dry mass by 274%; and produced alkaline end-product (8.0±0.7) high in toxic free ammonia (Solvita® 2.6±1.5). UDVCTs, conversely, had: low maintenance costs and risks; adequate worm density for pathogen destruction (0.03±0.04 g-worm/g-material); and produced material low in *E.coli (*200±244 CFU/g), neutral (pH 7.4±0.3), stable (60±10% volatile solids), and mature (4±0 Solvita® NH_3) end-product.

3.2. Introduction

The transport of human waste from remote or decentralized sites by helicopter is hazardous, expensive, and intensive. Systems with low maintenance costs are sought after and usually include a waste treatment process and onsite disposal of liquids and/or solids. The costs and benefits of urine diversion for onsite treatment by shallow septic field were explored in Chapter 2. There have been attempts to compost both liquids and solids together onsite in remote National Parks as far back as 1980 (Yosemite National Park 1994).

Composting toilets are marketed as waterless human waste treatment systems suitable for public service at remote sites. The overall goal of most compost toilet manufacturers is to facilitate the decomposition of human waste without reliance upon the surrounding soils to the point that end-products can be safely disposed onsite without further treatment. Marketing information from compost toilet manufacturers indicates that liquid leachate and solid end-products are suitable for on-site disposal in public parks, providing an attractive alternative to the expense incurred transporting biosolids offsite for disposal or further treatment.

There are risks associated with the operation of a compost toilet which continually discharges leachate and requires the periodic discharge of solid end-product including: direct pathogen transmission during maintenance; indirect pathogen transmission by vectors to visitors or environment; phytotoxicity; immobilization of nutrients in soil, reduced oxygen supply to soil, and eutrophication of aquatic environments (Fuller and Warrick 1985, Cilimburg *et al*. 2000, Wichuk and McCartney 2010, Moore *et al*. 2010).

Operational hazards associated with composting toilets are managed by workplace safety codes and commonly include: training in handling hazardous materials; confined space training; minimizing direct contact with fecal matter; wearing protective equipment; sanitation; and vaccination against blood borne diseases (Land 1995).

Onsite discharge of liquid effluent from composting toilets is managed similarly to septic tank effluent in order to prevent contamination of surface water or ground water by nutrients or pathogens (WSDOH 2007). Septic field guidelines and regulations are commonly enforced at the local authority level requiring construction of an engineer approved septic field (WSDOH 2007). Assuming proper construction, soil conditions, and separation distances to water, the impacts of leachate are low (Moore *et al*. 2010).

Onsite discharge of solid end-product from composting toilets is regulated as per larger biosolids composting facilities under federal (EPA 2003), state (e.g. WSDOH 2007), and local codes (e.g. Snohomish Health District 2004) in the USA and under provincial codes in Canada (e.g. B.C. 2007 and Alberta 1996) with federal guidelines (CCME 2005). Composting biosolids for reuse in a public park setting requires time-temperature monitoring and regular testing to ensure conformity and ensure pathogens are destroyed, nutrients matured, and material stabilized (Haug 1993, Alberta 1996, B.C. 2007). Most regulations stipulate compost must attain temperatures above 55°C for >3 days (CCME 2005). After thermophilic treatment fecal coliforms in finished material must be reduced to <1000 MPN/g (dry solids (ds)), indicative of a 4 log reduction (99.99%) in the wide range of more virulent pathogens found in human biosolids (de Bertoldi *et al* 1983, Haug 1993, Guardabassi *et al*. 2003). Ideally, a complete pathogen assay would be conducted, especially when employing a non-standard process, where a passing grade would find no infective viruses (<0.25 PFU/g ds), no viable *Ascaris* spp. (<0.5 viable ova/g ds), median of all samples <1 MPN/g ds for *Salmonella*, and median of all samples <10 MPN fecal coliform/g ds, not more than 20% >1000MPN/g ds, and none >10,000MPN/g (Haug 1993). Testing of this intensity would determine the actual safety of compost toilet end-product, but is not realistic at the scale of these small public composting toilets due to considerable costs necessary for statistical certainty and to cover the heterogeneity in the pile.

NSF/ANSI Standard 41 for Non-liquid Saturated Treatment Systems is one of the only certification standards in North America that directly addresses the biological treatment of waste in public waterless composting toilets (NSF/ANSI 2011). The standard's purpose is to establish minimum materials, design, and construction, and performance testing of waterless storage / treatment devices and is intended to protect public health and the environment (NSF/ANSI 2011). Testing is conducted once on a new system installed at NSF/ANSI headquarters and once on three mature in-field systems. It addresses maintenance hazards by mandating that the design must not require the operator to ever enter into the chamber for maintenance. Addressing the risk of pollution associated with ongoing liquid leachate, five samples of leachate from each system must no have foul odor and <200 MPN/100 ml fecal coliform. Solid end-products from five samples are also tested in order to determine whether units are capable of producing 'safe' material. These limits are: TS%>35; fecal coliforms <200 MPN/g; and no objectionable odor immediately following removal. No references are provided explaining how the criteria were established or what relation exists between the standards and pathogens of greater concern. Despite testing and certifying end-product, the standard states that it provides no guidance on end-product management and fails to reference national, state, provincial, or local codes governing biosolid reuse (NSF/ANSI 2011).

In addition to meeting the above regulations and NSF/ANSI Standard 41 criteria, compost by definition should have high quality soil amendment properties, which includes being stable, mature, of balanced pH, and free of foreign objects and heavy metals (CCME 2005, Wichuk and McCartney 2010).

There are two styles of public composting toilet facilities: mixed latrine style composting toilets (CT) where feedstock are mixed together to create the proper energy source for microbially driven aerobic decomposition; and source separating vermicomposting toilets (UDVCT) where urine and fecal matter are separated and earthworms are used to degrade fecal matter (Figure 3.1). There may be some cross-over between these systems, where earthworms inhabit CTs, but in the majority of cases, worms added to CTs die rapidly (ON-SITE NewZ 2000).

There are a number of brands of CT available in North America and all are operated in a similar fashion; urine, fecal matter, bulking agent, and toilet paper are added at the toilet hole, liquids leach through and out the bottom or side, and solids are periodically removed to allow room for raw inputs (Figure 3.1). White wood shavings, most commonly Pine spp., are used as bulking agent. All require: periodic surface mixing and raking to combine fecal matter with bulking agent; trash removal; and moisture manipulation. All systems are initiated ½ - ¾ full of bulking agent. Some recommend auxiliary heaters.

CTs are designed and operated to enable aerobic decomposition by including: porosity for oxygenation through structurally resistant bulking agent; homogeneity through mixing or raking bulking agent into fecal matter; elevation of C/N ratio from 6 (night soil) to 15-30 (optimal) by addition of carbonaceous bulking agent; and drainage to prevent saturated anaerobic conditions. In addition to decomposition of material, CTs are also expected to sanitize the pile of harmful pathogens as a result of long storage times and competition by benign micro-organisms.

The limited body of literature on CTs, especially field versus laboratory studies, generally does not prove them reliable for decomposition or sanitation of fecal matter. Adequate temperatures are seldom if ever attained eliminating this reliable mechanism of pathogen destruction (Redlinger *et al*. 2001, Holmqvist and Stenstrom 2002, Jenkins 2005, Tonner-Klank *et al* 2007, Jensen *et al*. 2009). Jenkins (2005) writes extensively about composting humanure in batches at private residences, yet only notes in passing that most commercial systems for public use fail to attain required temperatures to kill pathogens and are continuous flow designs. Should sufficient temperatures be attained at any point during the year, raw leachate contamination into finished material, a result of the design, could negate any sanitation achieved (Figure 3.1). Storage alone is unlikely to be a reliable pathogen destruction mechanism; short-lived pathogens such as fecal coliforms may diminish, but more resistant pathogens such as viruses (Rota virus, Hepatitis A virus) and hookworms can survive multiple years (Gray *et al*. 1995, Ramos *et al*. 2000, Guardabassi *et al*. 2003, Tonner-Klank *et al*. 2007). Redlinger *et al*. (2001) found desiccation to be a more effective than composting in pathogen destruction where less than 36% of toilets studied reduced fecal coliform to <1000 MPN/g over a six month period. However, despite efforts to provide drainage, saturation and foul smells consistently plague both public and private CTs (Matthews 2000, Jönsson and Vinnerås 2007). Despite efforts to increase C/N ratios, the actual carbon available at low temperatures to microbes may be much less than the value reported by loss of ignition (LOI) methods (Kayhanian and Tchabanoglous 1992), which may lead to high free ammonia concentrations and negative impacts on nitrification (Chapter 4). CTs were not expected to produce high quality compost, nor adequate or consistent sanitation. Nevertheless, field data were needed to determine the extend of the problem.

For the reasons above, alternatives to CTs are needed, such as urine diverting vermi-composting (UDVCT), which are commercially available in Europe. Urine and fecal matter, added at the toilet hole are separated from each other either by means of surface tension off the inside of the toilet riser to a gutter or by drainage away from fecal matter off an inclined foot operated treadmill. The accumulated solids are periodically shoveled to a finishing pile where earthworms, cultured in matured compost, convert the fecal matter and toilet paper into vermicompost. Moisture is conserved with impermeable pile coverings and/or delivery of rain-water captured on the roof. Process maintenance is minimal as worms are relied on to mix, aerate, homogenize, and sanitize the excrement.

Invertebrate driven decomposition as an engineered waste treatment process is gaining popularity in developing countries due to its low-tech nature and high quality end-product (Davison *et al*. 2006, Kumar and Shweta 2011). Environmental conditions appropriate for vermicomposting, such as ambient temperatures (5 - 29ºC) and high moisture content (60-75%) are easily maintained at remote or decentralized sites (Sinha *et al*. 2009). Worms can survive near zero temperatures, their cocoons can survive freezing temperatures, and viable populations have been maintained as high as 2400 m (Neau personal communications Jan. 2012). Source separated human waste (Yadav *et al*. 2010), human biosolids (Eastman *et al*. 2001), and sewage sludge (Dominguez *et al*. 2000), all viable feedstocks, have been transformed into vermicompost meeting standards for compost stability and maturity. The majority of research supports the ability of vermicomposting to reliably and greatly reduce pathogens from contaminated feedstock. Yadav *et al*. (2010) and Brambhatt (2006) documented total coliform elimination (>8 log reduction) in large 60 kg batch tests. Kumar and Shweta (2011) document complete removal of *Salmonella*, *Shigella*, *Escherichia* and *Flexibacter* spp. in an analysis pre, mid, and post worm gut. Eastman *et al*. (2001) documented more rapid and more complete pathogen destruction (bacteria, virus, and helminth) than standard microbial composting of the same feedstocks using high densities of earthworms. Intestine action, secretion of coelomic fluids, selective grazing, and alteration of microbial community composition, were all cited as important in pathogen destruction (Sinha *et al*. 2009, Kumar and Shweta 2011).

In a review of pathogen destruction through vermicomposting, Edwards and Subler (2011) note one study by Haimi and Huhta (1987) in which fecal *Streptococci* spp. increased after vermicomposting, which was one of the only papers found doubting the viability of vermicomposting in complete pathogen destruction given suitable conditions and time. It was hypothesized that UDVCT toilets will be more reliable and complete in the destruction of pathogens and production of high quality end-product than CTs.

Despite the dearth of data on the actual performance of composting toilets in the field, they continue to be used and actively marketed with promises of achieving waste reduction and sanitation goals. We visited and sampled end-product material from conventional composting toilets and vermicomposting toilets in a wide variety of environmental conditions and conducted stability, maturity, and sanitation analyses in order to compare the efficacy and reliability of these waste treatment systems in producing compost and in comparison to NSF Standard 41. To our knowledge, this is the first such study conducted in North America.

3.3. Materials and methods

3.3.1. Sites

Agencies operating public composting toilets in Western Canada, Pacific Northwest USA and Western Europe were contacted requesting permission to extract samples of end-product for analysis. All those granting permission, and where travel expenses were minimal or covered by the agency or a supporting grant were visited. Seventeen sites were visited. Eleven sites were designated remote (away from roads and motorized access); five of these remote sites were in campgrounds, and six were at backcountry destinations without overnight facilities. Two sites were found within buildings; one heated and one not. The remaining four were at trailhead parking lots.

Sites were between 50 m and 2200 m elevation. The sites received 360-35,000 uses/toilet/year with a concentration of usage in summer months and minimal usage in the winter months except at the CT within the public building where usage was more consistent throughout the year, and two UDVCTs beside ski lifts at ski resorts. A summary of site characteristics can be found in Table 3.1.

3.3.2. Chambers

Four brands of CT, two suitable for high use (A & C) and two suitable only for low use (B & D) were found in Western North America. One brand of UDVCT was found at the five sites in Europe. All were commercial units, sized to handle estimated use based on manufacturer instruction manuals, site use records, and installed professionally. All brands were found in a range of environmental conditions at various sites (Table 3.2). Solar heaters were found at a few of the sites visited and consisted of sheets of clear polymer affixed an exterior wall, which slightly warmed incoming air when the sun was shining on them; these were deemed to have an insignificant effect on the process occurring within the chamber. Due to similarities in process and design, all CTs were grouped together for comparisons against UDVCT (Table 3.3).

Pile temperature was measured in the deposition zone and at the point of sampling with a 10 cm long probe digital thermometer (three replicate measurements per sample). Annual toilet usage was estimated from the total number of annual site visitors plus day visitors estimated to use the toilet facilities (reported in use/year/per toilet chamber) provided by site operator. Each overnight visitor was assumed to use the toilet three times, day visitors 0.75 times, and public building occupants twice daily.

3.3.3. Samples

Grab samples were extracted from the compost chamber with a gloved hand from the bottom/oldest sections of the material pile in similar fashion to methods outlined in NSF/ANSI Standard 41 (2011). From the grab-sample, a subsample was scooped with a sterile glass sample jar. At each site reported, two to five samples were extracted from representative sections of the bottom of the pile. 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 the laboratory received the samples within 24-48 hours of sampling and a minority within 72 hours. Grab samples of raw excrement were extracted from the deposition zone (top) of some toilets in order to establish a baseline of feedstock characteristics. Two chambers were sampled on three separate occasions and a third chamber was sampled on two separate occasions to test the repeatability of the sampling method. Where CTs are grouped together and compared to UDVCTs, repeat measurements from the same sites were not included. Instead, the sample-visit with the greatest sample size was chosen to represent the site.

3.3.4. Biochemical analyses

E.coli is reported as the number of colony forming units (CFU), as membrane filtration techniques were more economical given our large sample size. Even though there are other species within the fecal coliform group, *E. coli* are commonly used to indicate fecal pathogens (Foppen and Schijuven 2006). A CFU/g count of *E.coli* will contain an equal or greater number of CFU/g fecal coliforms. Further, CFU values are generally more conservative and less variable than MPN (Gronewold and Wolpert 2008). Benchmark Labs (an ISO 17025 accredited Laboratory in Calgary, Alberta) ran simultaneous tests comparing their CFU/g assay to MPN/g values and a positive significant relationship was found (r^2=0.77, $p<0.0001$, MPN=4760+10*CFU) but in 17 of 20 samples MPN results were higher than CFU result (data not shown).). Dilutions and thorough agitation ensured a homogenous slurry from which the subsample was extracted for plating. Any samples with visible nematodes were sent for analysis. All values are reported as per dry solids (ds) unless specified otherwise.

3.3.4.1. CT samples

Benchmark Labs analyzed solid end-product samples for the following parameters (units in brackets) (followed by test procedure): TS (%)(APHA Method 3540B); VS (%)(APHA Method 2540); pH (cold water shake, 1:2, sample:water, followed by measurement with VWR symphony pH probe at 25ºC); *E coli* (CFU/g ds) (cold water shake extraction followed by USEPA Approved Method 1604, with only *E. coli* reported by membrane filtration using a simultaneous detection technique (MI Medium)); Nitrate (mg/kg ds)(by ion chromatography using APHA Method 4110A),

Product maturity as defined by NH_3 production was measured with the Solvita® ranking method (1-5) using a colorimetry scale read by eye off the chart provided after sample preparation according to the instruction manual: no moisture adjustment, 1-2 hrs equilibration without lid followed by 4 hrs at 20-25^{0}C with lid on. According to the Solvita® manual, at pH 8.0, values of 1 and 5 correspond to >20,000 mg/kg-ds and <1000 mg/kg-ds ammoniacal-N, respectively.

3.3.4.2. UDVCT samples

Laboratoire Departmental D'Analyses de la Drome Labs in Valence, France, with accreditation by Cofrac n° 1-0852, 1-1873, analyzed solid end-product samples for the following parameters (units in brackets)(followed by test procedure): TS(%)(NF EN 12880): VS (%)(NF EN 12879); pH (NF T90-008 avr 53), total ammonium (mg/kg-ds)(NF EN 25663), and nitrate (mg/kg ds)(NF EN ISO 10304-2). *E.coli* (CFU/g ds)(NF V 08-053 (11/02)) was sub contracted to Laboratoire Departmental d'Analyses et de Recherche, also having Cofrac n° accreditation 1-0551.

The conversion of total ammonium to free ammonia was based on the pH and temperature of the sample and values in the table of free ammonia percent in Emerson et al. (1975). The Solvita class was then calculated using a regression equation (Appendix Table 3.1, 3.2).

Worm counts were made from six 100-200 g samples of mature vermicompost. Worm density was determined by dividing worm mass by material mass (g-worm/g-material (wt)).

3.3.5. Statistical procedures

The detection limit value was assigned to samples when *E.coli* counts were reported at or below the detection limit. The detection limit value varied between 50-100 CFU/g (ds). JMP version 8 (SAS 2009) was used to perform analysis of variance (ANOVA) and non-parametric Wilcoxon – Kruskal Wallis tests. Parametric tests were used when all assumptions were met (normality, homogenous variances, linearity) and non-parametric methods were used when assumptions were not met. Any other transformations are described within the text.

3.4. Results and discussion

A framework emerged through discussion with senior park managers and operators, which was used to evaluate CT and UDVCT performance as waste management systems. Remote site waste management systems need to address four criteria before on-site land application of end-products becomes a realistic goal:

- Centralized collection
- Low operation and maintenance (O&M) costs
- Low risk to site operators, site visitors, local environment
- Mass reduction

Centralized collection brings fecal matter to one controlled location. This can be accomplished by a standard toilet building or a collection bin for personal pack-out containers. Operational costs vary by site and are impacted by many factors making comparisons between systems difficult. Human waste is not valuable until it can be proven otherwise and thus should be minimized to reduce costs, handling risks, and potential disposal costs. In a composting toilet mass can be lost by degrading the organic fraction of solid material which can be evaluated through reduction of volatile solids percent (%VS).

Operation and maintenance costs, exposure risk, and mass reduction were evaluated by creating a simple model based on an average remote site experiencing 5000 uses/toilet/year, located 0.5 hr by car and 0.75 hr walk away from agency headquarters, labor costs of $100/hr, $50/hr for travel expenses, and compost toilet specific input parameters sourced from manufacturer literature and from our results here (Chapter 7).

3.4.1. Centralized collection

All toilets visited were clearly labeled and recognizable as toilets, meeting the first objective of remote site waste management.

3.4.2. Operation and maintenance costs

According to instruction manuals CTs require much more operation and maintenance than UDVCTs. Field operators generally followed instruction manuals provided with units. In general CTs require daily, weekly, or monthly O&M and UDVCTs require annual O&M. The model calculations showed that it would cost $2665/year to service such a site with a CT ($0.38/use) and $273/year to service it with a UDVCT ($0.02/use). Should the site double in annual use to 10,000, the cost for CT O&M would rise to $4583 but would not change for UDVCT as the cost per use is essentially constant up to 40,000 users/year (Neau personal communications Jan. 2012). These predicted costs are the same order of magnitude as the actual ones. For example, the company supplying the UDVCT sells annual service agreements, after the initial 5 year service inclusion expires, for 500 € per year (Neau personal communications Jan. 2012). One site serviced by a CT, which experienced less than 5000 uses/toilet incurred 10 O&M visits per year, 1 hr travel/visit (50/50 toilets/other), 1hr/toilet/visit (50/50 cleaning/operations), 3hr bulking material supply and acquisition/yr, 8 hr disposal/yr (every 2-3 yr), plus $100 in fuel totaling $1800 before material disposal costs (Cieslak personal communications Sept. 2010).

3.4.3. Risk

3.4.3.1. Composting toilet pathogen reduction mechanism

Temperatures >35ºC were not recorded either at the top of the pile (most oxygen) or at the bottom of the pile at any of the sites visited. Composting regulations stipulate that temperatures >55 ºC be attained for consistent periods of times (3 days-3 weeks) to ensure adequate pathogen destruction (Haug 1993, CCME 2005, B.C. 2007). Toilets were sampled during periods of use (spring-fall for backcountry sites and winter for urban sites and ski hills) and it can be reasonably assumed that proper temperature conditions do not exist, or rarely exist, in any composting toilet sampled. All literature on the topic of compost toilets concurs that required temperature conditions do not develop in these small, field scale, decentralized systems (Redlinger *et al*. 2001, Holmqvist and Stenstrom 2002, Guardabassi *et al*. 2003, Zavala *et al*. 2004, Zavala & Funamizu 2005, Jensen *et al*. 2009, Tønner-Klank *et al* 2007). Niwagaba *et al*. (2009) was only able to attain high temperatures in insulated, bench-scale reactors with urine diverted fecal matter mixed with food waste; without food waste or insulation insufficient temperatures allowed *E.coli* and *Enterococccus* spp. to increase during the trial.

Even if proper temperature conditions develop for short periods of time, the toilet design confounds any sanitation as leachate can percolate from the above fresh fecal matter into the more mature material and re-contaminate it.

Storage time on its own is not an approved or reliable pathogen reduction mechanism (Vinnerås *et al*. 2003). Even in desiccating toilets where ash was used to elevate pH to >9, and *E.coli* counts were reduced 5 $\log_{10}$ units in one year, no reduction in helminth ova or *Clostridium perfringens* was found (Sherpa *et al*. 2009).

As no proven pathogen reduction mechanism appears to be present in CT systems, piles contaminated with resistant pathogens such as helminth ova, would contain this pathogen in end-product and potentially in leachate. Parasitic, free-living nematodes (*Rhabditis* spp. and *Diploscapter* spp.), were found together at Site 2, and the latter on its own, along with other natural soil nematodes, was found in an old dump pile adjacent to Site 1. *E.coli* in end-product are included in Section 3.5.

3.4.3.2. Urine diverting vermicomposting toilet pathogen reduction mechanism

The average worm density was 0.03±0.04 g-worm/g-material (0.024±0.030 g/ml). This density is similar to the maximum density obtained in experimental trials with sewage by Benitez (1999) (0.05 g-worm/g-feedstock), which produced mature and stable vermicompost in nine weeks, and Kumar and Shweta who eliminated all bacterial pathogens evaluated in 8 weeks with a starting density of 0.005 g-worm/g-feedstock (assuming the average mature *E. fetida* worm was 0.5 g (Yadav *et al*. 2010)). In order for earthworms to predate upon pathogens, consume fecal matter and indirectly destroy pathogens, and facilitate further pathogen reduction by altering microbial community composition, they must have a moist, neutral, and low ammonia environment (Sinha *et al*. 2009, Eastman 2011). The average TS(%) was 27±10, within the range of optimal moisture (Sinha *et al*. 2009). The vermicompost was of neutral pH and the ammonia content was low (Table 3.3). The worm density found in these samples confirms that healthy populations of worms had established, and were likely responsible for the considerable material degradation (section 3.4) and pathogen reduction (section 3.5) found here.

3.4.3.3 Risk exposure events

Exposure events were modeled at a 5000 user/toilet/yr site in similar fashion to the cost analysis; each O&M procedure (as suggested by instruction manuals) which placed the operator in close proximity or contact with raw or finished end-product was counted as an exposure event and during constant direct exposure (bi-annual shoveling) one exposure was allocated each hour. There were an average of 33 exposure events per year with CTs and 2 per year for UDVCTs.

Due to what appears to be adequate worm density and environmental conditions for pathogen destruction, UDVCTs are likely to create less O&M risk than CTs, which appear to lack a proven pathogen reduction mechanism and incur >10 times the number of annual exposure events during O&M visits. Operators of composting toilets, and especially CTs should minimize the exposure and consequences of O&M risks by wearing proper personal protective equipment, obtaining training on handling hazardous materials, working in confined spaces, and obtain blood borne pathogen vaccinations as outlined in Land (1995).

3.4.4. Mass reduction

Due to limited record keeping, difficulty collecting data on feedstock additions, leachate drainage, CO_2 respired, and end-product removal, mass reduction was estimated through the reduction of %VS, as decomposition of organic matter will reduce the organic fraction leaving behind higher ash content. The UDVCTs end-products had %VS (60±10%) that was significantly lower compared to CTs (82±13%) ($p<0.0001$). The average and standard deviation from raw faeces samples in the deposition zone was 85±4% suggesting that minimal decomposition takes place once fecal matter is buried under fresh matter in CTs.

Adding the %VS reduction to another simple model based on the same 5000 use/toilet/yr site, the required bulking agent additions as per the instruction manuals, an estimated ratio of feces:urine, average fecal deposit mass, and urine dry solids content (minimal), it was estimated that the average CT increased the total dry weight of fecal-contaminated end-product by 274% compared to original fecal dry weight. This was due to the addition of almost 4x the original mass of bulking agent as compared with fecal matter and the minimal reduction in volatile solids (Table 3.3). The bulking also will act to dilute the bacterial or other pathogen indicators, which are measured per gram total end-product. The bulking agent holds considerable water content, the average TS% being 25%, the other 75% being absorbed urine, further adding to total wet mass of end-product that must be periodically removed and disposed.

UDVCTs decreased fecal matter feedstock to 41% of the original dry weight. UDVCT average end-product TS% is similar to raw feedstock (28±11%, 25±7%); this high moisture content is necessary for worm survival. This high water content is maintained with rain-water despite leachate losses and water lost through cellular respiration.

3.4.5. Onsite disposal

As discussed earlier, proper temperatures were not attained in any cases, nor were they routinely measured by public agencies in any of the CT units. Following regulations governing discharge of human waste in British Columbia, Alberta, and the USA, this material should not be applied to public land, and should instead be sent to an approved biosolids treatment facility (B.C. 2007, Alberta 1996, WSDOH 2007). Common misinterpretation of these standards is made when a passing grade is assumed by fecal coliforms count alone (<1000 MPN/g); however, this index is only relevant when composting has been conducted and documented. Vermicomposting does not occur under temperatures required for bacterial composting, but perhaps due to its recent emergence as a waste treatment alternative, approved process conditions have not yet been developed for vermicomposting making legal land-application of vermicompost challenging. Industrial vermicomposting operations meet regulations by pre-composting vermicompost feedstock by standard microbial composting (Paul personal communications July 2011).

Full compliance with regulations at all sites visited would involve extracting material for off-site treatment. Given the stringent provincial and federal regulations governing onsite disposal of sewage/biosolids, the role of NSF/ANSI Standard 41 in assessing toilet capable of producing safe end-product is not clear. This is exacerbated by the standard's comparatively weak set of analytical methods required to achieve a passing grade in solid or liquid end-product quality (fecal coliforms, total solids, smell) and no recommendations for ongoing analysis. Fecal coliforms are one of the weakest pathogen indicators, having minimal adaptations, unlike protozoan cysts or helminth ova, for long-term survival outside of host organisms (Haug 1993, Sherpa *et al*. 2009). Furthermore, the standard appears to provide the purchaser/user an indication of material safety, which according to some manufacturers means end-product is suitable for reuse as soil amendment in residential gardening, yet the standard states that end-product management methods are not addressed by the standard. Based on what is know about composting toilet feedstocks, lack of proper temperatures, and NSF/ANSI Standard 41 test methods, it was predicted that end-product from CTs would contain highly variable fecal coliform counts and low total solids due to a high fraction of urine soaked bulking agent.

Figure 3.2 confirms this prediction; all CT site visits contained samples which failed due to high moisture content, high *E.coli* (fecal coliform) counts, or both. There was no significant difference between *E.coli* counts in CT end-product and raw material ($p=0.42$)(Table 3.3). High *E.coli* counts on their own are concerning, especially should they contain pathogenic strains such as 0157:H7, but of greater concern, these results suggest inadequate conditions exist to destroy more resistant, virulent and damaging pathogens such as hookworms as demonstrated by Sherpa *et al*. (2009). Failure to attain proper moisture content for aerobic decomposition plagues CTs (Matthews 2000, Jönsson and Vinnerås 2007). Desiccating environments are more effective at reducing fecal coliform populations in composting toilets (Redlinger *et al*. 2001). By diverting urine away from the pile aeration could be improved and possibly lower fecal coliform indicators through desiccation. However, this change alone would be unlikely to create safe end-product, as desiccation is not an approved process under provincial or federal regulations in Canada and the USA; protozoa, helminth ova, and viruses can survive desiccating environments (Haug 1993), especially where material is heterogeneous and contains fecal deposits (Gray *et al*. 1993). Sherpa *et al*. (2009) found these exact results, as they show one year of storage in urine-diverting dehydrating toilets achieved significant reduction in *E.coli* and *Enterococci* spp. but failed to induce any reduction in prevalence or abundance in hookworms, tapeworms, or roundworms (Sherpa *et al*. 2009). In a desiccating environment fecal deposits form a crust within which the moisture content remains higher and may prolong survival (Hill, unpublished). The total failure of all CTs in relation to NSF/ANSI Standard 41 and an unclear overlap of the standard in relation to provincial and federal regulations may indicate the need for an overhaul of Standard 41, especially in relation to testing and accreditation of end-product in relation to performance, safety, and regulatory standards.

UDVCTs have consistently and significantly lower *E.coli* than CTs ($p<0.0001$); Figure 3.3A displays *E.coli* counts (CFU/g-fecal), adjusted to eliminate the diluting effect of 25-50 g bulking agent added to every CT use and 2.7:10 fecal:urine usage ratio. This study is one of the first to document UDVCTs; as such there is little literature to compare to this result. The NSF/ANSI Standard 41 moisture content limit does not apply to UDVCT, which require near-saturated conditions (Sinha *et al*. 2009). These results, especially in comparison with the data from CTs, indicate that vermicomposting human waste may be a reliable process for pathogen destruction. An in-depth analysis of bacterial pathogens and bacteriophages was conducted on the end-product from UDVCTs, and will provide further insight (Lalander *et al*. unpublished).

Should end-products meet regulated standards, either through proper temperature attainment or by inclusion of vermicomposting as an approved process, other aspects of quality should also be achieved including neutral pH, high nitrate, and low ammonia (Haug 1993, Wichuk and McCartney 2010). pH was significantly lower, more neutral, and less variable in UDVCT end-product ($p=0.0047$, Figure 3.3B). Despite greater annual additions of nitrogen in urine added to CTs, total nitrate in the end-product was not significantly different from

UDVCTs (p=0.42)(Figure 3.3C). This unexpected result may have been caused by an inhibition of nitrification in CTs due to a significantly higher ammonia concentration, likely the result of urea hydrolysis. This was shown by Solvita® free ammonia value of 2.6±1.5, classifying end-product as phytotoxic and immature, which corresponds to a concentration of 4,000-10,000 mg/kg (ds)(Woods End 2000)(Figure 3.3D). Nitrifying bacteria are sensitive to high free ammonia concentrations but no composting toilets studies have evaluated the level at which toxicity occurs or addressed the impacts of high urine:fecal ratio on nitrogen biochemistry.

UDVCTs are built to hold 10-20 years of material before extraction of material is necessary. The average age of material sampled was 3 years old but some samples may have been contaminated by material 1-2 year old shoveled onto or near to oldest material. It is expected that as material ages indicators of maturity (and stability) will improve, especially if more care is taken to prevent the mixing of material each year.

3.5. Conclusion

UDVCT toilets have low annual O&M costs, employ a proven pathogen reduction mechanism and show consistently low *E.coli* counts in end-product, involve minimal operational hazards, minimize fecal contaminated leachate, require no bulking agent, reduce end-products to 41% of excrement feedstock dry mass and produce a pH neutral, mature end-product having low free-ammonia and abundant nitrate. CTs have 10 times higher O&M costs, employ no proven pathogen reduction mechanism resulting in high and highly variable *E.coli* counts, produce >10 times more operator exposure events, require bulking agent, increase contaminated end-product mass 274%, and fail to produce stable or mature end-product. CT end-product from all the sites studied is not suitable for discharge into public park environments. UDVCTs outperform CTs in relation to the fundamental objectives of remote site waste management, except in the provision of a centralized facility, which was accomplished adequately by both designs. Where worms are present in UDVCTs in sufficient density throughout the year, long storage times and adequate separation of new and old material should allow for a high degree of stabilization, maturation, and pathogen destruction. Nevertheless, disposal must be conducted according to relevant regulations and vermicomposting is not (yet) an approved method in most locations.

3.6. Tables

Table 3.1. Site characteristics of public latrine composting toilets sample. B = brand. SE = sample events. E. (m) = elevation of site. T (ºC) = the mean annual site temperature, obtained from weather station records, adjusted by dry adiabatic lapse rate. Rate (#/yr) is the toilet use per chamber per year, estimated from operator records (overnight visitors assumed =3 uses, day visitor = 0.75 uses, and full time building occupant = 2 uses). Age (y) = sample material age estimated by date of system startup and the frequency and fraction of end-product removed. RF & Freq. = the fraction and frequency of end-product removed.

Site	# of units	B	SE	E. (m)	Place	T (ºC)	Rate (#/yr)	Year Started	Year Sampled	Age (y)	RF & Freq
1	1	A	1	2200	Remote Camp	1	1700	1998	2011	>8	¼ Every 2 years
2	1	B	1	50	Urban Building	11	548	N.A.	2011	1	1 Every year
3	2	C	1	100	Urban Building	20	10220	2007	2011	5	1 Every 5 years
4	2	A	3	2100	Remote Camp	4	3213	2006	2010, 2011	3-6	¼ Every 2 years
5	2	C	1	900	Parking Day-Use	0	45000	2010	2011	1	1 Every year
6	2	A	2	2000	Remote Camp	4	2754	2007	2010	2-3	¼ Every 2 years
7	1	C	1	75	Parking Day-Use	11	20000	2010	2011	1	TBD
8	1	D	1	550	Remote Camp	-2	720	2010	2011	1	½-1 Every year
9	1	A	2	2000	Remote Day-Use	2	8000	Before 2007	2010, 2011	>3	¼ Infrequent
10	1	A	1	600	Remote Day-Use	9	3360	2000	2011	>6	¼ Every 2 years
11	1	A	1	2000	Remote Camp	1	5000	2009 Sept	2010	1	¼ Every 2 years
12	1	E	1	563	Parking Day-Use	7	5000	2006 June	2012	5	¾ Every 10-20yrs?
13	1	E	1	218	Remote Day-Use	11	6000	2009 April	2012	3	¾ Every 10-20 yrs?
14	1	E	1	218	Parking Day-Use	11	6000	2009 Nov.	2012	2	¾ Every 10-20 yrs?
15	1	E	1	2160	Remote Day-Use	-4	5000	2008 Oct.	2012	3	¾ Every 10-20 yrs?
16	1	E	1	2000	Remote Day-Use	-4	6000	2009 Oct.	2012	2	¾ Every 10-20 yrs?
17	1	B	1	2000	Remote Camp	20	360	2000	2011	0.5	Every few months

Table 3.2. Compost toilet brand characteristics. Style refers to system design: PF = Plug Flow; T= Tank; B= Batch. Process refers to the mechanism of decomposition and sanitation: MC = Microbial process; VC = Vermicompost. Size (L) refers to the average size of units found at each site. Max Rated Use (#/toilet/yr) refers to the designed max capacity as found in manufacturer reference documentation. O&M refers to the frequency of operational and maintenance tasks as per manufacturer reference documentation.

Brand	# Sites	# Cham.	Style	Process	Size (L)	Max Rated Use (#/toilet/yr)	O&M	Bulking agent suggested/use (g)	Leachate contamin. of end-product possible?
A	6	8	PF	MC	2500	36000	Weekly	50	YES
B	2	2	PF	MC	164	1460	Weekly	40	YES
C	3	5	PF	MC	4500	30000	Weekly	24	YES
D	1	1	T	MC	500	N.A.	N.A.	N.A.	YES
E	5	5	B	VC	3000	15000	Annual	None	NO

Table 3.3. Characteristics of raw material, CT end-product, and UDVCT end-product. Means (and standard deviation). T (ºC) = the mean annual site temperature, obtained from weather station records, adjusted by dry adiabatic lapse rate. *E.coli* (CFU/g-fecal) is *E.coli* (CFU/g) adjusted to remove dilution effect of bulking agent on CT samples based on the estimated fecal : total mass ratio 0.23:1 gram fecal origin : gram final dry mass. For sites sampled >1 time, only data from the site-visit with the greatest sample size has been included.

	Brand	n	Elev. (m)	T.(ºC)	Age (yr)	%*VS*	*E.coli* (CFU/g)	*E.coli* (CFU/g-fecal)	Nitrate (mg/kg ds)	pH	Solvita® NH_3
Raw	A	23	2075	5	<0.5	85 (4)	8550 (2340)	8545 (2340)	286 (1132)	8.1 (0.5)	1.4 (0.7)
CT	A, B, C	46	1202	7.5 (7.5)	3.6 (2.8)	82 (13)	19200 (60800)	71100 (225000)	1300 (429)	8.0 (0.7)	2.6 (1.5)
UDV CT	E	15	1031	4.4 (7.4)	3.0 (1.2)	60 (10)	200 (245)	200 (245)	1961 (700)	7.4 (0.3)	4.0 (0)

3.7. Figures

Figure 3.1. Mixed latrine style microbial composting (CT) toilet and source separating vermicomposting (UDVCT) toilet design diagram showing physical chambers, inputs, and solid and liquid material outputs.

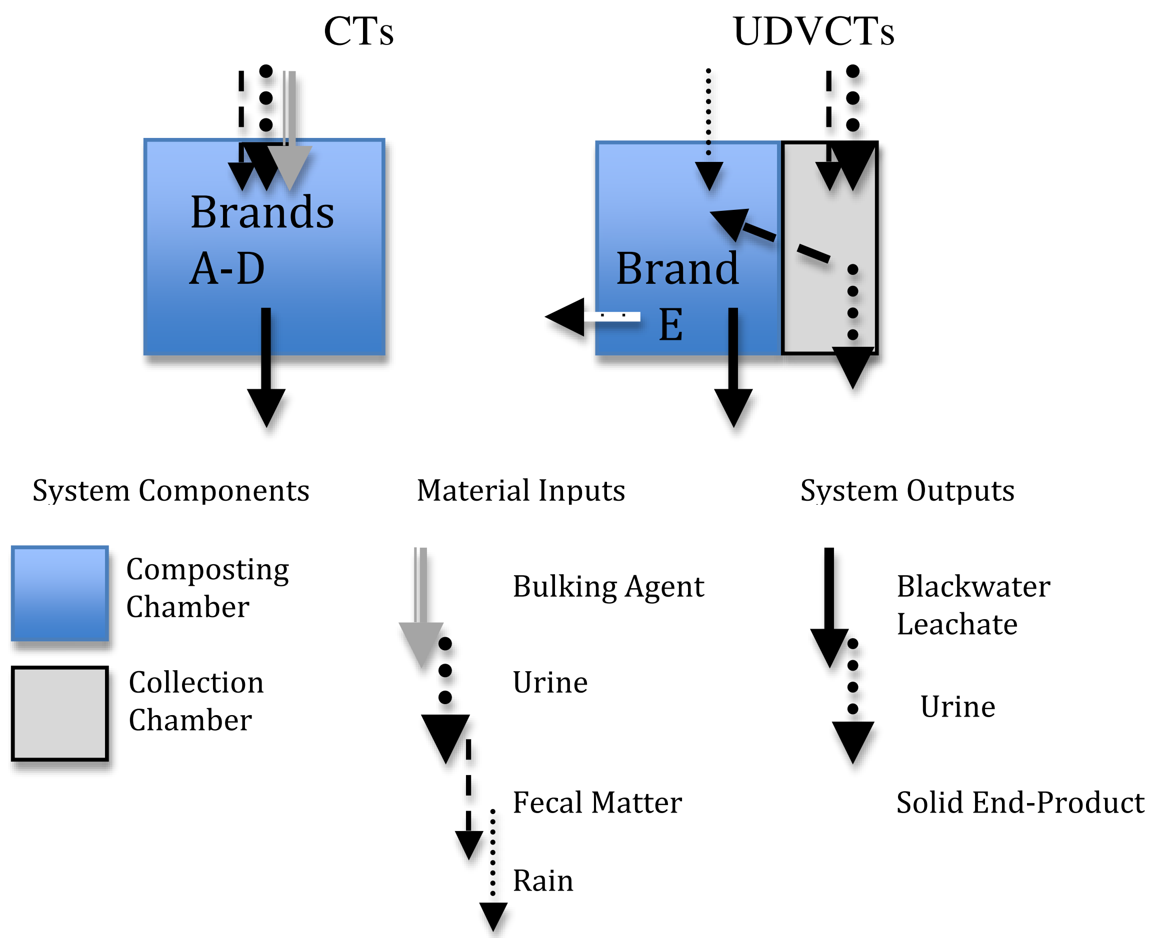


Figure 3.2. Performance of composting toilets with solid end-product material grouped in site-sample-visits (Site.Brand.Chamber.Year.Season) as compared with NSF/ANSI Standard 41 TS% and *E.coli* (CFU/g) as a subset of fecal coliforms. All data points per site-sample-visit plotted with site-sample-visit mean lines.

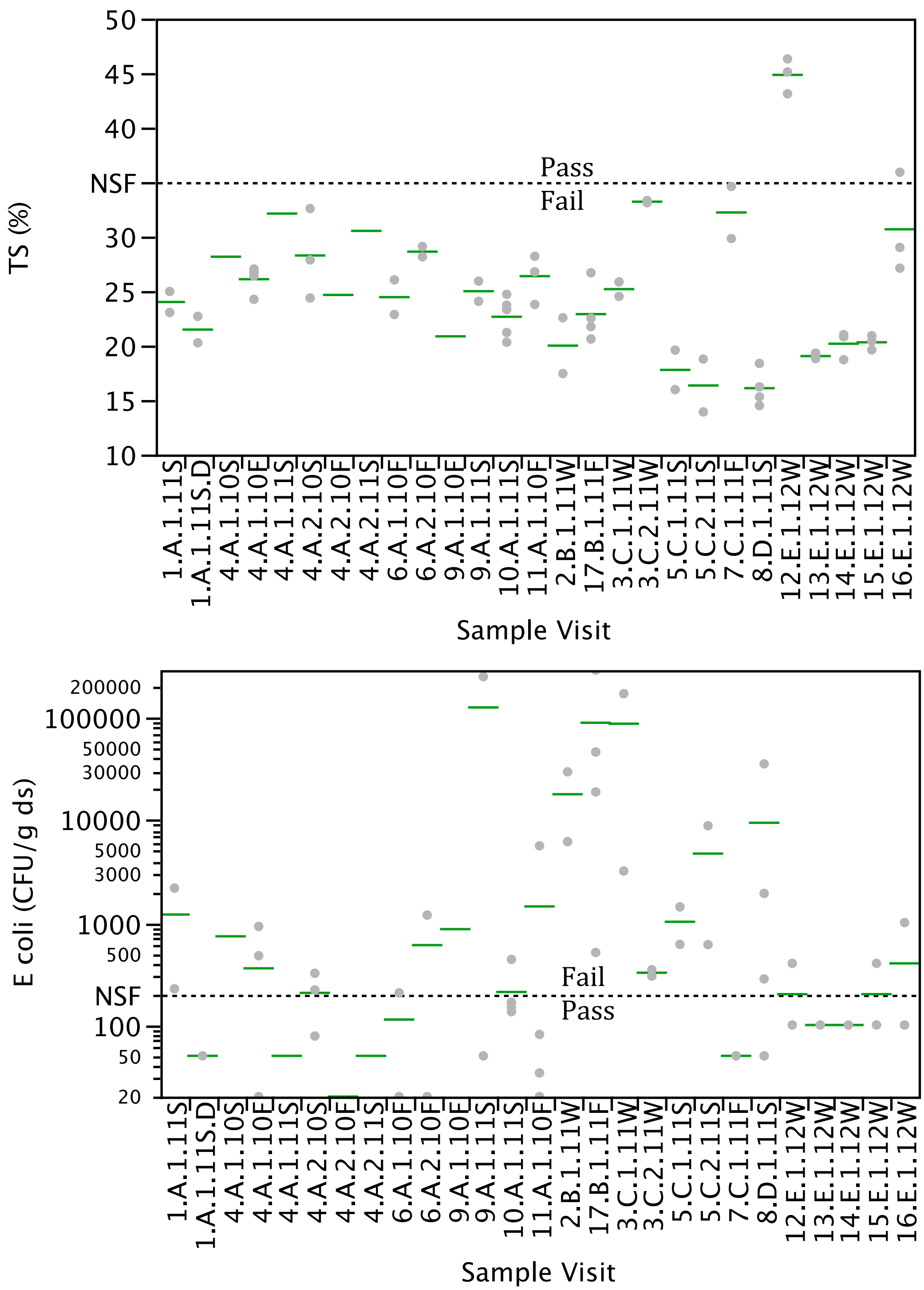
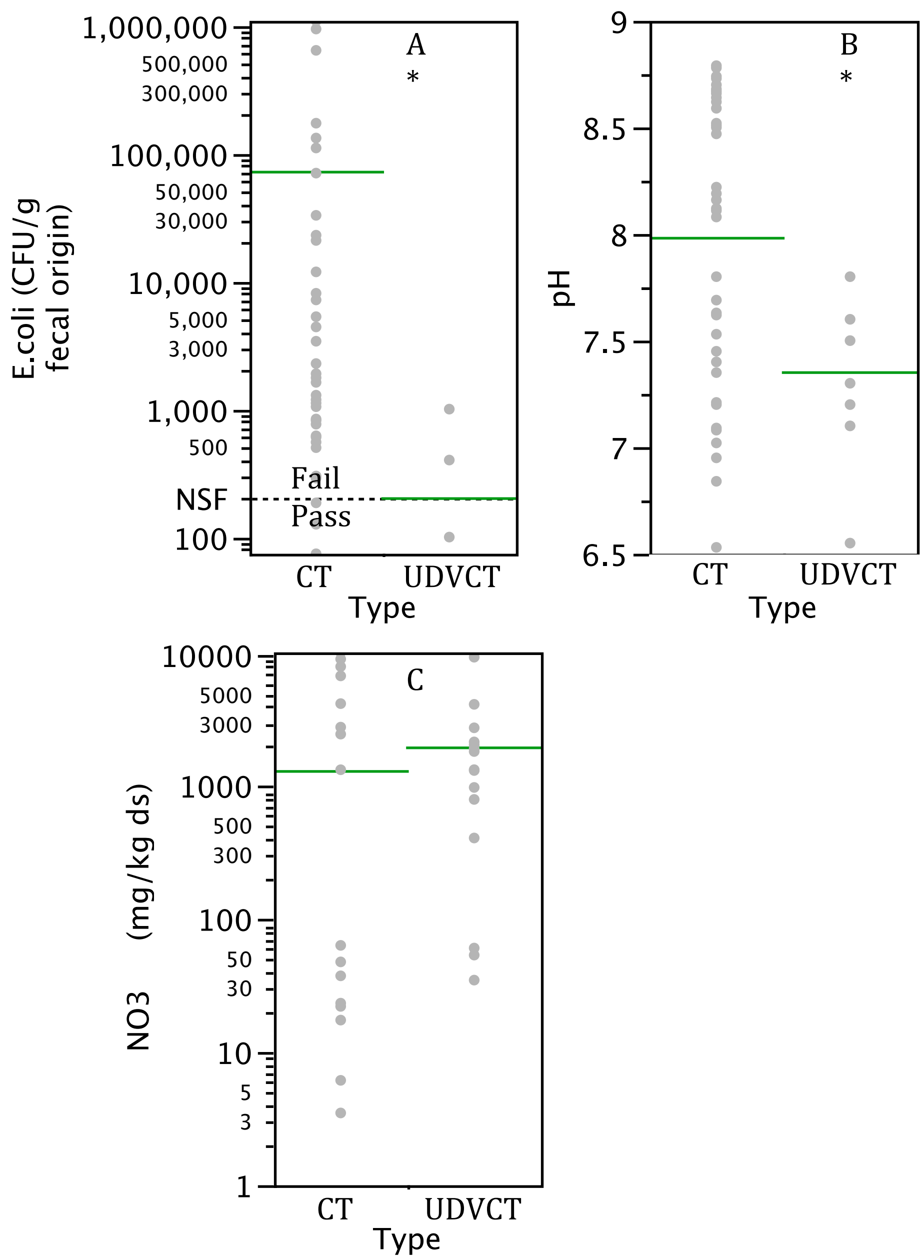


Figure 3.3. Comparison of average end-product quality between mixed latrine style microbial composting (CT) toilet and source separating vermicomposting (UDVCT) in A) *E.coli* (CFU/g(fecal origin)), B) pH, C) nitrate (mg/kg ds). All data points per site-sample-visit plotted with site-sample-visit mean lines. Significant ($p<0.05$) differences marked with a *.



Chapter 4. Composting toilets a misnomer: excessive ammonia from urine inhibits microbial activity yet is insufficient in sanitizing the end-product

4.1. Abstract

End-product from 15 public mixed latrine style composting toilets (CTs) at 11 sites between 50-2100 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%, *E.coli*, NO_3^-, and pH). Principal components analysis indicated that TS%, temperature, and material age equally contributed to reduction in VS%). NH_3-N had the greatest effect on NO_3^-, *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.

4.2. Introduction

Chapter 3 exposed the failure of CTs to meet relevant standards for compost quality, stability, maturity, and hygiene. However, the reasons for the failure have not been fully evaluated. It may be possible to isolate the root causes of the failure by applying multivariate statistical techniques to this large data set. Key sets of variables must be chosen which are known to affect the composting process and which can be used as indicators of performance. These variables are then used to explore and draw conclusions from the large data set, and make recommendations with regards to the fundamental objectives of waste management at remote sites.

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 microbial consumption of organic matter. To sustain microbial composting, organic matter must have an appropriate ratio of biodegradeable 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 55ºC or more for three days to three weeks (depending on which composting process is used) to kill and adequately sanitize pathogens (CCME 2005, B.C. 2007). The World Health Organization (WHO) guidelines recommend that composting of toilet waste should be performed at 50°C or higher for 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 at a centralized secondary treatment. CTs are commonly perceived and advertised as being capable of producing 'compost' onsite, a notion which can be traced 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 reuse 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, Tonner-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 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), Tonner-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), Vinnerås (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), Vinnerås *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 we know, a comprehensive exploration of root causes of failure from in-field public, mixed latrine style microbial composting toilets (CTs) has not been conducted in North America.

The objectives of this study were to apply multivariate statistics to our comprehensive data set of end-product quality and process variables from public and in-field CTs 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 be 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.

4.3. Methods

4.3.1. 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. Eleven sites, with 15 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 of the buildings utility space in which the unit was installed was heated. All toilets sampled were commercial units, sized and installed professionally. Despite four different brands sampled and 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 between 50 m and 2100 m elevation and between 46ºN and 61ºN. 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 Table 3.1 and Hill and Baldwin (2012).

4.3.2. 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 at 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 Section 3.3.

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.

4.3.3. Samples

Only samples from the oldest end-product in each chamber were 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 200ml 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 hours of sampling and a minority in 72 hours.

4.3.4. Biochemical analyses

Benchmark Labs in Calgary, Alberta, an ISO 17025 accredited Lab, analyzed solid end-product samples according to Table 4.1.

4.3.5. Statistics

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

Based on the literature and variables measured in our study the following key variables (and their impact on the compost process) were chosen for utilization in the PCA: 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 *E.coli* (pathogen content).

4.4. 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 (-2ºC) was installed in a significantly lower ambient temperature environment than brands A, B, & C (4, 7, 10 ºC). However, despite this, no difference was found between brands in end-product quality (pH, VS%, NO_3^-, *E.coli)* or NH_3-N ($p>0.05$). Samples were grouped together by brand.

Figure 4.1A 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 (Appendix Figure 3.1). 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 and trail toilet samples cluster away from camp and commercial sites positively by Component 1 (Figure 4.1B). The majority of samples in the lower left quadrant are <3yrs old and in the upper right corner are >5yrs old (Figure 4.1B). 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^- after temperature.

Regression analyses were used to explore meaningful correlations. Univariate relationships between NO_3^- and TS%, VS%, NH_3-N, pH, temperature, and material age were inspected and tested with linear regressions. Significant negative regressions were found between NO_3^- and NH_3–N ($p=0.0054$) (Figure 4.2A), and temperature ($p=0.032$) (not shown) but neither relationship was well described by a linear function. High NO_3^- 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 >8ºC (3589±3807 mg/kg, n=7, T_{avg}=13.0ºC) than in samples <8C (547±1943 mg/kg, n=40, T_{avg}=2.4ºC) ($p=0.0022$). The only significant and meaningful correlation was found between NO_3^- and pH ($p<0.0001$) (Figure 4.2B).

Toilets placed at trail 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 $p=0.0011$) (Appendix Figure 3.2).

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 4.3). All were weak fits having r^2 values below 0.2. The significant relationship between TS% and *E.coli* was found acting in the expected direction where drier samples contained less *E.coli*, but the probability and fit were the weakest of the three significant variables and TS% in all samples was lower than optimal (Redlinger *et al*. 2001, Zavala and Funamizu 2005).

4.5. Discussion

Despite minor differences in brand, the basic design, operation, feedstock, and amendments utilized by all brands were similar (Chapter 3). 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 differences with larger sample sizes, but the relevant magnitude of the results are unlikely to be meaningful in comparison to relevant standards as discussed in Chapter 3 where all CT sites tested failed NSF/ANSI Standard 41.

To obtain highest stability and most degraded end-product, samples should land in the upper right quadrant of Figure 4.2B; 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).

Composting for days to weeks at temperatures of 55ºC 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 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 (Chapter 3). Even when proper 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). Adequate 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 CTs 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 adqeuate 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, 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 2009, Nordin *et al*. 2009A). With lower urine inputs there will be less urea hydrolysis and less ammonia (and hydroxide) production. If ammonia is an inhibitory factor in nitrification, more nitrate should be found where ammonia concentrations are lower. 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 *Nitrosomoas* 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 Figure 4.1 and relationships plotted in Figure 4.2 suggest nitrification is most affected by pH and ammonia not just lack of oxygen (low TS%) or by high VS%. 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 Figure 4.2A. The data set was divided at this threshold, 386 mg/kg< NH_3-N <386mg/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. The strong negative correlation between NO_3^- and pH 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), having a pH of 8.7, 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 heterotophic uptake and for nitrification to occur during the low use season, the pile would need to reach proper temperature and moisture conditions. Denitrification could also be occurring where the right environment exists such as within fecal deposits, but it is suspected that the importance of this process of nitrogen loss is low, despite the high moisture content of the average sample, due to the low bulk density/high porosity bulking agent and presumed availability of oxygen.

The sample with both ammonia and nitrate came from a trailside toilet in a desert environment with a mean annual air temperature of 9ºC 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^-, a variety of explanations are likely: 4 of the 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 and where temperature may inhibit nitrification.

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

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 Figure 4.1B, suggesting that of all CT sites, trail placements produce the highest quality end-product (higher TS%, lower VS%, lower *E.coli*, and higher NO_3^-). There was no significant difference in *E.coli* content by toilet placement.

In general, it appears that sanitation and maturity in CT end-product are mutually exclusive; low urine additions may enable nitrification from NH_4^+ to NO_3^- but the lack of NH_3-N toxicity can result in higher *E.coli* (Figure 4.1A) (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 CTs should be clear. CTs 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). Higher NH_3-N and pH are most responsible for the decrease in *E. coli* numbers, both of which had significant negative impact on *E.coli* and have been found capable of disrupting cellular activity and destroying 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.,* unpublished) show that addition of wood shavings can increase the bacterial survival in source separated feces compared to plain storage. pH adjustment is a common method for pathogen reduction in biosolids (WHO 2006) and has a stronger negative linear regression 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 stable than lime, and is more effective in reducing pathogens including helminth ova in heterogenous fecal matter at lower pH (Allievi *et al*. 1994, Moe & Izurieta 2003, Mendez *et al*. 2004, Pecson and Nelson 2005, Nordin *et al*. 2009B).

Pathogen reduction in 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 14ºC 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 99% pathogen destruction in 6 to 60 days for *Ascaris suum* at 34 ºC and 24ºC, respectively (Nordin *et al*. 2009B). McKinley *et al*. (2012b) 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 500-1500mg/L and pH 10-12 at 20ºC.

Despite the long residence times in CTs studied here (1-8 years) and significant regression of NH_3-N with *E.coli*, natural ammonia 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 1.5-2 yr storage at 2-20ºC). If *E.coli*, are this abundant, and should leachate be primarily responsible for high pathogen counts through recontamination (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. Indeed, in most cases the ammonia levels found were below that necessary for hookworm ova destruction over 90 days at pH of 12 (Pescon and Nelson 2005). Free-living parasitic nematodes of the genus *Diploscapter* and *Rhabditis* were identified in material by Benchmark Labs 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 24ºC due to low ambient temperatures at most CT sites (Vinnerås 2007). Between 2-20ºC, the WHO (2006) recommends 1.5-2 years storage time (without recontamination risk) for adequate sanitation.

Vinnerås *et al*. (2009) estimated an NH_3-N concentration of 75 mmol/l as necessary to reduce *Ascaris suum* in human fecal matter by 6 $\log_{10}$ units at 34ºC in 28 days; the concentrations necessary to achieve the same result at 24ºC was ~10 times greater (610 mmol/l). Two percent urea by wet weight added to source separated fecal matter at 14ºC 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*. 2012b). 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 (UDVCTs) 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 UDVCTs should inspire a reanalysis of CT capabilities by current operators and re-design by manufacturers, especially when the intended site is in a remote park or protected area.

4.6. Conclusion

Temperature, moisture content, and material age act together as expected, in the aerobic decomposition of CT 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 (both of which are required for safe end-product production) appear mutually exclusive and controlled by urine, through toxic effects of ammonia and pH on microorganisms including nitrifying bacteria and *E.coli*. The adding of bulking agent exacerbates the O&M requirements yet does not contribute significantly to the sanitation, maturity, or decomposition of the primary waste, human feces.

The addition of chemical amendment would require constant 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 processes in the toilet chamber is defined by slow degradation that in many cases are hampered by high ammonia content from 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 UDVCTs demonstrate a potential model for simultaneous nutrient recovery from urine and stabilization and sanitization of fecees, end-products are still eventually disposed off-site to further minimize site and environmental impacts at apparently little extra cost over the life of the system. Disposal happens very infrequently (every 10-20 years) due to the highly effective urine diversion and solids decomposition process, resulting in very little average annual disposal cost. By focusing on efficient waste reduction processes the costs associated with final residuals management become relatively minor and the preservation of the environment can be prioritized, regardless of the reduction in risk achieved through vermicomposting. This set of priorities should set the standard in remote site human waste management.

4.7. Tables

Table 4.1. Parameters used to evaluate the root causes of CT failure. Analyses with a * were conducted by Benchmark Labs, Calgary, in the evaluation of compost quality.

Parameter	Test Name / Description	Indicative of:	Units
Annual average ambient site temperature (Temperature)	Extrapolated from nearest climate station and adjusted for elevation (1ºC / 100m)	Pile temperatures which seldom rise above ambient temperatures	ºC
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-N (NH_3-N)*	Cold-water-shake 1:2 sample:water, followed by measurement with a Thermo Scientific Orion high performance ammonia ion electrode at 20ºC according to manual instructions for free ammonia concentrations of >1ppm	Urine	mg/kg (ds)
pH*	Cold water shake 1:2 sample:water, followed by measurement with VWR symphony pH probe at 25ºC	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	%

Table 4.2. Correlations between NO_3^- (a compost maturity index) and TS%, VS%, NH_3-N, pH, temperature, 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 (ºC)	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

Table 4.3. *E.coli* (CFU/g) in end-product from public latrine style composting toilets linearly regressed with ammonia, pH, and TS%.

	p value	r^2	*E.coli* (CFU/g) regressions	n
NH_3 (mmol/l)	0.03	.085	= 6.7-0.001*NH_3	47
pH	0.0038	0.17	=17.9-1.5*pH	47
TS%	0.049	0.07	=9.27-0.13*TS%	53

4.8. Figures

Figure 4.1. 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 (y) and toilet placement as per legend.

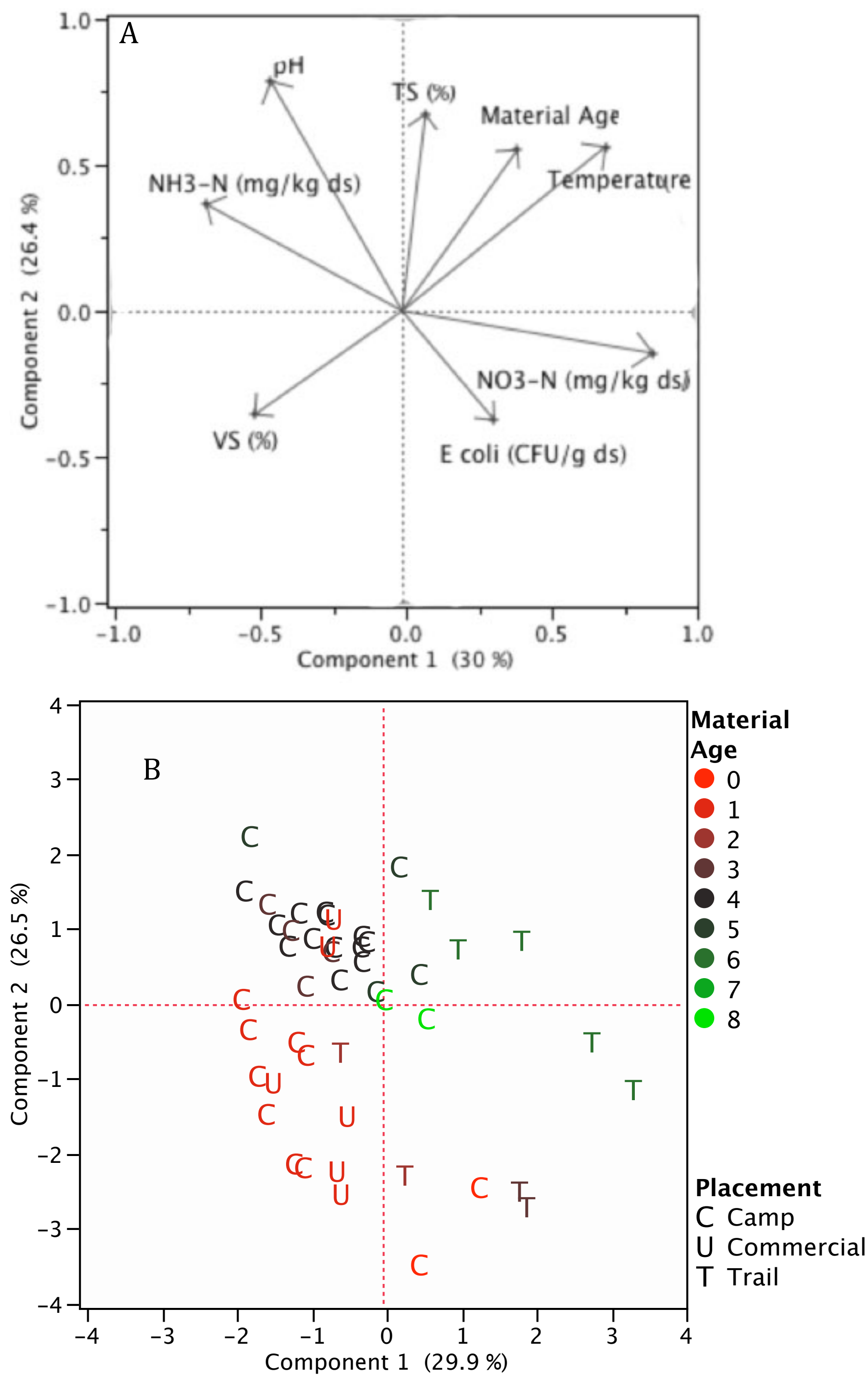
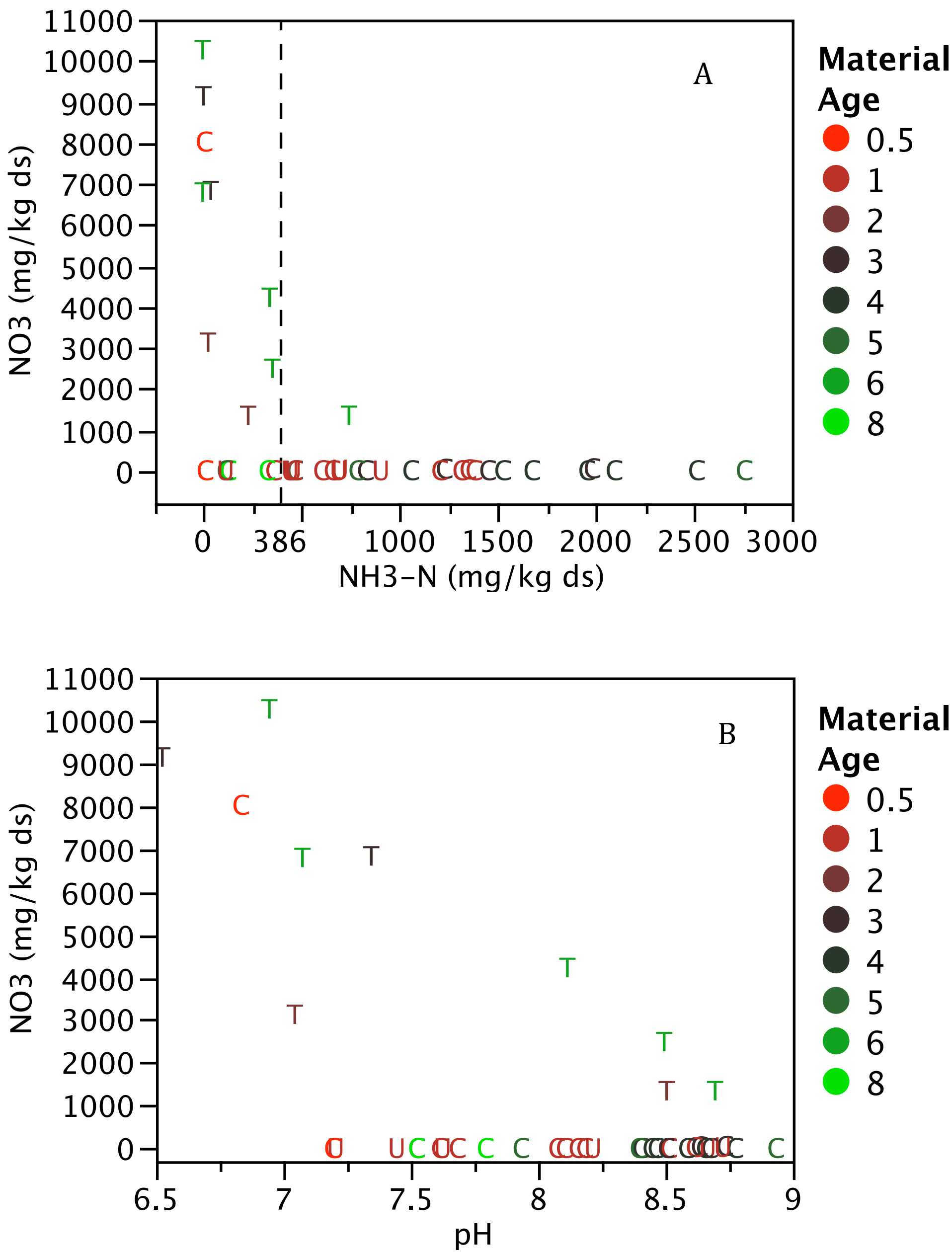


Figure 4.2. Nitrate concentration from public mixed latrine style composting toilet end-product plotted against NH_3-N (A) and pH (B). The dotted line in Figure 4.2A is at 386 mg/kg (ds) NH_3-N; the proposed upper limit below which significantly more NO_3^- is found than above ($p<0.05$).



Chapter 5. Evaluating Solvita® compost stability and maturity tests in the assessment of mixed latrine style composting toilet end product quality and safety

5.1. Abstract

It is challenging and expensive to monitor and test decentralized composting toilet systems, yet critical to prevent the mismanagement of potentially harmful and pathogenic end-product. Solvita® compost quality tests are simple and inexpensive and may be suitable for this application. Solvita® stability (CO_2) and maturity (NH_3) test results from end-product extracted from 15 public, latrine style composting toilets, were compared to laboratory tests for stability (CO_2 evolution and percent volatile solids), maturity (ammonia and nitrate), safety (*Escherichia coli*) and compost quality (pH and percent total solids). Samples with Solvita® NH_3 values 1-3 were confirmed immature by high ammonia and low nitrate concentrations whereas high value (4-5) samples were not consistently 'mature' due to the continued presence of high ammonia concentrations. *E.coli* increased with increasing Solvita® NH_3 (decreasing ammonia) and a causal relationship whereby ammonia toxicity eliminates both pathogens and beneficial microorganisms is supported. A significant negative relationship between Solvita® NH_3 and CO_2 evolution was found suggesting an *insitu* inhibition of respiration by ammonia. An unexpected significant positive relationship was found between Solvita® CO_2 (high numbers indicate stability) and CO_2 evolution (high numbers indicate *in*stability) indicating that the two methods produce inversely correlated results for respiration. This was due to the measurement of *insitu* respiration using Solvita® CO_2 paddles whereby low respiration results were caused by either an inhibitory agent (ammonia) or true material stability. The CO_2 evolution procedure eliminated toxic inhibitions prior to testing and reported a value relating to respiration potential. Many inhibited samples had high potential respiration, giving rise to the unexpected relationship between the field test (Solvita®) and the lab test (CO_2evolution). Should high Sovlita® CO_2 values (stable) be found together with high Solvita® NH_3 values (not inhibited by ammonia) it would be

reasonable to interpret that composting is moving towards completion. However, few samples having both were found in the field. Finally, in order to make comparisons to relevant performance and quality standards, the large diluting effect of inert bulking agent should be removed by sieving prior to testing.

5.2. Introduction

Decentralized waste treatment systems experience greater variability than centralized systems due to many factors include inconsistent feedstock supply rates, high surface area to volume ratios, and a greater influence of uncontrollable environmental parameters. Composting toilets (CTs) are a commonly used system for decentralized waste treatment at remote sites. As presented in Chapters 3 and 4, the end-product quality from CTs was found to vary greatly. Toilets receiving large amounts of urine developed high concentrations of ammonia, which was easy to detect in end-product by its foul odor (Chapman 1993). Raw fecal matter has a distinct light brown color whereas humified organic matter is usually dark brown or black. However, beyond these coarse indicators, most quality measurements require special laboratory equipment for analysis of: microbial activity (stability), concentrations of nitrogen species (maturity), and pathogen content (safety).

CTs are typical of decentralized systems and suffer from similar management challenges including: poor operational understanding; limited accountability; vague responsibility; and limited data collection (Willetts *et al*. 2007). As a result, with regards to public composting toilets in North America, there is no standard protocol for testing, so testing is not commonly or regularly conducted.

The lack of proper protocols and low regulatory control combined with erroneous popular perception of CTs capability results in end-product from CTs in some locations being discharged onsite without any testing. Accompanying the discharge of inadequately decomposed human biosolids are a suite of human health and environmental risks including pathogen transmission, phytotoxic effect on terrestrial plants, and eutrophication of aquatic environments (Wichuk and McCartney 2010).

In larger waste treatment systems, tests are commonly conducted to verify that the process is operating within acceptable limits, proven to accomplish desired level of treatment and sanitization and consistently produce desired quality end-product (Haug

1993, Wichuk and McCartney 2010). End-product testing focuses on microbial pathogens, degree of organic matter stability, chemical nutrient maturity, and more generic biochemical quality parameters. Samples are generally extracted from representative locations and sent to laboratories capable of conducting the desired analyses, which are summarized in Wichuk and McCartney (2010). However, laboratory analyses can be expensive and often require samples be received rapidly after extraction. Both factors introduce further obstacles to the testing of remote CT end-product as many facility operators do not have budgets for ongoing testing nor skilled personnel to conduct it.

Solvita® compost test paddles, made by Woods End Labs Inc. (Mount Vernon, Maine, USA) can be used without special laboratory equipment or training to evaluate compost stability and maturity making them a valuable, inexpensive, and simple tool suitable for decentralized compost assessment (photograph in Figure 5.1) . Solvita® is an approved analytical method for commercial compost evaluation (Wichuk and McCartney 2010). Despite minor limitations imposed by the volumetric basis of measurement and subjective analysis of water content, these tests have proved reliable in differentiating mature from immature composts, correlate well several indices of stability and maturity, and are easily conducted in the field (Brewer and Sullivan 2001), Yet, as far as we or the manufacturer know, Solvita® test paddles have not been used to evaluate end-product in CTs (Brinton, personal communications July 2010).

The Solvita® test is conducted at room temperature (20ºC) for four-hours. The samples are placed with two gel paddles, one for ammonia and one for carbon dioxide in a standardized sealed jar and from the change in colour of the paddles the maturity and stability can be interpreted. Corresponding values of ammonia volatilized and carbon dioxide respired are roughly translated to more standard maturity and stability indices but are noted to vary by pH, temperature, and bulk density; comparisons can be found in Wichuk and McCartney (2010).

It was the aim of this study to test the reliability of Solvita® NH_3 and CO_2 test paddles on CT samples collected from public CTs against equivalent quantitative laboratory measures of: maturity (ammonia measured with an ammonia gas probe); stability (percent volatile solids and CO_2 evolution); content of *Escherichia coli*; and generic compost quality (percent total solids and pH). We tested these relationships with alternate hypotheses based on progressions in standard microbial composting processes where stable and mature compost end product (high Solvita® CO_2 and NH_3 values) is indicated by low CO_2 evolution rates, low volatile solids percent, low ammonia concentrations, neutralized pH, unsaturated moisture conditions, and ideally low *E.coli* indicator concentration (Haug 1993).

If the test accurately reflects the state of stability and maturity and provides insight into material safety, it may become an inexpensive and valuable tool in the challenging assessment of CT end-product quality. This could lead to improved management of CTs and a reduction in risks and consequences associated with the discharged of incompletely sanitized, immature or unstable human biosolids.

5.3. Methods

5.3.1. Sites

The same set of CT data used in Chapters 3 and 4 was used in this Chapter. A summary of site characteristics can be found in Section 3.3 and Table 3.1.

5.3.2. Composting chambers

The same set of CT data used in Chapters 3 and 4 was used in this Chapter. A summary of chamber characteristics can be found in Section 3.3.2.

5.3.3. Samples

The same set of CT data used in Chapters 3 and 4 was used in this Chapter. A summary of chamber characteristics can be found in Section 3.3.3.

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5.3.4. Biochemical analyses

Samples were extracted from chambers, placed in sterile jars, and shipped by courier with ice packs as soon as possible (usually <24hrs) to Benchmark Labs in Calgary, Alberta, an ISO 17025 accredited Lab, analyzed solid end-product samples according to Table 5.1. Samples were either processed immediately upon receipt or stored for 24-48 h at 4^{o}C prior to processing.

5.3.5. Statistics

E.coli was log-transformed to meet modeling assumptions. Lines fitted through log *E.coli* data were log transformed. *E.coli* counts below the limit (which varied from 20-50 CFU/g) were reported at the detection limit. No other adjustments were made to the data set. JMP version 8 (SAS 2009) was used to produce graphical displays and for correlation / regression procedures and statistics.

5.4. Results and discussion

5.4.1. Maturity

The Solvita® NH_3 test correlated significantly with ammonia measured in solution with an Orion probe (Table 5.2). The linear fit line shows close confidence intervals with limited scatter in Solvita® NH_3 values 3-5 (Figure 5.2A). Considerably more scatter exists at Solvita® NH_3 values 1-2 which are categorized as immature and are at the highest end of ammonia concentration by the Solvita® NH_3 scale (Figure 5.2A). This variability may have evolved from a range of ammonia concentrations able to produce the maximum color change in less than the four-hour test period. As predicted by equilibrium equations, pH and Solvita® NH_3 are inversely correlated (Figure 5.2B); a decrease in pH will increase the ratio of ammonium:ammonia which would result in a high Solvita® NH_3 value showing a false indication of higher maturity . However, maturity is generally evaluated with total ammonium or ammonium:nitrate ratio (which incorporates a measure of completeness), where a pH adjustment step shifts all ammoniacal nitrogen into the ammonium form (Wichuk and McCartney).

Values 1 through 3 from the Solvita® NH_3 test clearly categorize CT samples here as immature as per accepted indices of compost quality (Table 5.3). Conversely, maturity characteristics of the 10 mature samples having Solvita® NH_3 values 4-5 presented in Table 5.4, did not relate to thresholds of maturity; mean total ammonia and mean ammonia:nitrate was much higher than the 300 mg/kg and 3:1 limit designated by ASCP (2001). Solvita® NH_3 tests can definitively inform operators when CTs are being impacted by ammonia from excessive urine (values 1-3 designated immature) but with this limited data set, they appear unable to provide a consistent, definitive measure of maturity for CT samples having values 4-5.

The Solvita® manual provides means of estimating total ammoniacal nitrogen (mg/kg) from Solvita® results (Woods End 2000, Appendix Table 3.1). Ten samples were randomly selected from the data set, total ammoniacal nitrogen estimated using tables from Emerson *et al*. (1975) and compared with data provided in the manual for a crude evaluation of fit (Appendix Table 3.2).

The majority of samples (10/31)were Solvita® NH_3 value 1; total ammonia in five of six randomly selected samples having a Solvita® NH_3 value of 1 fit with predictions in Table #6 of the Solvita® manual (>4000 mg/kg ammoniacal nitrogen) (Woods End 2000, Appendix Table 3.2). However, ¾ of the remaining samples (having Solvita® NH_3 values 2-4) were not accurately estimated in comparison with the values provided in the Solvita® manual. This error may have arisen because CTs have very high ammonia and total ammonium concentrations in comparison with most other composted end-products (Tables 3 & 4) for which the Solvita® tests were likely built and calibrated. The Solvita test is less accurate for ammonia concentrations higher than 4,000 mg/kg. However, the ammonia test indicates when the ammonia levels are high and require mitigation measures, even if the accuracy of the concentration is weak.

If immature CT material is discharged onsite in public environments further additional measures such as direct incorporation into adequate soil into areas where public is informed with signage and access is prevented in order to reduce risk for disease transmission and negative environmental effects. A better method of risk reduction would involve sufficient treatment of residuals prior to discharge (Chapter 3).

5.4.2. Safety

The positive (albeit weak) correlation between *E.coli* and Solvita® NH_3 was unexpected (Figure 5.2C; it was hypothesized that pathogen content would decrease with increasing maturity (inverse relationship). The underlying relationship may be causal as ammonia has a toxic effect on pathogens at high enough concentrations (Mendez *et al*. 2004, Vinnerås *et al*. 2008) and can inhibit the decomposition process (Anthonisen *et al*. 1976, Woods End Labs 2000, Chapter 4). In-situ techniques such as sentinel chambers have been useful in evaluating impacts on pathogens without contaminating the entire pile (Tønner-Klank *et al*. 2007). Research is required on the holistic impacts of naturally produced ammonia from excessive urine in a CT on pathogens such as *E.coli*, more infectious and resistant pathogens such as helminthes and viruses, and on composting bacteria, nitrifying bacteria, and detritivore invertebrates.

However, given the seemingly mutually exclusive objectives of simultaneously (in the same pile) eliminating pathogens with ammonia, oxidizing nitrogen from ammonium to nitrate, and stabilizing fecal matter, it may be prudent to move beyond the mixed latrine style CT for designs which can accomplish objectives simultaneously. Source separation of urine from fecal matter enables simple and efficient treatment (and utilization) of nutrients in urine whilst separately stabilizing and destroying pathogens in fecal matter through vermicomposting or invertebrate decomposition (Davidson *et al*. 2006, Hill and Baldwin 2012).

5.4.3. Stability

Figure 5.2D shows significant correlation between Solvita® NH_3 and CO_2 evolution. This correlation suggests either a strong underlying co-variate or supports the hypothetical causal relationship where high ammonia concentrations inhibit in-situ decomposition and result in high respiration rates when equilibrated and optimized for this decomposition test in the laboratory. Neither Solvita® NH_3 nor CO_2 evolution correlated significantly with TS% (not shown) and material age does not appear to be influencing the relationship (Figure 5.2D) lending support to the inhibition hypothesis.

Stabilized organic matter (high Solvita® CO_2 values) should have low microbial activity and low CO_2 evolution rates (Wichuk and McCartney 2010). However, Figure 5.3A shows that samples with stable Solvita® CO_2 values (6-8) were associated with some of the highest CO_2 respiration rates; this positive and significant relationship was not expected and provides further support for in-situ inhibition of decomposition by ammonia. Considerable variation and a weak r^2 may limit the value of this relationship, but nevertheless the significant p value suggests that the two tests are collecting different and opposing information. When the Solvita® CO_2 test is conducted according to the instruction manual, and shortly after removal of the sample(s) from the surrounding pile, the results provide a measurement of the state of microbial respiration and activity within the pile. Whereas the CO_2 evolution test required samples be equilibrated for 3 days to remove inhibitory substances prior to testing in sealed containers and more accurately identify potential (or absolute) stability (TMECC 2002, Wichuk and McCartney 2010).

The relationship between pH and Solvita® CO_2 also supports this theory of inhibition by ammonia: pH trends from alkaline to neutral as material decomposes as a result of the production of intermediate acids, volatilization of ammonia, heterotrophic assimilation of ammonium, and nitrification of ammonium to nitrite (in inquilibrium with nitric acid) and nitrate (Anthonisen *et al*. 1976, Haug 1993, Wichuk and McCartney 2010). Yet Figure 5.3B and C indicate that samples with highest (most mature) Solvita® CO_2 values (6-7) had the highest pH values (>8) and highest NH_3-N concentration as measured by the Thermo Scientific Orion ammonia probe. Stable Solvita® CO_2 results (6-8) should be interpreted with one or more of the following tests: Solvita® NH_3; CO_2 evolution; pH, or ammonia concentration; in order to reduce the risk of mis-interpreting stability in raw and inhibited end product.

The CO_2 evolution values presented, ranging here between 0.00195 and 3.18 mg CO_2/g OM/day would classify under TMECC 05.08-B all as mature to very mature (Wichuk and

McCartney) but these values are likely underestimated by the high fraction of bulking agent (acting as a diluting agent) in each sample (Table 3.2). End product likely contains more bulking agent than dry fecal matter (Hill and Baldwin 2012) and little of this high lignin content wood shavings may actually be available for utilization by microorganisms (Kayhanian and Tchabanoglous, 1992). If the CO_2 rate were calculated per fraction of fecal matter origin, where ammonia inhibition was removed, the rates evolved could be double or greater. CT samples could be sieved (0.5-1cm) to remove bulking agent prior to CO_2 evolution testing in order to make comparisons with accepted stability standards.

5.5. Conclusion

Both Solvita® NH_3 and CO_2 probes are useful tools in the analysis of compost toilet end-product quality. However, the probes should be used together so as to be able to identify the effects of ammonia inhibition on microbial respiration.

Solvita® NH_3 probes correlated significantly with ammonia measured in solution with a Thermo Thermo Scientific Orion ammonia probe. Samples having immature Solvita® NH_3 values 1-3, were found having high levels of total ammonium, high ratios of ammonium:nitrate, and low total nitrate, all of which are classified as immature by other standards.

Solvita® NH_3 correlated positively with *E.coli* and negatively with CO_2 evolution suggesting a possible causal relationship whereby high ammonia concentrations negatively affect bacterial pathogens and inhibit decomposition. This inhibitory effect was supported by unexpected positive and significant correlations between Solvita® CO_2 maturity and CO_2 evolution, pH, and ammonia concentration.

Solvita® CO_2 provides a measurement of in-situ microbial activity whereas CO_2 evolution provides a measurement of the actual degree of material stability or alternatively an estimate of potential respiration. Solvita® CO_2 values must be interpreted with Solvita® NH_3, ammonia measured by lab probe, pH, or CO_2 evolution in order to eliminate the risk of misinterpreting stability which may in reality be a lack of microbial activity caused by ammonia inhibition. In order to produce comparable results to published quality indices including CO_2 evolution rate and stability classification, CT end-product samples should be sieved prior to testing to remove the diluting effect of bulking agent on final results.

5.6. Tables

Table 5.1. Parameters tested by Benchmark Labs, Calgary, in the evaluation of compost end product stability, maturity, safety, and general quality.

Parameter	Test Name / Description	Units
Total Solids (TS)	APHA Method 3540B	%
Volatile Solids (VS)	APHA Method 2540	%
CO_2 Evolution	TMECC 05.08-B	mg-CO_2/g-OM/d
Ammonia-N NH_3-N	Cold-water-shake 1:2 sample:water, followed by measurement with a Thermo Scientific Orion high performance ammonia ion electrode at 20ºC according to manual instructions for free ammonia concentrations of >1ppm	mg/kg (ds)*
pH	Cold water shake 1:2 sample:water, followed by measurement with VWR symphony pH probe at 25ºC	na
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	CFU/g (ds)
Nitrate NO_3^-	APHA Method 4110A	mg/kg (ds)

* dried solid

Table 5.2. Correlation matrix of r^2 values, probabilities, and relation to alternate hypothesis (affirm or contradict). N.A. = not applicable.

		TS%	VS%	CO_2 Evolution	NH_3	pH	*E.coli*
Solvita® NH_3	r^2 p hypothesis	0.004 0.72	0.01 0.55	-0.52 <0.001 Not made	-0.57 <0.0001 Affirm	-0.71 <0.001 Affirm	+0.23 <0.01 Contradict
Solvita® CO_2	r^2 p hypothesis	+0.21 <0.01 Positive	0.05 0.21	+0.31 0.02 Contradict	+0.18 0.018 Not made	+0.26 <0.01 Contradict	0.05 0.25

Table 5.3. Mean, standard deviation, and range of "immature" compost samples as determined by Solvita® NH_3 (values 1, 2, and 3) compared to referenced standards for "immature" indices.

Parameter	Mean	Stdev	Range	Immature Rating	Reference
Solvita® NH_3	1.71	0.78	1-3	1-3	TMECC 05.08-E
Ammonia	1094	631	116.5-2523	na	None found
pH	8.3	0.5	7.2-8.8	No consensus	Na
Total ammonium	6577	1541	4762-10180	Immature when >300mg	ASCP 2001
Nitrate	7.17	14.54	0.1-48	Immature below 50mg/kg (ds)	ASCP 2001
Ratio ammonium: nitrate	49192	34751	141-102000	Immature when >3	TMECC 05.02-C

Table 5.4. Mean, standard deviation, and range of compost samples within Solvita® NH_3 values 4 & 5 compared to referenced maturity index ratings.

Parameter	Mean	Stdev	Range	Mature Rating	Reference
Solvita® NH_3	4.6	0.52	4-5	4-5	TMECC 05.08-E
Ammonia	113	200	4.3-663	na	None found
pH	7.29	0.53	6.53-8.11	No consensus	Na
Total ammonium	7273	5651	3306-17400	Mature when <300mg	ASCP 2001
Nitrate	3127	3979	0.1-9134	Mature when >50mg/kg (ds)	ASCP 2001
Ratio ammonium: nitrate	53500	72060	0.37-173600	Mature when <3	TMECC 05.02-C

5.7. Figures

Figure 5.1. Photograph of using Solvita® compost stability and maturity tests in the field.

Figure 5.2 Linear regressions between compost toilet end product samples measured with Solvita® NH_3 test paddles and measured with a (A)Thermo Scientific Orion probe, (B), pH (C) *E.coli*, and (D) CO_2 evolution. Significant regressions marked with *. 95% confidence curves fitted with dashed lines.

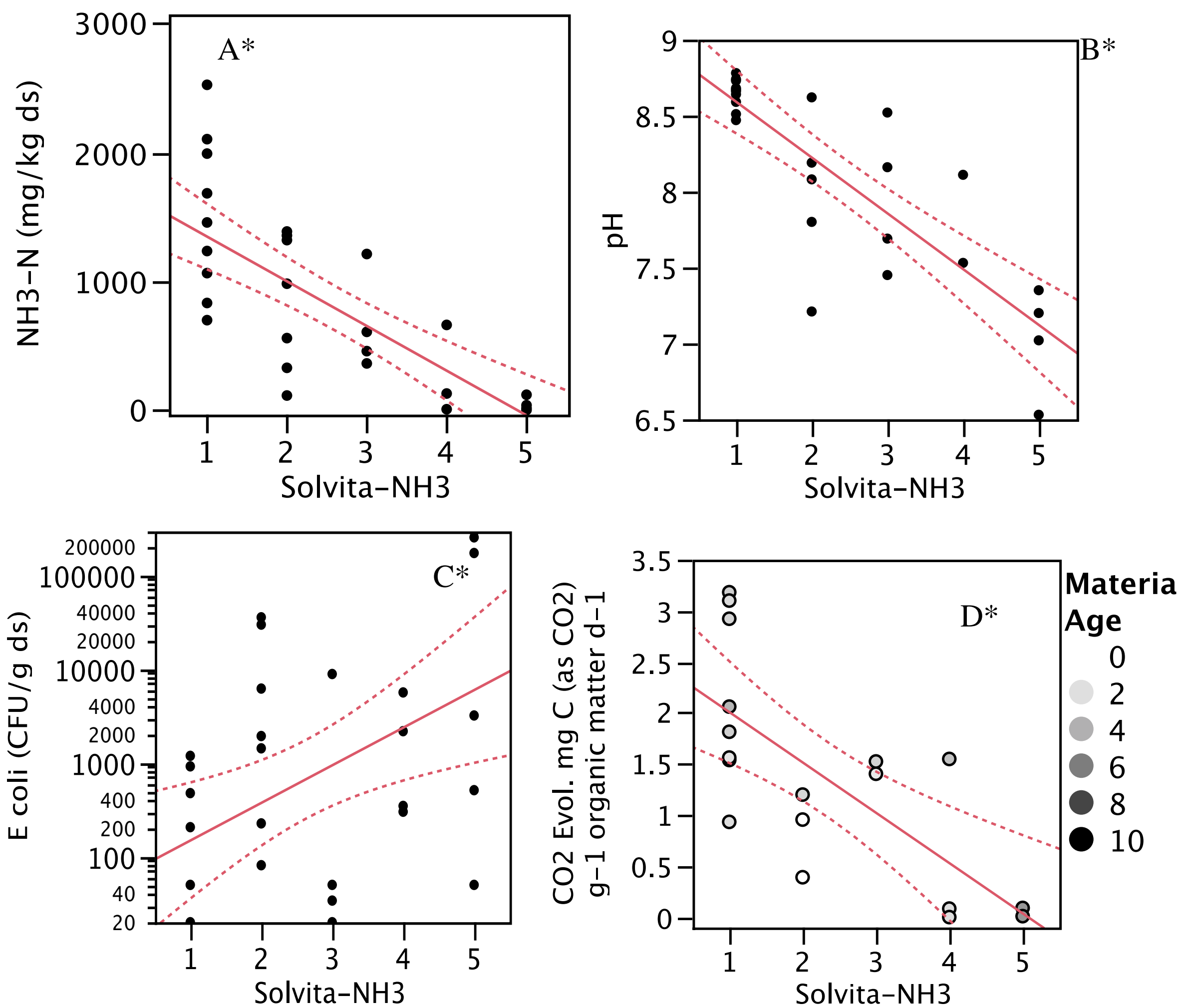
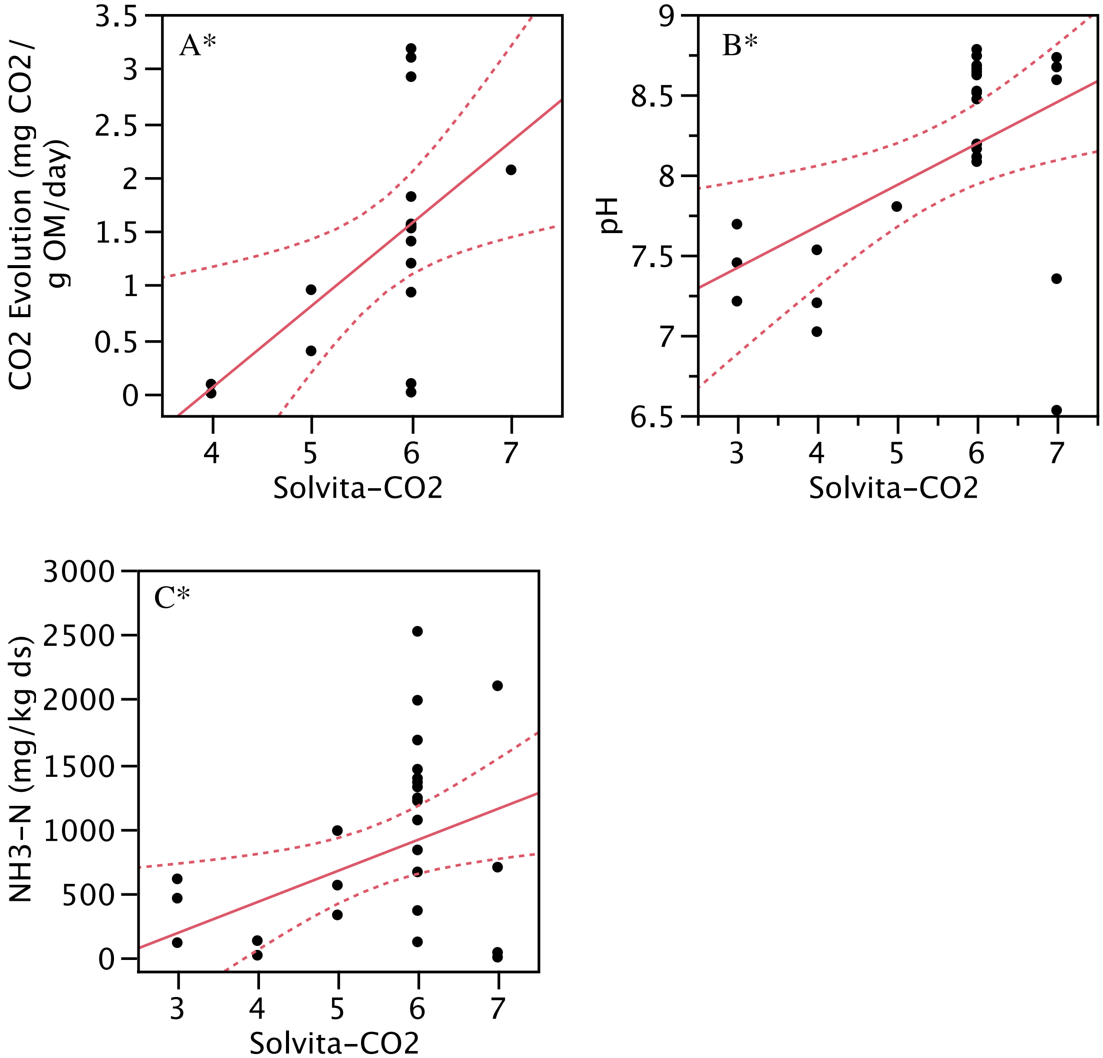


Figure 5.3. Linear regressions between compost toilet end product samples measured with Solvita® CO_2 test paddles and CO_2 evolution (A), pH (B), and ammonia (C). Significant correlations ($p<0.05$) marked with *. 95% confidence curves fitted with dashed lines.



Chapter 6. Vermicomposting effects on *Ascaris suum* ova

6.1. Abstract

Vermicomposting may emerge as a useful tool in the decentralized management, maturation, and stabilization of urine diverted human fecal matter. However, in order for the process to be solely relied upon for sanitation it must be proven capable of destroying helminthic parasites such as hookworm ova. To test the capability and resolve considerable debate on the topic, an experiment was conducted in which highly concentrated and viable *Ascaris suum* (2626±1306 ova/g, 61.6±8.7% viable) were inoculated into fecal matter mixed with coir (30:70 ratio) and placed into six 500 ml glass jars, three of which held 15 large *Eisenia fetida* worms. After 90 days at 19±3ºC six, eight, and 12 worms were found alive with no significant difference between treatments or through time found in: TS% (12-15%); ova concentration; and ova viability. A 2 $\log_{10}$ (100x) reduction in the concentration of *Escherichia coli* occurred between day 0 and day 90; however, no difference between the worm treatments and the controls was detected. Significantly higher nitrate and lower pH were found in the treatment (22735±4741 mg/kg NO_3^-)(pH 4.60±0.01) compared to the control (5078±2167 mg/kg NO_3^-)(pH 6.56±0.30). The extracellular processes accompanying vermicomposting, exemplified by high nitrate and low pH, combined with adequate opportunity for selective grazing were ineffective at reducing *Ascaris suum* ova concentration and viability. Unlike other decentralized toilet waste treatment systems, vermicomposting is highly effective in stabilization and maturation of fecal matter. However, an additional sanitation step is necessary if unrestricted use of end-products is desired.

6.2. Introduction

The ideal onsite human fecal matter waste treatment system would be a continuous flow one step process to produce stabilized, sanitized and mature end-products that can be handled without health risks and disposed onsite without environmental impacts, bringing about, at very low cost and risk, production of a soil amendment that could be used for rehabilitation projects (Jönsson et al., 2008). Results presented in Chapters 3 and 4 clearly indicate that CTs fail to deliver on all these claimed objectives.

Urine-diverting dehydration toilets (UDDTs) divert urine, evaporate moisture, and require ash amendment in order to reduce pathogens through desiccation and alkalization. Despite the appropriate focus on pathogen destruction over stabilization (Chapter 4), 31% of samples from Bolivian UDDTs that met WHO guidelines for UDDTs (high pH, low moisture, and long storage times), still contained viable *Ascaris lumbricoides* ova indicating that UDDT systems are not reliable sanitization systems (McKinley *et al*. 2012a).

Despite the ability of incinerating toilets to sanitize, stabilize, and mineralize human excrement, the soil improving benefits of the organic material are lost in the process. Furthermore a constant supply of fuel (usually propane), a trained operator, and enough units to ensure toilet availability during the inaccessible 1-4 hour burn period is required, making them inappropriate for applications in low-income countries and as remote public facilities, where the greatest need for sustainable, low cost, dry toilet solutions exists.

Although vermicomposting has not been approved by Canadian or US federal agencies as a pathogen reduction and stabilization pathway, it has been shown that vemicomposting (Eastman *et al*. 2001) and UDVCTs (Chapter 3) may have the capability to deliver on both of these objectives. By diverting urine the majority of nutrients in human excreta can be easily captured, utilized, cycled (Steinfeld 2007), and possibly denitrified. Separated fecal matter and toilet paper thus become a suitable feedstock for decentralized vermicomposting, a process that was shown, in Chapter 3 and by Hill and Baldwin

(2012), to produce sanitized (low *E. coli*), mature (Solvita® 4±0) and stable (VS 60±10%) solid end-product from human excrement, as has been shown similarly in numerous other studies including Bajsa *et al.* (2005)(sewage sludge),Yadav *et al.* (2010)(source separated fecal matter), Benitez *et al.* (1999) (sewage sludge), and others for numerous animal manures and industrial wastes as reviewed by Sinha *et al.* (2009) and Edwards *et al.* (2011). Vermicomposting toilet systems have, moreover, been shown to be safer, easier and less expensive to manage than traditional composting toilets (Chapter 3, Hill and Baldwin 2012).

Hookworm ova are one of the most resistant human pathogens commonly found in fecal matter (Eastman *et al.* 2001, Bowman *et al.* 2006, Jimenez-Cisneros and Maya-Rendon 2007). Hookworm ova shells are highly resistant to salts, chemicals, dessicaiton, acids, bases, oxidants and reductive agents (Jimenez-Cisneros and Maya-Rendon 2007). Bean *et al.* (2007) found no effect of pH adjustment to 12 for two to 72 hours had no effect on ova viability compared to controls at netural pH. Ammonia has been shown to have greater destructive potential than pH adjustment alone (Mendez *et al.* 2004) presumably due to the permeability of the ova shell to gases especially with increasing temperature (Jimenez-Cisneros and Maya-Rendon 2007). Temperatures greater than 40ºC (with residence times determined by temperature and process) are also utilized to destroy hookworm ova (Jimenez-Cisneros and Maya-Rendon 2007).

Despite hookworm ova defences, Eastman *et al.* (2001) reported that vermicomposting could reduce hookworm ova to below acceptable EPA limits. Bowman *et al.* (2006) found holes in the methodology, which cast doubt on the validity of the earlier results. Nevertheless, if vermicomposting were demonstrated to eradicate hookworms from human excrement, the process could be relied upon as the sole sanitization step in a decentralized treatment system of human waste in which the residual organic matter were to be reused for land application. Two laboratory experiments were designed and conducted to evaluate the effect of vermicomposting on hookworms using *Ascaris suum* ova spiked fecal matter. Additional variables were recorded along with *Ascaris suum* concentration and viability to correlate with degradation commonly seen with vermicomposting.

6.3. Methods

Fecal matter from two male volunteers was collected over the course of two weeks and stored at 4ºC for 2 weeks prior to initiating the experiments. Four kilograms of fecal matter was saturated with distilled water, drained over a 24 hour period, inoculated with three million *Ascaris suum* ova purchased from Excelsior Sentinel Inc. (Trumansburg, NY) and mixed into four kilograms of damp coconut coir, making a 50:50 fecal:coir wet weight feedstock. Coir was chosen as it is commonly used in vermiculture in Europe (Anbuselvi 2009). It has little nutritional value but considerable water holding capacity which makes it a great bulking agent for fecal matter, as was discovered in pre-experiment trials (Anbuselvi 2009). In addition to testing for *Ascaris suum* concentration and viability (method described in Appendix 5), the feedstock (before being added to the coir) was sampled for a variety of physicochemical variables to measure decomposition and maturity in end-products (Table 6.1). Feedstock and end-products were vacuum sieved with a 250 μm sieve and volatile solids (%) were calculated on both fractions. All analyses, unless otherwise stated, were conducted by Benchmarks Labs (Calgary, AB).

The feedstock was initially used in an experiment that was abandoned after 48 hours after the majority of the worms died. Dead earthworms were removed; the material was recombined, thoroughly mixed, and placed at 4ºC for 3 days while modifications to experimental design were made. Due to the initial failure, two different experimental setups were developed in order to introduce a degree of redundancy and to maximize the collection of valuable data. One of the two experiments was established by burying fabric sacs filled with fecal feedstock into moist coconut coir in a 20L plastic container. This coir provided a refuge for worms. Two types of bags were used, those with a weave enabling worm access (to test vermicomposting treatment effects) and those excluding worms (to act as the control). The material was to be sampled at 30, 60, and 90 days.

However, at the 60 day inspection it was found that all the bags had been compromised and worms had gained access to all feedstock in all bags.

The second experiment was of similar design to the first experiment, where 200g of the original feedstock was placed into six 500 mL glass jars upon a bedding of 150 g of damp coir, diluting it from its 50:50 fecal:coir ratio to 3:7. Fifteen mature *Eisenia* sp. worms, were added to three treatment jars creating a worm density of ~0.013 grams earthworm per gram material. The remaining three jars were left as controls without worms. Only three replicates were established because all the feedstock had been utilized and it was estimated that the chosen feedstock amount (200g) was the minimum required to sustain worms through the 90day processing period. Permeable geotextile was placed on the top of the jars and sealed with threaded rims. The jars were placed in a stratified pattern in a rectangular plastic container. A Hobo® air temperature and relative humidity logger were added to the plastic container and the container was placed into a covered, insulated, temperature regulated (18-22ºC) and humidified (80-90%) chamber. The cover of the chamber was not secured, leaving ample space for gas exchange. After 90 days the experiment jars were sent by overnight courier to Benchmark Labs for testing. Material was thoroughly mixed before sub-sampling. Upon completion of testing, Benchmark Labs shipped samples back to the authors for storage at 4ºC. After 15 days, samples were sent back to Benchmark Labs for total nitrogen and phosphorus testing. Coir was also sampled at this time for the full suite of tests. Remaining sample was kept in the fridge at 4ºC.

Vermicomposting was expected to produce greater amounts of nitrate; in order to verify this result we extracted genomic DNA from duplicate 10g wet weight samples using the MoBio® PowerSoil DNA extraction kit (MoBio Labs, Solana Beach, CA) according to the manufacturer's instructions three weeks after final sampling. Total nucleic acid concentration and DNA purity were measured using a NanoDrop® ND-2000 UV-Vis Spectrophotometer (NanoDrop Technologies, Wilmington, DE). Appropriate dilutions with sterile nucleotide-free water were carried out so that all DNA samples were the same concentration. Total bacteria 16S rDNA and the bacterial ammonia monooxygenase structural gene, *amoA*, were quantified using primers Bac27F 5' AGAGTTTGATCCTGGCTCAG 3' (Lane 1991); Bac519R 5' GNTTTACCGCGGCKGCTG 3'; and amoA-1F; 5'-GGGGTTTCTACTGGTGGT; amoA-2R; 5'-CCCCTCKGSAAAGCCTTCTTC (Rotthauwe *et al*. 1997), respectively. Quantitative polymerase chain reaction with SsoAdvanced™ SYBR® Green Supermix (Bio-Rad Labs Inc., Hercules, CA) and primers concentrations of 200nM was performed on a CFX Connect™ Real-Time PCR Detection System (Bio-Rad) with reaction conditions: 98ºC 2min; up to 40 cycles of 98ºC 5sec; annealing temperature of 55ºC (total bacteria) or 60ºC (AmoA) 30sec; with a final melting curve at 65-95ºC at 0.5ºC increments of 2sec. Reactions were performed in triplicate and all amplicons produced only one melting curve peak.

ANOVA tests were used to evaluate between treatments and controls after checking univariate assumptions, which were met in all cases.

6.4. Results and discussion

A temperature of 19±3ºC was maintained through the duration of the 90 day experiment which is within the tolerance for *Eisenia* sp. (0-35ºC with an optimum between 20-25˚C, Neuhauser *et al*. 1988). The relative humidity in the covered chamber averaged 95±6%, which assisted in the maintenance of soil moisture favored by earthworms without requiring frequent manual watering to replace water lost through cellular respiration and evaporation.

The starting worm density was 0.013 g-worm/g-material (15 worms, average 0.3 g/worm, in 350 g of wet material) is within the range of worm densities 0.005-0.05 g-worm/g-material found to successfully sanitize sewage sludge and fecal matter producing stable and mature end product in 8-9 weeks during experimental trials (Benitez *et al*. 1999, Yadav *et al.,* 2010). Worm count by the end of the 90 day experiment had dropped to 8.7±3.0 from 15 (all had >5 alive); nonetheless, the treatment effect of vermicomposting was considered sufficient as the sample with the lowest final worm density (0.007g/g) was still higher than that found sufficient in other studies (Benitez *et al*. 1999, Yadav *et al.,* 2010).

The cocoon density (0.069±0.087 cocoons/gram total feedstock (Table 6.2)) is in the middle of a considerable range of cocoon densities found in other experiments including Yadav *et al*. (2010) where 0.0026-0.0032 cocoons/g feedstock were found after 30 worms were placed into 40 kg feedstock for 4 months, and Sangwan *et al*. (2008) who found 0.42-1.36 cocoons/g feedstock (from 5 worms placed into 150 g feedstock in 1 L containers for 13 weeks). Adequate nutrition is required to support growth and reproduction in earthworms (Sangwan *et al*. 2008) and the cocoon density found here can be interpreted as an indicator of adequate nutrition and affirmation of the vermicomposting process in the treatment jars (Dominguez and Edwards 2011). Nevertheless, receiving no additional feedstock, the process was far from optimized; mature *Eisenia* sp. can produce 0.35 cocoons/worm/day (Dominguez and Edwards 2011), which would equate to a cocoon density of 1.4 cocoons/g material if the rate were maintained throughout the 90 day experiment.

The TS content of the feedstock, coir, treatment and control were all between 12-15%, which is within the optimum range (75-90% moisture content) for vermicomposting (Neuhauser *et al*. 1988) (Table 6.2). Dominguez and Edwards (1997) suggested the optimum moisture be between 80-90%. End-product from non-bulked commercial vermicomposting toilets - which sustained adequate worm density (0.03±0.04 g-worm/g-material), reduced *E.coli* to 200 CFU/g, and produced mature, stable, high density (850 kg/m^3) vermicompost - was considerably drier, a factor which is important in order to prevent anaerobic conditions (27±10% TS)(Chapter 3). However, the high lignin content (30%) and low density (70-80 kg/m^3) of the coconut coir bulking agent likely maintained oxygen transport preventing anaerobic conditions from developing (Anbuselvi 2009, Chapter 3). As desired, there was no significant difference in TS between treatment and control eliminating any effect or interaction moisture content may have had on decomposition or pathogen destruction (Table 6.2).

Vermicomposted material had a pH of 4.60±0.01 which was significantly less than the control (6.56±0.30)(p=0.0074) and lower than the feedstock (8.04±0.17), coir (5.88±0.12) and weighted average of the two (6.53±0.21) (Table 6.2). A wide range of vermicompost end-product pH values have been reported and generally related to feedstock characteristics, as noted in Table 6.3. *Eisenia* sp. are suggested to have a pH preference as low as 5.0 (Dominguez and Edwards 2011) and as high as 7.0 (Sinha *et al*. 2010). The decrease in pH was expected for both the control and even more so for the treatment: decomposition produces organic acids and CO_2, while nitrification (known to be amplified during vermicomposting) consumes hydroxide ions (Haug 1993, Anthonisen 1976, Parkin and Berry 1999).

When the feedstock and coir were mixed together the total (weighted) average volatile solids (VS) percent was 94.8±2.7%, which after 90 days was reduced to 91.7±0.35% in the vermicompost treatment and 92.2±0.5% in the control (Table 6.2). The treatment and control were not significantly different. However, when sieved through a 250μm screen, the fine fraction of starting material, treatment, and control VS values were all lower than the total VS values and as predicted, the vermicomposting VS was significantly lower than for the control (Table 6.2, p=0.0314) indicating accelerated decomposition in the fine fraction which is presumably the fraction that can be actually ingested by worms. Conversely, in the fraction >250μm the VS of the starting material, treatment and control were all higher than the total VS values, and the control almost had significantly less VS than the treatment (92.3±0.5 and 93.7±1.4), respectively p=0.0674)(Table 6.2). Large filamentous fungal growths were observed in the control jars of similar previous experiments (Figure 6.1)(results not shown). Fungi are one of the main agents in decomposing large complex lignin particles (Kirk and Farrel 1987) and may have degraded this fraction faster in the controls where there was less predation by worms (Dominguez 2011).

It is common in vermicomposting human fecal matter to see VS values decrease to 60-80% from starting values of 90% (Table 6.3), however, the significant results found here are likely to still have important functional meaning. The relatively small magnitude of

VS reduction was likely due to a small fraction of biodegradable carbon, and a large fraction containing lignin, the majority of which was not biodegradable, yet incinerated at 500ºC and dominated the VS result (Kayhanian and Tchobanoglous 1992). Chandler *et al*. (1980) proposed Equation 1, which can be used to calculate the biodegradable fraction. Coconut coir has very high lignin content (30%); and resulting decomposition can take many years (Anbuselvi 2009). Applying Chandler *et al*.'s (1980) equation to coir, the result is -1%; very little of coir's VS is thus biodegradable.

Equation 1. $B=0.830-(0.028)X$,

Where:

B= biodegradable fraction of the VS

X=% lignin content as % fraction of the VS

Coir was chosen as a bulking agent into which the *Ascaris* sp. inoculated fecal matter was placed in order to have worms preferentially consume fecal matter over bulking agent maximizing the ingestion of *Ascaris* sp. ova. The percent reduction in volatile solids as a result of vermicomposting can be estimated by dividing the recorded percent reduction in fine volatile solids between treatment and averaged starting material (4.1±7.8%) by the wet weight ratio of fecal matter to coir (0.3 (30:70)) to remove the influence of coir and by the fraction of material <250μm:>250μm (0.40±0.14) to include only the fine fraction of material. The result is 34±8% reduction in volatile solids, bringing the reduction into a similar range found in other vermicomposting studies (Table 6.3).

The original VS reduction through vermicomposting may be even larger; a considerable fraction of the VS may have been turned into CO_2 subsequently fixed into microbial mass by lithoautotrophic nitrifying bacteria as the nitrate content of the treatment material was very high (22735±4741 mg/kg NO_3^-); 10-500 times greater than previously reported for vermicomposts from human waste (1750-3750 mg/kg NO_3^-) (Table 6.3) or by commercial compost quality standards (>50 mg/kg NO_3^-N)(Wichuk and McCartney 2010).

The high level of nitrate found in the vermicomposted material was significantly greater than that of the control (5078±2167 mg/kg NO_3^-) ($p=0.0042$) both of which were greater than the starting material, which was below the detection limit (Table 6.2). Nitrate is produced by aerobic lithotrophic bacteria and archaea (Leininger *et al*. 2006) through the oxidation of ammonium (Haug 1993). This is the rate-limiting step in the nitrogen cycle due to the high oxygen demand (amongst other limiting factors) of these microbes (~4g O_2/g-NH_3-N), which can prevent complete conversion of all available nitrogen to nitrate. Comparing the total nitrogen (16700±4100 mg/kg) to nitrate (22735±4741 mg/kg NO_3^-) in the treatment it appears that complete nitrification took place whereas only ¼ of the TKN (19000±1500 mg/kg) was converted to nitrate (5078±2167 mg/kg) in the control. There is ample supporting evidence for significantly increased mineralization rates, high nitrate content, and nitrifying microbial mass associated with earthworm activity in burrows (Parkin and Berry 1999) and in vermicomposting operations (Subler *et al*. 1998, Dominguez 2011). There is also research indicating that coir is an excellent medium for nitrifying bacteria (Reghuvaran *et al*. 2012). Indeed, based on $\log_2\{-\text{Delta(Cq)}\}$ values from the qPCR experiment, the ratio of bacterial ammonium oxidizing DNA (*AmoA*) to total bacteria DNA (16S rDNA) was ~40 times larger in the vermicomposting treatment as compared to the control ($p < 0.01$) (Figure 6.1). Negligible amounts of archaeal ammonia oxidation *AmoA* were amplified using the primers from Francis *et al*. (2005). Vermicomposting human waste with coir may be a highly effective process for the production of nitrate fertilizer.

The small amounts of ammonium-N found in the treatment (66.8±25.1mg/kg) may have resulted from recent worm death and decay, which was not present in the controls. There was no significant difference in zinc or potassium between treatment and control and no trend compared to starting material (Table 6.2).

Our results indicate no difference in *Ascaris* sp. concentration ($p=0.56$) or viability ($p=0.50$) between treatment and control and no reduction in viability after 90 days compared to initial feedstock (Table 6.2). It appears as though concentrations of ova are greater in the treatment and control than in the feedstock, and while this may be due to loss of organic matter from decomposition, it was not a significant effect and the variability was likely caused by incomplete mixing during inoculation.

E.coli was reduced from a starting concentration of 6.1×10^4 CFU/g to <1000 CFU/g in both treatment and control but no differences were found between treatment and control and no conclusions can be drawn in regards to the efficacy of vermicomposting on bacterial pathogens. A considerable number of studies have been conducted on bacterial pathogen destruction resulting from vermicomposting, the vast majority of which indicate more rapid and complete bacterial pathogen destruction than a control lacking earthworms (Mitchell 1978, Brown and Mitchell 1981, Murry and Hinckley 1992, Finola *et al*. 1995, Eastman *et al*. 2001, Buzie-Fru 2010). We conducted a similar experiment, with similar feedstock, sampled at 30 days which showed *E. coli* in worm accessible material to be $1.1 \pm 0.7 \times 10^4$ CFU/g compared to $3.6 \pm 1.4 \times 10^4$ CFU/g (data not shown), lending weight to the likelihood that *E. coli* in the treatment of the 90 day experiment were eliminated more rapidly. More importantly, *E. coli* concentration had no correlation to *Ascaris* sp. viability indicating that *E. coli* elimination alone cannot be proof of sanitation through vermicomposting.

Postulated mechanisms for pathogen destruction by earthworms include: selective predation / consumption (Edwards and Bohlen 1996, Bohlen and Edwards 1995, Kumar and Shweta 2011); mechanical destruction through action of gizzard (Edwards and Subler 2011); microbial inhibition through humic and coelomic acids or other enzymes secreted within the digestive tract and extracellular and within gut (Edwards and Subler 2011); stimulation of microbial antagonists including the ones of the *Streptosporangium* and *Pseudomonas* genera (Kumar and Shweta 2011); and indirectly through stimulation of endemic or other microbial species which outcompete, antagonize, or otherwise destroy pathogens (Edwards and Subler 2011). Nevertheless, ova defences against destruction are formidable; thermophilic aerobic digestion at 48ºC for 30 days is required to achieve 78% efficiency in ova reduction from 4.5 to <1 ova/10g TS (Gantzer *et al*. 2001). Based on hookworm ova superior chemical defenses and resistance to aerobic processes, it seems unlikely that a single passage (12-20 hours residence time) through the gut of the earthworm (Wood 1995) would be adequate to digest the three layers of protective ova shell and accomplish a reduction in viability or ova concentration. It is possible that chitinase excreted within the earthworm gut (Wood 1995) could begin the degrading the ova's middle chitin based protein layer (Jimenez-Cisneros and Maya-Rendon 2007), which confers mechanical resistance and prevents passage of material into the cell, but it seems likely that multiple passes or considerable subsequent degradation outside the gut of the worm would be necessary to accomplish significant decrease in ova viability or complete destruction and decrease in concentration.

While is it reasonable to assume the majority of ingestible material had been processed at least once and more likely multiple times based on material consumption rates (0.5-1.0g/g-worm/day), the high lignin content of the coir and considerable fraction >250um make it not possible to assume all matter and ova in the jars was ingested. However, it is also unrealistic to rely on worms to consume the material completely enough to be assured 4 $\log_{10}$ reduction in hookworm ova and meet WHO (2006) guidelines for unrestricted use (10^4 ova/g reduced to 1 ova/g). Moreover, ingestion of ova by *Lumbricus terrestris* appears to have little effect on *Ascaris* sp. ova (Rysavy 1969, Hartenstein and Mitchell 1978). The lack of ova reduction despite a full range of biochemical effects brought about by vermicomposting casts doubt on the efficacy of ova pathogen destruction reported by Eastman *et al*. (2001).

The data presented here should help to clarify some debate on the topic of *Ascaris* sp. ova destruction through vermicomposting. Eastman *et al*. (2001) report significant helminth ova reduction from $8.26\text{x}10^5$ ova/4g (dw) to $9.33\pm1.45\text{x}10^3$ in 6 days through windrow vermicomposting with *Eisenia* sp. as compared to $2.16\pm0.18\ \text{x}10^5$ windrow composting without earthworms. However, as Bowman *et al*. (2006) point out, some aspects of Eastman *et al*.'s experimental design and analyses appeared to be flawed; Eastman *et al*. (2001) report inoculating $1\text{x}10^6$ ova into the windrow, estimated to be 531 kg of raw material (which should result in an average concentration of 2 ova/gram), yet contradictorily report sampling $2\text{x}10^5$ ova/gram at the start of the test. Eastman *et al*. (2001) also report adding 1:1.5 earthworm mass:biosolids mass; it seems unlikely that hundreds of kilograms of earthworms could be sourced, let alone transported and applied. Bowman *et al*. (2006) also found flaws with Cardosa and Ramirez (2002) who reported successful helminth ova destruction by vermicomposting to acceptable limits (<1 ova / 4 g end product) yet fail to mention that starting concentrations of ova were also below this limit.

Bowman *et al*. (2006) conducted similar experiments using *Ascaris* sp. ova spiked potting soil, treatments with worms, and controls without worms and found similar results: no reduction in *Ascaris sp*. concentration was found after 183 days vermicomposting and no reduction in ova viability between vermicomposting and a control after 7 days and less than a 1 log reduction after 6 months.

It is concluded that additional treatment is necessary to ensure the destruction of hookworm ova, as the vermicomposting process does not appear to have the capacity to do this. Post-treatment with urea and ash to elevate ammonia concentrations, as per Nordin *et al*. (2012b) or McKinley (2012b), should accomplish the desired sanitization step. While vermicomposting was shown here failing to accomplish complete sanitization, it can be relied upon as a low cost method to stabilize and mature fecal matter; processes which are both valuable in the practical aspects of waste management and essential in the development of compost suitable and beneficial when reused. Vermicomposing is also very useful for stabilization and increasing the nitrate content of the end product increasing its value as a fertilizer, if desired. More research is needed on chemical and physical post-treatment sanitization.

6.5. Conclusion

A vermicomposting treatment was successfully sustained with adequate worm densities in three small glass treatment chambers for 90 days as shown by the production of 0.07±0.09 cocoons/g. In comparison to a control without earthworms, the treatment induced expected, significant, and considerably increased nitrate, lower pH and significant yet minimal reduction in volatile solids due to low biodegradability of coconut coir used as a bulking agent. Despite reduction in *E.coli* to <1000 CFU/g (ds) from 60000 CFU/g (ds) in both treatment and control, no reduction in *Ascaris suum* egg concentration or viability was found between final and initial stages in either treatment. No significant difference between treatment and control was found in the concentration of *E.coli*, and concentration and viability of *Ascaris suum* after 90 days. While the entire mass of material may not have been ingested by worms, these results indicate that pathogen destruction through selective grazing or extracellular process associated with vermicomposting had no effect on *Ascaris suum* concentration or viability.

6.6. Tables

Table 6.1. Parameters tested and test names / description, used to evaluate vermicomposting effects on *Ascaris suum*. Parameters tested by Benchmark Labs (Calgary, AB) denoted with *.

Parameter	Test Name / Description	Units
Percent total solids (TS%)*	APHA Method 3540B	%
Percent volatile solids (VS%)*	APHA Method 2540	%
Percent volatile solids >250um	APHA Method 2540 with sieve fraction >250um	%
Percent volatile solids <250um	APHA Method 2540 with sieve fraction <250um	%
Ammonium-N (NH_4^+-N)*	Mod. ASTM D6919 (ion chromatography)	mg/kg (ds)
Free ammonia-N (NH_3-N)*	$[NH_3] = \frac{[NH_4]}{10^{9.3-pH}}$ (at 20ºC)	mg/kg (ds)
pH*	Cold water shake 1:2 sample:water, followed by measurement with VWR symphony pH probe at 25ºC	-
*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	CFU/g (ds)
Nitrate (NO_3^-)*	APHA Method 4110A	mg/kg (ds)
Ascaris sp. ova concentration*	As per Satomi *et al*. 2003 with minor modifications	#/g
Ascaris sp. ova Viability*	As per Satomi *et al*. 2003 with minor modifications	% viable
Zinc (Zn)*	Mod. EPA 3050A (Digestions), Mod. EPA 6020 (ICPMS)	mg/kg (ds)
Phosphorus (P)*	Mod. EPA 3050A (Digestions), Mod. EPA 200.7 (ICPOES)	mg/kg (ds)
Total Kjeldahl Nitrogen (TKN)*	Watson,M.*et al*. (2003). *Recommended Methods of Manure Analysis*. A3769: University of Wisconsin-extension	mg/kg (ds)

Table 6.2. Feedstock, coir, starting material, treatment, and control means and standard deviations for parameters tested in determination of effect of vermicomposting on *Ascaris suum*.

Parameter (units)	Feedstock (mean±SD)	Coir (mean±SD)	Starting Avg+ (mean±SD)	Treatment (mean±SD)	Control (mean±SD)	p value (T vs. C only)
TS (%)	14.53±0.17	14.70±0.46	14.65±0.49	12.38±1.34	13.23±0.59	0.49
VS (%)	93.22±2.18	95.32±1.55	94.80±2.70	91.69±0.35	92.21±0.50	0.209
VS >250um (%)	95.65±0.58	96.27±1.18	96.09±1.31	93.68±1.14	92.34±0.48	0.0674
VS <250um (%)	89.67±4.75	88.91±8.73	90.22±7.82	86.13±2.09	91.89±3.29	0.0314*
NH_4^+-N (mg/kg)	402.0±58.5	N.A.	N.A.	66.8±25.1	<2.5	0.011
NH_3-N (mg/kg)	24.3±8.2	N.A.	N.A.	<2.5	<2.5	N.A.
pH	8.04±0.17	5.88±0.12	6.53±0.21	4.60±0.01	6.56±0.30	0.0074*
NO_3^- (mg/kg)	<2.5	0.23±0.40	<2.5	22735±4741	5078±2167	0.0042*
NO_3^- N (mg/kg)	<2.5	<2.5	<2.5	5138±1071	1148±490	0.0042*
TKN (mg/kg)	34000±15400	9600±2200	16900±15600	16700±4100	19000±1500	0.51
E.coli (CFU/g)	61422±9042	0++	18426±9042	442±290	310±360	0.65
Ascaris ova (#/g)	2626±1306	0++	787.8±1306	6269±3226	4638±3095	0.56
Ascaris (% viable)	61.6±8.7	N.A.	61.6±8.7	52.7±8.4	57.4 ±7.2	0.50
Zn	112.5±12.6	0++	33.75±12.6	45.6±11.2	26.8±15.2	0.17
P	5833±1072	285.7±321.9	1949±1119	2554±889	3299±1527	0.51
Cocoons (#/g)	0	0	0	0.069±0.087	0	N.A.
Worms (#)	0	15	15	8.7±3.1	0	N.A.

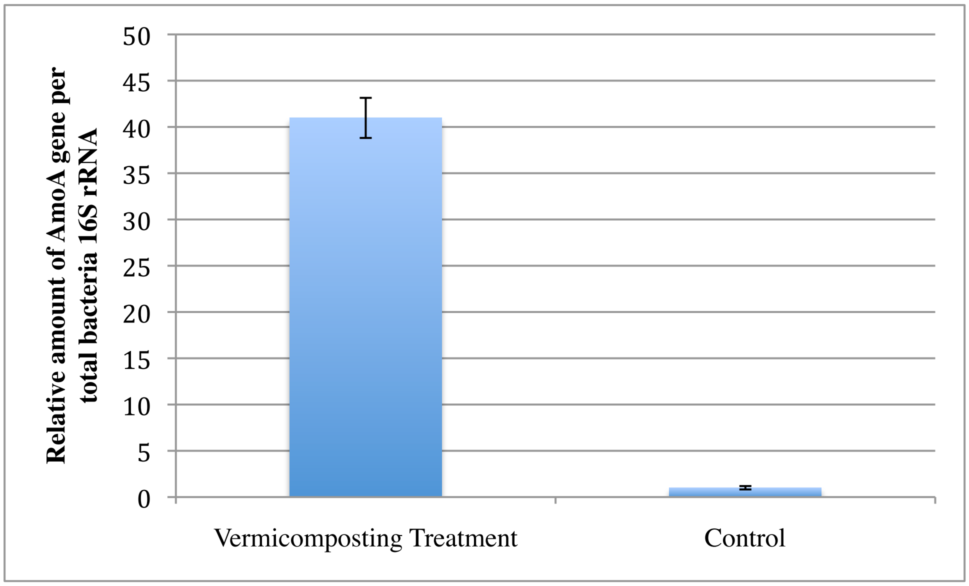
+: Weighted average by wet weight of coir and feedstock added to each container
++: Assumed, not measured.
N.A.: Not measured

Table 6.3. Comparison of studies on the effects of vermicomposting on pH, nitrate, total available nitrogen (TAN), and volatile solids (VS) or total organic carbon (TOC).

			pH		Nitrate-N (mg/kg ds)		Total Nitrogen (units)		Decreased VS or TOC
Source	Feedstock	Time	Initial	Final	Initial	Final	Initial	Final	
Bajsa *et al*. 2005	Sludge		9.5	4.5	500	3750			
Bajsa *et al*. 2005	Sludge and sawdust		8	4.5	2500	1750			
Hill and Baldwin (2012)	Source separated fecal matter	3±1 years		7.4±0.3		1961 ±700			Yes, 20-30%
Yadav *et al*. (2010)	Source separated fecal matter	4 mo	5.3±0.2	8.0±0.3			41 (g/kg)	28 (g/kg)	Yes, 48%VS
Kumar & Shweta 2011	Cow dung with other amendments	60 days					0.73-0.86%	1.0-1.4%	Yes, 2-4%
Sangwan	Mixtures of cow dung, biogas plant slurry and press mud	91 days	7.4-8	6.1-7.4			15-19 (g/kg)	21-27 (g/kg)	Yes, 7-12%.
This study	Source separated fecal matter and coir	90 days	6.5±0.2	4.6±0.0	<2.5	5138± 1071	17±16 (g/kg)	17±4 (g/kg)	

6.7 Figures

Figure 6.1. Relative amounts of AmoA gene per total bacteria 16sRNA between vermicomposting treatment and no-worm control (difference is significant between treatments, $p<0.01$). Error bars indicate standard deviation.



Chapter 7. Synthesis

The previous Chapters focused on in-depth performance analyses of the various water-less toilet waste management systems commonly utilized in remote sites in North America. Much erroneous information was uncovered and improvements tested. However, what remains to be addressed is the process by which remote site waste management systems are selected as determined by system: cost, capability, management objectives, climate, soil conditions, and site user characteristics and how these systems are constructed and managed in order to minimize operational cost, biohazardous exposure, and environmental impacts.

7.1. Remote site waste management system summary

Taking a global perspective on remote site waste management systems, a lack of appropriate technology can be found midway between low intensity management / high environmental impact systems (such as Pit Toilets (PTs)) and high intensity management / low environmental impact systems (such as Vault Toilets (VTs) and Barrel-Fly-Outs (BFOs)). Composting Toilets (CTs) attempted to fill this 'onsite' void, but the evidence presented in Chapter 3 and 4 indicates their failure to accomplish these goals. My thesis research was also focused on designing and testing components that could realistically fill this void whereby human waste could be more efficiently and effectively collected, treated, and managed onsite (at least temporarily, even if final end-products are to be exported offsite) without substantially impacting the local environment. All systems analyzed in this thesis fit between the two extremes of waste management: polluting PTs and total containment VTs (Table 7.1). In order to develop a useful reference for public land managers, these systems need to be further compared by cost, hazard, maintenance, and site considerations. See the glossary of acronyms for the various waste management systems investigated.

7.2. Cost per use, life cycle cost, and hazard comparison

Capital cost, operating cost, life cycle cost (LCC), and occupational hazards (biohazard exposure events) associated with system management will affect the selection of waste management systems by public agencies (Leffel, personal communications May 2012). Twenty year life cycle cost, cost per use, and exposure hazard for each system were estimated for a typical remote site by estimating inputs, treatments, outputs, and management costs for each system. A copy of this budget model is attached to this thesis in digital form on a CD. The main parameters are presented in Table 7.2. The model was run four times with 500, 1000, 2000, and 3000 overnight visitors per year which produce annual toilet uses of 1875, 3750, 7500, and 11250, respectively, covering the majority of the usage range experienced by remote sites in North America. The site visited in Chapter 3 which experienced >40,000 toilet visits is a very popular hot spring tourist attraction, being classified as a remote site due to its lack of road access, but in reality a short board walk makes it very accessible. Input parameters for the site, maintenance costs, excrement flows, and system capabilities were collected from manufacturer instruction manuals, data collected for this thesis, price quotations, and from personal communications with staff and operations managers (Leffel, Cieslak, Hansen, Trewitt, Weston, Bunge, Bianco personal communications 2011-2012) Table 7.2).

The majority of variables varied independently of time and systems primarily differed as a result of whether urine was diverted and drained, whether solids were abandoned or removed, and on the size of the collection vessel. The exception would have been CTs, which were supposed to degrade material with time, however, no reduction in volatile solids was found (Chapter 3) so this rate was assumed to be negligible. UDVCTs were successful in reducing volatile solids over time (Chapter 3), but the impact on end-product was of such little significance compared to the diversion or urine and absence of bulking agent added each use, that the degradation was merely adjusted by a small annual mass reduction factor.

Model outputs were verified in collaboration with park managers at comparable sites and referenced were possible. According to this model, PTs and a variants of pit toilet described in Appendix 4, whereby urine is diverted from feces which are collected in a sealed pit liner (DPTs) and whereby a pit liner and the introduction of rainwater into the pit can turn the pit into a septic tank (WPTs) (both of which require septic fields) have the lowest costs per use ($0.02-0.03/use depending on use intensity), followed by VTs and UDVCTs ($0.03-0.14/use depending on use intensity) (Figure 7.1). At 500 annual overnight visitors the BFO was most expensive at $1.06/use with UDBFO and CT only slightly less ($0.93-0.94/use) (Figure 7.1). At higher use (3000 overnights), the CT was the most expensive per use ($0.49/use) with BFO and UDBFO slightly less ($0.32-0.39/use)(Figure 7.1). Rollins (2012) report largely similar cost per use figures for an 800 overnight users site (PT $0.22/use, BFO $1.44/use, CT $0.98/use). Her elevated PT cost per use may be due to the inclusion of solar panels and fans which are not required and possibly mathematical errors, as the method is unclear.

Costs per use decrease with increasing usage, as fixed costs, such as travel to the site and helicopter travel time, are independent of use. In general, these results match with reality; PTs are the least expensive to operate, cheaper than CTs and VTs (B.C. Parks Facility Standards 2005) and one of the most common toilet systems in the world (Esrey *et al*. 1998 and Franceys *et al*. 1992). VTs and UDVCTs are competitive and this is likely why UDVCTs can be found in Europe in the types of locations where conventional VTs would be found in North America. BFOs are found only where no other options are viable, suggesting that the high cost per use is well known and unacceptable in the majority of sites. UDBFOs are only slightly less expensive than BFOs as only urine from urinals were diverted, which did not contribute much mass loss. BFO and UDBFO cost per use estimates may be considerably higher than actual costs incurred by agencies who require volunteers to manage barrel changes and combine waste collection trip for multiple sites as is done by the Alpine Club of Canada to minimize costs.

Life cycle costs (LCCs) increased with site usage due to positive relationship between use and maintenance costs. The LCCs for systems with high costs per use (such as CTs) can be many times greater at high use sites than low use sites. PTs had the lowest LCCs ($7000-12000) across all site usages due to their low capital cost and low cost per use. Rollins (2012) report a similar cost with a similar LCC accounting ($11000). However, as discussed in Chapter 7, PTs are potentially one of the most polluting toilet systems where seasonal ground water depth is close to the bottom of the pit or unmeasured. PTs are also not appropriate at high-use sites due to the necessity for frequent pit excavation and the eventual challenge in finding undisturbed land for excavation at the desired location (Catto, personal communications July 2011, Cieslak, personal communications September 2010).

The life cycle costs for DPTs and WPTs were slightly higher ($11000-15000) due to added capital costs of a septic field and culvert tank. For better estimates, more DPTs and WPTs need to be constructed and evaluated.

Urine diverting vermicomposting toilet LCCs were the least variable with usage ($34000-35000) due to infrequent maintenance (annual) and disposal (decadal). Composting toilet LCCs ranged from $58000-120000 due to high capital costs ($30000), high cost of maintenance labor, frequent maintenance, large fraction of product occupied by bulking agent, and frequent disposal (every 2 years), by helicopter. Rollins (2012) reports a much lower LCC for CTs ($47000) at an 800 overnight user site. This undervaluation is likely due to underestimated labor costs ($600/yr), volume and mass of resulting end-product (30g/use), and removal costs ($625/yr); annual CT O&M costs were estimated by the operator of a similar use site to be $1800/yr, which does not include disposal fees (Cieslak personal communications 2011).

Barrel-fly-out and UDBFO LCCs varied from $45000-87000 with site usage where UDBFO was $1500-12000 less than BFO at low and high usage. Rollins (2012) reported similar LCC ($70000) for a 800 overnight users site. Vault toilet LCCs ranged from $23000-35000. Despite their unsuitability for true remote sites, which would not have road access, VTs were included as a benchmark for technology that minimized both environmental contamination (unlike the PT) and handling costs (unlike BFOs and CTs) and are the USA National Park Service preferred front country system (Leffel personal communications, May 2012). UDVCTs are the closest to VTs in terms of life cycle costs.

Composting toilets had the greatest number of exposure hazards over a twenty year period (535-2410) due to the requirement for frequent maintenance of raw excrement. Barrel-fly-outs and UDBFOs had 100-1000 exposure events with UDBFOs having ~40% less exposures due to thicker excrement (less sloshing contact risk) and less frequent barrel changes. Urine diverting vermicomposting toilets had a consistent 32 operating exposures due to annual maintenance and decadal disposal. Pit toilets, DPTs, and WPTs had zero exposure events as operators are never in contact with raw excrement.

The effect of CT maintenance frequency was not explored in Chapter 4, but it was concluded that fundamental design flaws causing short-circuiting and mutually exclusive objectives of sanitation and degradation in mixed latrine CTs prevents the production of sanitized, stable, and mature end-product despite operators' best efforts to conduct maintenance as per instruction manuals. In short, no valid reason to conduct the expensive and risky operation of a CT could be found. Projects striving to demonstrate nutrient re-use while maintaining a functional public toilet system should focus on capturing and utilizing urine on-site as a fertilizer, as urine contains low pathogen levels, can flow with gravity from excretion to a final destination eliminating handling, and contains the vast majority of plant available nutrients in human excreta (Vinneras 2002).

Cost per use, LCC, and exposure results validate the common-place decision to operate front country CTs as VTs as was discovered by Metro Vancouver after purchasing a CT which rapidly proved incapable of delivering on its stated objectives (Schade personal communications 2012). Unfortunately, this option isn't valid for backcountry CTs and in many of these situations, operators are left shoveling and hauling excrement by labor intensive / expensive means (Swenson, Cieslak, Armand, Lechleitner personal communications and or personal observations with these operators 2010-2012). CTs are not considered viable for remote site waste management and thus not included in further analyses here.

7.3. Urine diversion

7.3.1. Urine treatment method

By diverting urine away from fecal matter it is possible to: gain better control of moisture content in remaining solid excrement (Hill and Henry 2012); prevent toxic ammonia conditions from developing and thus enable vermicomposting (Chapter 3, 4, 5); seal the container into which fecal matter collects preventing pathogen transfer to groundwater (Appendix 4); and if the urine can be treated or disposed onsite, the total mass of excrement transported for off-site for treatment can be greatly reduced.

Due to limited resources, only two methods of urine treatment can be realistically applied in remote locations: dispersion into shallow septic fields and/or evaporation. The conditions which affect the rate and reliability of evaporation depend on latitude, climate, weather, season, aspect, and site topography (shade effects). The reliability of a urine evaporation system will depend on the evaporative demand produced by these conditions and on the rate and timing of urine supply to the evaporation system. A ~$2m^3$ prototype urine evaporation system was able to evaporate 4.5 L of urine over a 24h, hot, dry, August 24^{th} 2012 in Calgary, Alberta. This equates to 26 urination uses and is equivalent to the urine production of 7 overnight visitors per day throughout the season (using the adjusted backcountry urination estimate 0.176 L (Chapter 2) and model estimates (Table 7.2). However, there are numerous practical challenges that must be overcome before urine evaporation becomes a reliable and low cost waste management tool including: storage of urine during low evaporative demand periods (night, poor weather, winter); supply of urine during high evaporative demand (daytime + summer); odor management (volatilized ammonia); pest management (mountain goats, rodents); simple and infrequent O&M; and structural resiliency against mountain weather.

Many soils have a high capacity to absorb wastewater and nutrients and the use of soils to accept and treat wastewater is commonplace in North America (Sherlock *et al*. 2002). Septic fields work well (Meile *et al*. 2010) and require very little O&M if designed properly according to actual waste water flows (CMHC 2012). Discharge of wastewater to soils is regulated locally through design standards, certified designers, certified installers, a permit review process, and annual monitoring in order to prevent ground water contamination by ensuring chemical oxygen demand, nutrients, and pathogens will be removed by natural soil processes (Sherlock *et al*. 2002, Snohomish Health District 2004, Meile *et al*. 2010). It is the objective of septic field treatment to oxidize organic compounds, immobilize nutrients, and subject human pathogens to a wide variety of biologically and chemically destructive processes (Meile *et al*. 2010). The capacity of septic field soils to accomplish these objectives decreases when saturated due to reduction in aerobic conditions and as a result saturated soils are generally not approved for septic field installation. However, through the use of curtain drains (also known as French drains) the frequency of annual saturation at 60cm (2') depth in one septic field study was reduced from 60% to 2% (Brown 2007), suggesting the possibility that even wet soils could be drained and used in the treatment of urine. If successfully prevented from leaching to subsoil or ground water, nutrients in urine will eventually be assimilated, and in nutrient limited ecosystems, this could lead to fertilization effects, loss of diversity, soil organic matter loss, or ammonia toxicity (Pysek and Leps 1991, Chapter 4). Nitrogen and phosphorus are the most limiting nutrients in terrestrial ecosystems, but nitrogen will be focused upon as phosphorus does not leach from soil as easily (Alberta and Rural Development 2001).

If the risk of local ecosystem impacts cannot be accepted, and the site conditions do not make urine evaporation viable, managers should not bother diverting urine and should collect combined excrement in containers convenient for removal (Hopkins, personal communications September 2012, Bethune, unpublished). These conditions exist in Maori heritage sites in New Zealand (Hopkins, personal communication 2011) and may exist elsewhere.

Despite eutrophication risks, discharge of urine to soil is more likely than urine evaporation to be adopted by public agencies in North America due to the ease and prevalence of septic field design, construction, maintenance, and reliability. The treatment of concentrated urine by soil in septic fields has not been well studied, however, the same processes by which residential septic fields work (Appendix 4) should be able to be relied upon for treatment of urine which despite being higher in concentration is lower in BOD and pathogens (Vinneras 2002). In the following section I outline design recommendations for implementation, monitoring and future research of urine-only septic fields.

7.3.2. Septic field sizing

Soils have varying septic effluent treatment capabilities due to the complex biochemical nature of soil texture, composition, density, and amount of organic matter. Soils are assigned effluent loading rate based on their average ability to retain and treat effluent (Indiana State Department of Health 2010). Septic field size is calculated by dividing daily maximum effluent flow by the septic field soils loading rate. Daily effluent flows are calculated per bedroom (Septic Design 2012). For example, a three bedroom house would require $84m^2$ ($900ft^2$) of septic field in very fine sand ($0.5g/d/ft^2$, $22l/d/m^2$).

Recognizing that urine is much more concentrated than residential wastewater and likely requires a larger area per volume to effectively, a starting point can be made by applying calculations used for conventional septic field sizing to the treatment of diverted urine form a 500 overnight person waterless urine-diveting toilet using the parameters described earlier for urine excreta rates. ,The septic field with very fine sand would need to be $0.15m^2$ (1.7sqft). With this size of field, the nitrogen fertilization rate in this small area would be approximately 2222kgN/ha/y (assuming 2gN per urination), which is quite high compared to the experimental rate of fertilization simulating cattle urination (200-800 kgN/ha/y) (Oenema *et al*. 1997, Orwin *et al*. 2010).

With high application of nitrogen in the form of urea there is considerable concern for ammonia inhibition of microbial activity (Chapter 4). The effects of fertilization on natural soil microbial activity are minimal at low concentrations (Williams *et al*. 2000, Nunan *et al*. 2006, and Rooney *et al*. 2006). However, Petersen *et al.,* (2004) and Orwin *et al*. (2010) reported nitrification delays and toxic build up of ammonia and nitrite at 430 kg-N/ha and 750 kg-N/ha respectively, in arable soil. Orwin *et al*. (2010) report ammonium concentrations in the soil after fertilization by livestock urine of 2000 mg/kg (ds), which raised the pH from 6.1, ~2-3 units to a pH of 8-9. The estimated free ammonia concentration at a pH of 8.5 at 20°C is 500 mg/kg(ds) (according to the equilibrium equation $[NH_3]=[NH_4]/10^{9.3\text{-}pH}$) which is at or above the toxic limit of: earthworms (Chapter 3); nitrifying bacteria (Chapter 3, 4); and may inhibit or be above the limit tolerable by other microorganisms and invertebrates required for healthy soil processes.

Negative effects on microbial activity are likely should standard design criteria be followed because of the high concentration of nutrients in urine as compared to dilute residential effluent. To minimize the risk of ammonia inhibition, urine could be discharged into a field ten times larger than required by standard calculations to bring the fertilization rate down to below toxic thresholds (2222kgN/ha/y reduced to 222kg-N/ha which is considerably below the lower limit of 430 kg-N/ha (Orwin *et al*. 2010)). The standard septic field daily maximum flow calculation could be adjusted by increasing the urine output by a factor of 10. Using the same example as above, the daily flow from a 500 annual overnight user site would require 1.5m^2 (17sqft) of sandy septic field soil. However, this design adjustment factor may need to be larger if future research discovered lower toxic limits for poorly developed soils found commonly in higher elevation backcountry areas. To estimate maximum daily urine inputs, the estimated daily maximum number of visitors could be multiplied by the average urine volume and adjusted by the proposed factor of 10 before determining soil area required. These suggestions are at best starting points. Ideally, lab and field experiments should be designed and conducted to evaluate the ability of various soils to nitrify, immobilize, and possibly denitrify concentrated urine.

The gravity fed design of the field will be important as the low flow volumes may take a preferential flow path that may not use the entire distribution field. More applied research is needed here in order to determine the best method of effluent splitting. Concentration may also occur if certain distribution ports become clogged with precipitates or organic matter, thereby reducing the size of the field. A changeable coarse filter slightly smaller in pore size than the distribution ports would reduce the risk of organic matter clogging. Precipitates can be flushed by directing rain water into the septic field. The slightly acidic and diluting solution should also aid diluting urine and more evenly distribute nutrients over the entire field.

Soil and soil water could be sampled after high visitation periods and sent to a laboratory for testing or tested onsite with Solvita® NH_3 test paddles to ensure ammonia is below toxic concentrations (386 mg/kg (ds) (Chapter 4), Solvita® NH_3 value 4 or 5 (and Chapter 5)). Paired catchment studies could also be conducted to evaluate the actual impact urine diversion has on downstream water quality by sampling background nitrate, nitrate found within soil and ground water after urine diversion, and nitrate in downstream water bodies.

7.3.3. Horizontal separation from surface water

In order to prevent minimize leachate entering water bodies and the risk associated with downstream eutrophication discharge fields are required to have minimum setback distances from surface water (15-75m) (CHMH 2012. More research is needed on the actual setback required for the low daily flux associated with urine only discharge. With regards to residential effluent, a horizontal setback of 60m achieved >80% nitrate removal in wet a marshland study of 111 residential septic systems (Meile *et al*. 2010). Enteric viruses are able to travel further and be infective at low doses (Moore *et al*. 2010) and the risk of transmission should be mitigated more closely. Where fecal pathogens are included in effluent flow (WPTs), the horizontal setback calculations outlined in Moore *et al*. (2010) should be conducted based on soil type. Setbacks required for adequate reduction in virus concentration can range from 70 to 2000m depending on soil type and thickness (Moore *et al*. (2010)

7.3.4. Fertilization effects

A very limited set of literature exists on the topic of urine fertilization in wilderness areas (Bridle and Kirkpatrick 2003). Rauch and Becker (2000) applied sludge from toilets at five mountain refuges between 2000 and 3000 m in Austrian Alps and document changes in localized biodiversity and increased plant growth. Localized but scattered urine fertilization (campgrounds without toilets) was found by Bridle and Kirkpatrick (2003) to increase native plant growth without effect on weed species and reduce moss at one of two alpine heath sites in Western Tasmania. Bridle and Kirkpatrick (2003) note in passing that exotic species are commonly found in areas of high urine concentration around backcountry hut doors, but no data were provided.

Studies on ecosystem responses to fertilization are more common than human or animal urine fertilization studies. Tundra ecosystems are one of the most nutrient limited terrestrial ecosystem and should be the most susceptible to fertilization impacts (Callaghan & Jonasson 1995). Alpine areas are also one of the most challenging environments for backcountry waste management due to their remoteness and lack of soil (Rauch and Becker 2000). Accordingly, focus will be placed on the response of alpine tundra communities to fertilization; the relevant literature is summarized in Table 7.2.

The literature summarized in Table 7.2 generally agrees that tundra responses to fertilization include: variation by functional group with graminoids, and grasses in particular, showing greatest increase in annual productivity due to phenotypic plasticity; increases in deciduous shrubs; no response or negative response in forbs presumably due to shading and or being out-competed for nutrients; no response or negative response for mosses; decrease or no change in species diversity or richness; and increases in litter and microbial biomass (measured less frequently) (Bowman *et al*. 1995, Richardson *et al*. 2002, Bret-Harte *et al*. 2004, Wang *et al*. 2010, Dormann & Woodin 2002). It should be noted that the experiments discussed here employed lower application rates (100-200 kg-N/ha) at or lower than the rate suggested earlier (200 kg-N/ha).

It should be noted that while these fertilization inputs are very concentrated in a small area they are minor compared to background anthropogenic rates. For example, 10 high use toilets (50 users per day for 4 months a year) in a 200 km^2 high use alpine recreation area in Colorado would constitute <0.5% of the total annual atmospheric additions, which are presumed anthropogenic (Bowman *et al*. 1995, Gardner *et al*. 2011).

Within tundra ecosystems, moist or wet meadows were found best able to immobilize and assimilate nutrients due to highly competitive, nitrogen limited, and plastic responses of moisture loving sedges, grasses, and deciduous shrubs and soil microbes as compared to less adaptable dry tundra species and (presumably) microbial populations (Bilbrough *et al.* 2000, Bret-Harte *et al.* 2004, Buckeridge 2009, Wang *et al.* 2010). Brooks *et al.* (1996) did not find leaching occurring as a result of atmospheric nitrogen additions to tundra (150 mgN m^{-2}) if sufficient soil was available, as competition for nutrients between microbes and plants is very strong. Nutrient uptake and turnover can occur within seconds to minutes during the growing season in boreal forest ecosystems (Jones and Murphy 2007), however, this needs to be ensured at the higher fertilization rates likely to result from urine diverted toilets.

7.3.5. Preventing leachate to groundwater

If effluent passes through septic field soil to ground water prior to pathogen removal and nutrient uptake, the risk of eutrohication (Sherlock *et al.* 2002, Meile *et al.* 2010) and pathogen transmission (particularly viral pathogens (Moore *et al.* 2010) increases. Leachate and eutrophication risks can be best minimized by following local septic design codes with the size calculation outlined above (or larger for more sensitive soils) to prevent ammonia toxicity.

In some locations soils may not meet septic design regulations. Under some circumstances the treatment of diverted urine by a shallow field may still be the best option. The situations outlined below would likely not work with high flux residential effluent and are only plausible because of the very small actual daily discharges associated with urine diversion even at high use sites (20l/day for a 3000 annual overnight user site).

- Where unacceptably high ground water occurs during the off-season and during the season of use the soil is predominantly non-saturated meeting septic design codes

- Where PTs are currently employed discharging nutrients and pathogens far below the surface. By diverting urine pathogen transmission prevention becomes possible (Chapter 7).
- Where vertical flux to deep soil ground water is greatly reduced or prevented by an impermeable barrier (bedrock or clay), keeping effluent in active soil, and where adequate setback from surface water is ensured. A natural lagoon or wetland may exist in these locations. Wetland vegetation is highly capable of utilizing nutrients in wastewater.
- Where vertical flux to ground water can be greatly reduced with an impermeable synthetic barrier, keeping effluent in active soil, and where adequate setback from surface water is ensured. Features of a constructed lagoon or wetland may develop in these locations. Constructed wetlands are an approved system for wastewater treatment; details are outlined by the EPA (2004).

In order to further minimize nutrient leachate immediately after construction while vegetation re-establishes, vegetation on the surface of the trench, into which urine will be diverted, could be carefully extracted and replaced after the installation of perforated pipe and fill.

7.3.6. Urine diversion components

Currently urinals and urine diversion seats are the only commercially available urine diversion components in North America. Urinals are reliable and commonly utilized in N.A., but only capture male urine and commonly have small pipes, which are prone to clogging with precipitates without flush water (Jönsson and Vinnerås 2007). Urine diversion seats may be appropriate in the private realm where users can be trained to use the technology but they proved unreliable in the public realm due to rapid clogging with toilet paper, fecal matter, and user applied bulking agent (composting toilets) (Chapter 2). A robust mechanism for the waterless diversion of urine, at the toilet seat, is lacking in the North American market. The urine diverting vermicomposting toilets studied in Chapter 3 are manufactured in France and are not available in North America except by individual unit importation, which was quoted at 5000€ (does not include shipping, duty, or taxes) (Neau, personal communications June 2012).

7.4. System selection decision charts

The appropriateness of each toilet system is determined by parameters which vary on a site by site basis (such as depth to seasonal high ground water). Few publically available toilet selection guides could be found (Yosemite National Park 1994). Rollins (2012) recently produced a decision tree for selecting waste treatment systems, however it is likely to be of limited practical use for the vast majority of public site operators for a variety of reasons. The first criteria is set according to site use where sites experiencing <90uses/yr, pack-out or PTs should be used. While these recommendations are reasonable, the daily visitation resulting from such a site with a four month summer window (late June, July, August, September, early October) as is the case with treeline or sub-alpine sites, is <1 person per day; compared to the annual visitation to back-country, these sites are likely to be very low on the list of priorities and unlikely to have budget for soil surveys or to introduce a pack-out program. Site with >90uses/yr are subdivided into custodian serviced or non-custodian serviced. The majority of backcountry sites in North America are not serviced with a custodian, but are regularly visited by staff for maintenance. The only recommendation for a non-custodian site was a BFO. Relatively few sites are serviced by this expensive method.

Other decision trees or decision matrices that were found were all out-of-date or erroneous. Synthesizing the results in this thesis, a simple toilet guide, suitable for most locations in North America, was developed to assist park operators choose the most appropriate, lowest operational cost, lowest risk, proven, and reliable waste management systems for their site (Figure 7.3). For example, a moderate usage, near-urban, road accessed park site in British Columbia could utilize a pit toilet but unless the seasonal high ground water level is known to be >600-1200 mm below the bottom of the pit and ample space was available for future pit excavations, it would likely be much easier, cheaper and safer in the long run to install a vault toilet. In this case, if the agency desires to demonstrate local nutrient recovery or onsite treatment, urine diversion can be added to the VT (similar to a DPT or WPT) or a UDVCT system could be chosen. As defined earlier, low use sites will provide service for less than 500 overnight visitors per year (Figure 7.3). Each step of the decision making tool are explained below. (Figure 7.3-1 to Figure 7.3-7).

In the vast majority of cases, where sewers are present, they should be used to transport fecal matter to centralized treatment systems to minimize risks associated with fecal matter handling (Figure 7.3-1). In order to demonstrate nutrient reuse, urine diversion can be incorporated very simply and safely by providing low flow or waterless urinals in the men's toilet room which are plumbed directly to gravity fed septic field, or to a tank for pre-treatment prior to onsite discharge or for storage and eventual collection and offsite distribution as is done in Sweden (Steinfield 2007, Jönsson and Vinnerås 2007).

If the need for a toilet is temporary and road access is present, the least expensive and lowest risk system is a portable toilet rented and serviced by a septic contractor (Figure 7.3-2).

In the vast majority of cases, where road access is present and a septic trucking company can be contracted to haul waste, the VT is likely to be the safest choice (Figure 7.3-3A). Where soil conditions permit septic field construction, urine could be diverted from urinals and/or by an under-the-seat mechanical urine diversion system (Figure 7.3-3B). If haulage fees were high due to remoteness or lack of local septic truck contractor, and if UDVCT were commercially available and similar in capital cost as estimated in Table 7.1, they may be a viable alternative to VTs as they have lower cost per use at higher use sites and are less likely to have offensive odors and can be less offensive to service (Figure 7.3-3C).

At very cold, high elevation, temperate remote sites (>2400 m) the BFO system is likely to be the most appropriate due to the simplicity and robustness of the collection system (Figure 7.3-4A). At elevations below 2400 m, UDVCTs are likely to incur much lower costs per use and occupational hazards (Figure 7.1, Figure 7.2). Invertebrate decomposition in UDVCTs appears to need at least three months of temperatures >10ºC for successful decomposition, which, in the French Alps, occurs at elevations <2400 m (Neau, personal communications 2012). Urine has also been successfully diverted and discharged into subsurface soil overwinter at 2200 m (Perrier, personal communications 2012). If soil depth allows, urine can be diverted from urinals or through a mechanical under-the-seat urine diversion system (Figure 7.3-4B). If haulage fees are high and usage is low, a suitable sized pit can be excavated, and if adequate soil exists for discharge, a UDPT could be considered (Figure 7.3-4C). If park access points are few and tightly regulated by staff, pack out systems should be considered for the entire park, which can drastically reduce O&M costs and hazards (Robinson 2010) (Figure 7.3-4D).

At very low use backcountry sites where sufficient soil allows pit excavation, the PT is the least expensive system requiring no direct contact with fecal matter (Figure 7.3-5A). If inadequate soil depth to seasonal high ground water exists, a UDPT should be considered to prevent environmental contamination (Figure 7.3-5B). If no soil exists or surface flooding occurs, the BFO should be considered as a last resort. A monitoring system must be instituted to prevent barrels from overflowing and introducing pathogen transmission risk and management hazards during clean-up. If park access points are few and tightly regulated by staff, pack out systems should be considered for the entire park, which can drastically reduce O&M costs and hazards (Robinson 2010) (Figure 7.3-5D).

At sites with very hot and dry climates, UDDTs may be the most appropriate (Figure 7.3-6A). Due to the small fraction of desert or hot, dry, summer use alpine areas (Arnold 2010) in North America, UDDTs were not explored beyond the unsatisfactory results found through solar and 110V dehydration in an alpine setting in Chapter 2.

Sites which are not sewered, have no (or limited) road access, are neither high alpine (>2400 m) nor hot desert, moderate to high use, and have adequate soil for urine diversion, would be most economically serviced by the UDVCT (Figure 7.3-7). These conditions are likely to reflect the majority of backcountry sites in North America. UDVCTs exist in the full range of French climates from Mediterranean to alpine, service both front country city parks, trailheads, and highway rest stops and backcountry huts, lodges, ski hills, and campgrounds (Ecosphere Technologies 2012). The wide applicability of UDVCTs results largely from the design and integration of a urine diversion mechanism which uses gravity to divert urine (the major fraction of remote site waste), minimize fecal contamination (disease transmission), and treat effluent by means of well proven and reliable septic field technology whilst simultaneously preventing toxic contamination of fecal matter by ammonia, keeping it in the 'food' category for a host of detriovores, principally earthworms. The environmental conditions optimal for detritivory and vermicomposting (cool and damp) are naturally found slightly below ground level (exactly the conditions into which UDVCTs are installed) in the majority of locations in the northern hemisphere. Even without all the empirical evidence presented here, the matching of UDVCT process optimums with the ambient environment, as compared to reliance on high temperatures to sanitize and compost mixed latrine waste in CTs, should be enough to stimulate product development and market interest. A decrease in capital cost and increase in market uptake would likely occur if the UDVCT were produced in North America making these even more competitive with VTs and PTs.

The Solvita® test kit could be a useful too in the establishment of vermicomposting feedstocks and assessment of end-product stability. One of the most important aspects of vermicomposting toilet systems is the diversion of urine away from fecal matter to ensure low ammonia concentrations. Solvita® NH_3 test paddles could be used to during the initial stages of toilet operation to ensure sufficient urine diversion from the solid excrement (fecal matter and toilet paper). Solvita® NH_3 values four and five (low and very low) should be sought prior to introducing invertebrate decomposers such as earthworms which are sensitive to higher levels of ammonia. Residual matter stored in vermicomposting toilets will eventually need to be removed for final off-site treatment (except in extreme circumstances described below). However, in order to ensure material extracted is completely vermicomposted and the mass and volume have been minimized, the Solvita® CO_2 test could be utilized; values between seven and eight are correlated with finished compost (Woods End 2000).

7.5. End-product management conclusions

In order to treat the end-products of various waste collections systems to standards acceptable in the USA and Canada for unrestricted use of residuals in public park environments, an approved treatment process must be applied which has proven reliable in elimination of pathogens and stabilization of organic matter. Approved processes by which residuals at remote sites could be theoretically treated include microbial composting and pH adjustment (EPA 1999, B.C. 2007).

Despite considerable recent literature (Sinha *et al*. 2009) vermicomposting is not an approved method of waste treatment or compost production in either Canada or the United States. Chapter 3 documented the benefits of material stabilization, mass reduction, and nutrient mineralization. However, in Chapter 6, contrary to literature on the topic, results indicated the inability of vermicomposting to destroy or render *Ascaris suum* ova non-viable, lending support to the exclusion of vermicomposting from the list of approved processes in producing a residual waste stream that can be legally disposed into a public park environment. Nevertheless, application of vermicomposting following urine diversion appears to be the most effective (Table 7.3), lowest LCC cost (Table 7.5), proven (DPT and WPTs are not produced yet), and low-pollution system for onsite fecal waste reduction (PTs are not considered comparable due to the high ground water pollution risk). On the infrequent occasions that the chamber in which the process is conducted must be entered for annual maintenance or decadal end-product extraction offensive odors are unlikely to be encountered and end products are likely to be non-offensive to handle making the task of material removal an insignificant aspect of annual management. UDVCTs designs should guide the development of remote site waste management systems by inspiring similar focus on: employment of a reliable mass or volume reduction process; limited handling requirement or operator involvement; ambient thermodynamics parameters; long storage times; elimination of short circuit contamination of end-product; and ease of access for maintenance and removal.

Adequate temperatures do not spontaneously develop in-situ within composting chambers due to inadequate thermodynamics (Chapter 3, Chapter 4, Guardabassi *et al*. 2003, WHO 2006) and the attainment of these temperatures is likely to require external heat, which will add expense, or centralization of material (Chapter 3, Chapter 4, WHO 2006).

Adjustment of pH is attained by homogenizing lime into sludge ensuring a pH of 12 is held for four hours (Bean *et al*. 2007); a small scale reactor for this process could not be found and would likely be an expensive piece of equipment that would be difficult to power, operate, clean, and service at remote sites due to lack of running water and power. Further, pH adjustment has been shown inadequate in complete sanitation, failing to eliminate hookworm ova (Bean *et al*. 2007).

The application of urea (increased ammonia) and ash (increased pH) to fecal matter has been found more reliable in the decentralized sanitation of human feces than the increase of pH alone (Mendez *et al*. 2004, Nordin *et al*. 2009a,b). However, in order to ensure reliable sanitation, material will need to be stored isolated from fresh feedstock contamination, after post-treatment, for a period of time determined by site temperatures. There may be foul odors associated with this step that would need to be mitigated. More research is needed in this field and on this ambient temperature chemical post-treatment sanitation process, particularly on the dosages of urea and ash, residence time necessary for suitable sanitation (based on temperature), resulting nutrient concentrations, equipment requirements and design, and fertilization effects upon discharge. If this post-process were found reliable in pathogen reduction and were included as an approved process nationally, it could be employed to ensure the sanitation of highly stabilized and mineralized UDVCT end-product, which would be likely to result in a final end-product meeting all the biochemical conditions of true compost: sanitized, stabilized, and mineralized.

Post-processing by sieving to remove foreign objects will be important in order to avoid trash and sanitary products in final discharged material. Should this post-treatment processes prove viable, reliable, and low risk, it is unlikely that these steps will be cost effective management strategies for the vast majority of sites in North America due to the time consuming nature of carrying out and documenting the procedures, except in extremely remote sites in the Arctic where the transport of residual solids is prohibitively expensive or not permitted.

An exception for onsite discharge of untreated solid excrement is made for pit toilets. Regulation of pit depth and separation distances from seasonal high ground water limit contamination and eutrophication risks in some jurisdictions, but in essence, each pit becomes a small landfill for raw fecal matter. It is not the topic of this thesis to debate the appropriateness of placing landfills into remote backcountry parks and protected area, but the banning of new PT construction by Alberta Tourism Parks and Recreation may indicate a long term trend whereby PTs are gradually phased out, being deemed inappropriate practices for wilderness waste management (Cieslak, personal communications 2011). DPTs and WPTs may fill an interim role in preventing direct contamination of ground water, but their contents would still be abandoned in the earth. An eventual shift away from abandoning excrement in the earth could bring increased demand, research, and funding for remote site toilet systems and further expand upon and develop the components, systems, and strategies for human waste management covered in this thesis.

7.6. Outlook

This thesis should stimulate a considerable amount of additional research on the treatment of human waste. Both urine-diversion and vermicomposting are scalable processes and have benefits that could span a variety of scales of application. The benefits are likely to increase with scale due to the costs associated with denitrification at centralized waste water treatment plants and costs associated with biosolids treatment, transport, and disposal.

The experiment conducted to evaluate the effect of vermicomposting on hookworm ova could be repeated and expanded to include: a larger sample size to reduce error and obtain tighter confidence limits; use of finer grain material, which could be entirely consumable by earthworms ensuring >99% of material (and ova) pass through the worms; and a wider variety of pathogens such as viruses (using bacteriophages perhaps) and protozoa.

Vermicomposting as a managed process for degradation of waste generally employs epigeic earthworms, most commonly *Eisenia fetida*. Other worms used in vermicomposting are reviewed by Dominguez and Edwards (2011). The majority of research has focused on epigeic (surface dweller) species because of a set of shared characteristics that result in rapid degradation of organic matter (and thus minimize operating costs (faster rate and less area required)). However, the main vermicomposting species all originate in Europe or Africa and this exotic status has raised concern with some parks in North America including the US National Park Service. The affinity for consuming source-separated fecal matter is shared by species in phyla other than annelid and are more likely to be found native to North America given the lack of epigeic worms here. Should local species be found having affinity for source-separated human feces, yet possessing lower processing capacity, waste treatment systems could be enlarged in order to increase the storage capacity as space is generally not a constraint at remote sites. More work needs to be conducted to determine which invertebrates have the highest affinity for feces and to explore the use of these populations regionally.

The design and production of a urine-diversion system, which can serve in the public realm, is essential for the toilet systems proposed here. Ecosphere Technologies in France is not willing to sell their urine-diversion device separately from their toilet structures (Neau, personal communications). This is unfortunate, as this utility has been proven highly effective and reliable (Chapter 3) and could be well received in the North American market. Until such time, urine from urinals can be diverted for cost reduction from vaults and fly-outs.

The use of septic fields to treat household effluent including high pathogen content effluent is common place. While similar septic-like fields are used to treat the effluent from UDVCTs in France, and are proposed here (Appendix 4) to treat diverted urine, there is much that could be done to optimize the treatment of this highly concentrated effluent. Much research could be conducted. It will be important to assess the ability of various soils and field designs to nitrify and potentially denitrify nitrogen in the presence of other components in urine including salts. It is suspected that periodic flushing of the field by rainwater would be an important method of diluting salts. This same flushing of salts could however, leach accumulated nitrates into water tables. Accordingly it may be wise to incorporate denitrifying environments into the septic-like field design. This could be accomplished by lining the trench with an impermeable liner upon which various layers of material could be placed. These various layers would encourage the growth and proliferation of aerobic lithotrophic nitrifying bacteria near the surface and anaerobic denitrifying bacteria near the bottom of the trench. The aerobic lay could consist of low density, high lignin, wood shavings which were shown capable of producing nitrate in CTs where ammonia was <386mg/kg (ds) (Chapter 4) in order to proceed with rapid urea hydrolysis to nitrite and nitrate. The bottom layer could be comprised of high density yet high surface area mineral such as clay or an equivalent synthetic material to promote anaerobic conditions where nitrate can be reduced to nitrogen gas and eliminated from the terrestrial system.

The performance of urine diversion fields would be best evaluated in a controlled setting in order to test the limits of their capability safely and easily. Nitrogen isotope tracer techniques and a nitrogen balance model would be possible in this controlled setting. However, where appropriate, especially where pit toilets or leach fields already exist, additional value would come in true field monitoring to understand the influence of uncontrollable variables such as temperature and climate on performance variability. While tracer studies and a nitrogen budget approach could also be used in the field the accuracy would be reduced due to estimations of use. In the field scenario, it may be more prudent to utilize a paired catchment/watershed method especially where before/after data can be collected on both catchments.

It is the expected that nutrients in urine can be largely integrated into the terrestrial ecosystem or denitrified into gaseous forms of nitrogen, by a variety of soil types found in remote North American sites, without leaching to down-stream water bodies (despite the small relative impact compared to other uncontrollable anthropogenic nitrogen additions). If this proves to be the case with the additional research recommended here, it is expected that the proposed strategy of hybrid onsite / offsite treatment, where urine is treated onsite by soil and feces are treated onsite by earthworms but where residual and trash are eventually removed offsite for disposal to landfill, will become the standard operating paradigm for remote site human waste management in North America.

7.7. Tables

Table 7.1. Summary of remote site waste treatment systems by name (Glossary), management intensity (intensity), liquid and solid treatment location (onsite or offsite), means of liquid and solid treatment, generalized risk of pathogen (P Risk) and nutrient transmission (N Risk) to environment / visitors directly or through drinking water (High, Mod(Moderate) and low), and key criteria which strongly affect the selection, suitability, or efficacy of each treatment systems.

System	Intensity	Liquid treat location	Liquid treat means	Solids treat location	Solids treat means	P Risk	N Risk	Key Criteria
None	Low	Onsite	Variable	Onsite	Variable	High	High	Low visitor density
Pack-out	Low	Onsite	Surface soil	Offsite	na	Low	Low	Few park access points, tight access control, compliance
PT	Low	Onsite	None proven	Onsite	None proven	High	High	Deep soil, low use, large area for many future pits.
DPT	Low	Onsite	Septic field*	Onsite	Long-term isolation	Low	Mod	Availability of an UD system, septic field soil during period of use, ≥60m+ separation distance from water
WPT	Low	Onsite	Septic field*	Onsite	Septic field + long-term isolation	Mod	Mod	Unsaturated surface soil during season of use, periodic rainfall, ≥60m+ separation to water
CT	High	Onsite	Septic field*	Onsite	None proven	High	Mod	Not recommended (Chapter 3 and 4)
UDVCT	Low	Onsite	Septic field*	Onsite + offsite	Vermicomposting + long-term storage + offsite	Low	Mod	Reasonable septic field soil during period of use, ≥60m separation to water
BFO	High	Offsite	na	Offsite	na	Low	Low	Limited soil, low use
UDBFO	High	Onsite	Septic field*	Offsite	na	Low	High**	Limited soil with natural barrier to groundwater / reasonable septic field soil in period of use, high use, ≥60m+ separation from water
VT	Mod	Offsite	na	Offsite	Na	Low	Low	Road and septic truck access

* or urine evaporation where climactically feasible and should a commercially viable system become available

** BFOs are more likely to be found in alpine areas where soils are thinner and are more susceptibility to leaching nutrients, extra care should be taken when designing septic field in such soil conditions to ensure sufficient separation distance from surface water and limited capacity to infiltrate directly to groundwater (ideally by a natural barrier such as bedrock).

+ depends on soil conditions and threat of pathogens (primarily associated with fecal matter or blackwater as urine has few pathogens at low concentration). Moore *et al.* (2010) should be consulted when designing septic fields based on local conditions and risks to mitigate.

Table 7.2. Life cycle cost analysis input parameters for the 500 overnight visitor model iteration, units, and references (where available). Where no units are provided the parameter description was deemed sufficient. Where no references are provided, the number was estimated or calculated from earlier estimates.

Site Parameters	Input	Units	References
uses / overnight visitor	3		
uses / day visitor	0.75		
# Overnight visitors/year	500		
# Toilet uses from overnight visitors	1500		
Overnight users/day (year round)	1		
Over night users/day (half season)	3		
Day visitors / year	500		
# Toilet uses from day visitors	375		
Total toilet uses/year	1875		
hrs drive to site	0.5	hrs	
hrs hike to site	0.75	hrs	
kms to site from operations facility	50	kms	
Labor Parameters			
Labor rate	100	$/h	http://www.toronto.ca/invest-in-toronto/labour_costs.htm
Helicopter rate	2500	$/h	Alpine Club of Canada
Septic truck rate	250	$/h	Canmore Septic
Septic hours / barrel	0.15	hours/barrel	Alpine Club of Canada
Septic hours / barrel (thick)	0.3	hours / barrel	Alpine Club of Canada
Septic hours / vault	4	hours/vault	Canmore Septic
Sewage disposal cost	0.00003	$/g	Canmore Septic
Travel mileage expense	0.4	$/km	http://garmin-reports.com
Helicopter flight time / trip	0.15	hrs/trip	
Helicopter max load / trip	600000	g	Alpine Club of Canada
Maximum barrel mass	200000	g	
Hours per barrel swap	0.1	hours/barrel	Personal observation
Travel time per barrel	0.15	hrs/barrel	Alpine Club of Canada
Annual disposal labor	16	hrs/site	2 people 8hr day
Cost to excavate pit and move building	1333	$/pit change	Trewitt personal communications July 2011

Labor Parameters	Input	Units	References
# exposure / barrel manage	1	#	Personal observation
# exposure / barrel fly	2	#	Personal observation
# exposure / compost maintenance task (raking & trash)	2	#	Personal observation
# expsoures / hour (disposal)	1	#	
Excrement parameters			
Wet fecal matter per defecation	250	g (ws) / event	See Chapter 2
Dry fecal matter per defecation	50	g (ds) / event	Schouw *et al*. 2002
Urine per urination	200	g (ws) / event	Table 2.1
Dry matter per urination	2	g (ds) / event	Vinneras 2002
fecal TS%	0.2	fraction	
urine TS%	0.01	fraction	
feces VS%	0.85	fraction	
urine/use	1.00	fraction	
urine/use (wet/use)	200	g (ws) / use	Median from Table 2.1
fecal wet/use (g wet/use)	60	g (ws) / use	Median from Table 2.1
TP (g ds/use)	10	g / use	20 sheets
fecal ds/use (g ds/use)	12	g (ds) / use	
density fecal	1	g (ws) / ml	Data not shown

Table 7.3. Model outputs, units, and references (where available) for the 500 overnight visitor iteration. Where no units are provided the parameter description was deemed sufficient. Where no references are provided, the number was estimated or calculated from earlier estimates.

System	Output	Units	Reference
Pit			
fraction urine contained	0.10		
moisture remaining in feces	1.00		
pit size	3530	L	1.8m deep 1.4m wide crib
managed mass / use (g/use)	90.00	g (ws) / use	
managed volume / use (g/use)	90.00	ml / use	
uses/pit	39222.22		
pits/year	0.05		
years to fill pit	20.92		Pits at low use sites take many years to fill up (Trewitt personal communications 2011).
cost/use	0.03	$/use	
cost / year	63.74	$/yr	
Exposure events	0.00		
DPT			
fraction urine diverted	0.95		
fraction urine contained	0.05		
moisture remaining in feces	0.90		
mass remaining	0.90		
pit size	3690	L	2.4m deep 1.4m diameter culvert
managed mass / use	68.60	g (ws) / use	
managed volume / use	68.60	ml / use	
uses/pit	53790.09		
pits/year	0.03		
years to fill pit	28.69	yrs	
cost/use	0.02	$/use	
cost / year	46.48	$/yr	
Exposure events	0.00		

System	Output	Units	Reference
WPT			
fraction urine collected	0.00		
moisture remaining in feces	1.00		
fraction of dry solids removed from tank to field by rainwater	0.20		suspended solids and dissolvable solids carried by rainwater during rain events to septic field where they are mineralized
pit size	3690	L	2.4m deep 1.4m diameter culvert
managed mass / use	67.60	g (ws) / use	
uses/pit	54585.80	uses/pit	
pits/year	0.03		
years to fill pit	29.11	yrs	
cost/use	0.02	$/use	
cost / year	45.80	$/yr	
Exposure events	0.00	#/yr	
BFO			
uses / barrel	740.74		
managed mass / use	270.00	g (ws) / use	Chapter 2
barrel change labor	0.01	$/use	
barrel fly labor / use	0.02	$/use	
heli $ / use	0.17	$/use	
Septic $ / use	0.05	$/use	
Disposal / use	0.81	$/use	
Total uses / year	1875.00	uses/yr	
mass collected / yr	540	kg (ws) /yr	
barrels / yr	2.70	barrels/yr	12 barrels removed from Bugaboo Park (1000-2000 overnight visitors) in 2010. Alpine Club of Canada 2010.
Total Cost / use	1.06	$/use	

System	Output	Units	Reference
BFO			
Cost/year	1989.80	$/yr	Bugaboo Park (1000-2000 overnight visitors) helicopter expenses for 2010 $4725. Sewage cost $1072 (includes one pit toilet). Alpine Helicopters and Alpine Club of Canada 2010.
Exposure events / year	8.10	#/yr	
UDBFO			
Urine remaining	0.50	%	50% diverted through urinal (Hopkins personal communications September 2012)
managed mass / use	170.00	g (ws) / use	
uses / barrel	1250.00	uses / barrel	
barrel change labor	0.01	$/use	
barrel fly labor / use	0.01	$/use	
heli $ / use	0.11	$/use	
Septic $ / use	0.06	$/use	thick barrels (more time)
Disposal / use	0.76	uses/yr	
mass collected / yr	340	kg (ws) /yr	
barrels / yr	1.70	barrels/yr	40% reduction compared to BFO (60% reduction occurs with urine diversion at seat as well as urinal)
Total Cost / use	0.94	$/use	
Cost/year	1770.55	$/yr	
Exposure events / year	5.10	#/yr	
VT			
tank capacity	4000	L	
managed mass / use	270.00	g/use	
uses / tank	14814.81		
Septic $ / use	0.07	$/use	
Disposal cost / use	0.01	$/use	
mass collected / yr	506	kg (ws) /yr	
tanks / yr	0.13	yrs/tank	

System	Output	Units	Reference
VT			
years / tank	7.90	$/use	
Total Cost / use	0.08	$/use	
Cost/year	141.75	$/yr	
Exposure events / year	0.00	#/yr	
Composting Toilet*			
period O&M (uses)	200.00		Compost toilet instruction manuals dowloaded from websites
mixing	0.10	hours	
trash removal	0.10	hours	
moisture manip	0.10	hours	
disposal frequency	0.50	every 2 years	
frequency of O&M / use	0.01		
hours compost O&M/use	0.00	hrs/use	
fraction of travel for compost O&M	0.40		
O&M travel hours / use	0.00	hours	
O&M $ / use	0.40	$/use	
mass bulking agent /use	40.00	g (ds) / use	Median value Table 3.3
feces mass / yr	113	kg (ws) / yr	
bulking mass / yr	75	kg (ds) / yr	
urine soaked bulking mass / yr	150	kg (ws) / yr	
combined mass / yr	263	kg (ws) / yr	
final VS	0.85	%	No significant reduction in VS% (Chapter 3)
final TS	0.20	%	Same as raw excrement (Chapter 3)
% total reduced	0.00	%	
end mass / yr	263	kg	
end mass / use	140.00	g	
uses / barrel	1428.57	#/barrel	
# of 200L barrels/year	1.31	#	333L removed on average every year or two (200-240L) (Cieslak, Compost Toilet Instruction Manuals

System	Output	Units	Reference
Composting Toilet*			
disposal $ / use	0.53	$/use	Disposal every 2 years takes 16hours total, removed by helicopter
total $ / use	0.93	$/use	
total $ / year	1745.38	$/use	$1800 before disposal, moderate use site, (Cieslak personal communications)
Exposure events / year	26.75	#/yr	
*Liquids freely draining to septic field but saturate dry bulking agent on their way through vessel			
UDVCT			
period O&M	1875.00	1/uses	Neau personal communications 2012
mixing task time	0.10	hrs	Personal observation
trash removal task time	0.00	hrs	Personal observation
moisture manipulation task time	0.10	hrs	Personal observation
disposal frequency	0.10		Neau personal communications 2012
frequency of O&M / use	0.00		
hours compost O&M/use	0.00	hrs/use	
fraction of travel for compost O&M	0.40	fraction	
O&M travel hours / use	0.00	hours/use	
O&M $ / use	0.04	$/use	
mass bulking agent /use	0.00	g (ds) / use	
feces mass / yr	113	kg (ws) / yr	
bulking mass / yr	0.00	g (ds) / yr	
urine soaked bulking mass / yr	0.00	g (ws) / yr	
combined mass / yr	113	kg (ws) / yr	
final VS	0.60	fraction	Chapter 3
final TS	0.20	fraction	Chapter 3
% total reduced	0.06	fraction	

System	Output	Units	Reference
UDVCT			
end mass / yr	106	kg (ws) / yr	
end mass / use	56.47	g (ws) / use	
uses / barrel	3541.67		
barrels/year	0.53	#/yr	
disposal $ / use	0.11	$/use	sewage fees, helicopter removal (no piggy back)
total $ / use	0.14	$/use	
total $ / year	270.68	$/yr	Ecosphere charges $500E/yr (Neau personal communications)
exposure events / year	1.60	#/yr	Personal observation

Table 7.4. Estimated capital costs for a complete single stall toilet installed at a remote site (transportation costs not included). Pricing estimates referenced where possible.

Minimum Capital Cost	Excavation & Installation	Structure	PV	Septic Field	Professional Fees	Urinal	UD Riser	Total
VT	$6,000*	$20,000*			$500			$26,500
PT	$1,500**	$5,000**			$500			$7,000
DPT	$3,500**^	$5,000**		$1,000	$2,000	$100	$2,000	$13,600
WPT	$3,500**^	$5,500		$1,000	$2,000	$100		$12,100
BFO	$1,500	$15,000-			$0			$16,500
UDBFO	$1,000	$15,000-		$1,000	$2,000	$100		$19,100
CT	$3,000	$30,000^^	$1,000	$1,000	$2,000			$37,000
UDVCT	$4,500***	$28,000		$1,000	$2,000		$2,000	$37,500

*Vault excavation and structure average of two quotes for Washington state sites by Park and Restroom (Spokane Valley, WA)
**Trewitt, BC Parks, BC
***Whalen Designs, Monroe, WA
Professional fees for soil analysis, septic field survey, permitting
^Includes cost of culvert
^^CT quotation for site in Eastern Washington (Espinosa personal communications 2012)
- Cost for a two story structure, as required for BFO and UDBFO, estimated by dividing the CT quote in half.

Table 7.5. Capital-salvage costs, maintenance costs (MC), life cycle costs (LCC), and total exposure events (EE) over a 20 year period for all suitable remote waste site management systems suitable for public remote sites at low (500), moderate (1000, 2000) and high (3000) numbers of overnight users (ONs).

500 ONs (1875 toilet uses)				
SYSTEM	C-S	MC	LCC	EE
VT	$21,200	$2,317	$23,517	0
PT	$5,600	$1,042	$6,642	0
DPT	$10,880	$760	$11,640	0
WPT	$9,680	$749	$10,429	0
BFO	$14,000	$32,531	$46,531	162
UDBFO	$16,080	$28,947	$45,027	102
CT	$29,600	$28,535	$58,135	535
UDVCT	$30,000	$4,425	$34,425	32
1000 ONs (3759 toilet uses)				
VT	$21,200	$4,635	$25,835	0
PT	$5,600	$2,084	$7,684	0
DPT	$10,880	$1,520	$12,400	0
WPT	$9,680	$1,498	$11,178	0
BFO	$14,000	$40,539	$54,539	324
UDBFO	$16,080	$34,812	$50,892	204
CT	$29,600	$40,926	$70,526	910
UDVCT	$30,000	$4,641	$34,641	32
2000 ONs (7500 toilet uses)				
VT	$21,200	$9,270	$30,470	0
PT	$5,600	$4,168	$9,768	0
DPT	$10,880	$3,039	$13,919	0
WPT	$9,680	$2,995	$12,675	0
BFO	$14,000	$56,554	$70,554	648
UDBFO	$16,080	$46,544	$62,624	408
CT	$29,600	$65,707	$95,307	1660
UDVCT	$30,000	$4,745	$34,745	32
3000 ONs (11250 toilet uses)				
VT	$21,200	$13,905	$35,105	0
PT	$5,600	$6,252	$11,852	0
DPT	$10,880	$4,559	$15,439	0
WPT	$9,680	$4,493	$14,173	0
BFO	$14,000	$72,570	$86,570	972
UDBFO	$16,080	$58,275	$74,355	612
CT	$29,600	$90,488	$120,088	2410
UDVCT	$30,000	$4,848	$34,848	32

LCC =20 life cycle costs (LCC=Capital costs - salvage costs + maintenance costs)

Table 7.6. Effects of nitrogen fertilization on productivity, diversity, litter, and microbial communities in tundra communities.

Source	Ecosystem	Fert. Rate (kg/ha/yr)	Duration (years)	Diversity / Richness Changes	Productivity Changes
Bowman *et al*. 1995	Rocky Mountain Alpine meadow	158	2	nr	Graminoids ++ Forbs +
Bret-Hart *et al. 2004*	Alaskan Tussock Tundra	100	2	nr	Graminoids ++ Evergreen Shrubs + Deciduous Shrubs + Non-vascular -
Wang *et al*. 2010	*Kobresia humilis* meadow	200	3	-	Graminoids ++ Sedge nc Forb -
Wang *et al*. 2010	*K. tibetica* meadow	200	3	nc	Graminoids nc) Sedge (nc) Forb (nc)
Richardson *et al*. 2002	Scandinavian sub-Arctic dwarf shrub heath	100	9	nr	Grasses (++) Mosses (++) Dwarf shrubs (+)

nr=not reported
nc=no change
+ increase
++greater increase
- reduction

7.7. Figures

Figure 7.1. Cost per use estimates of low-high use remote site toilet systems

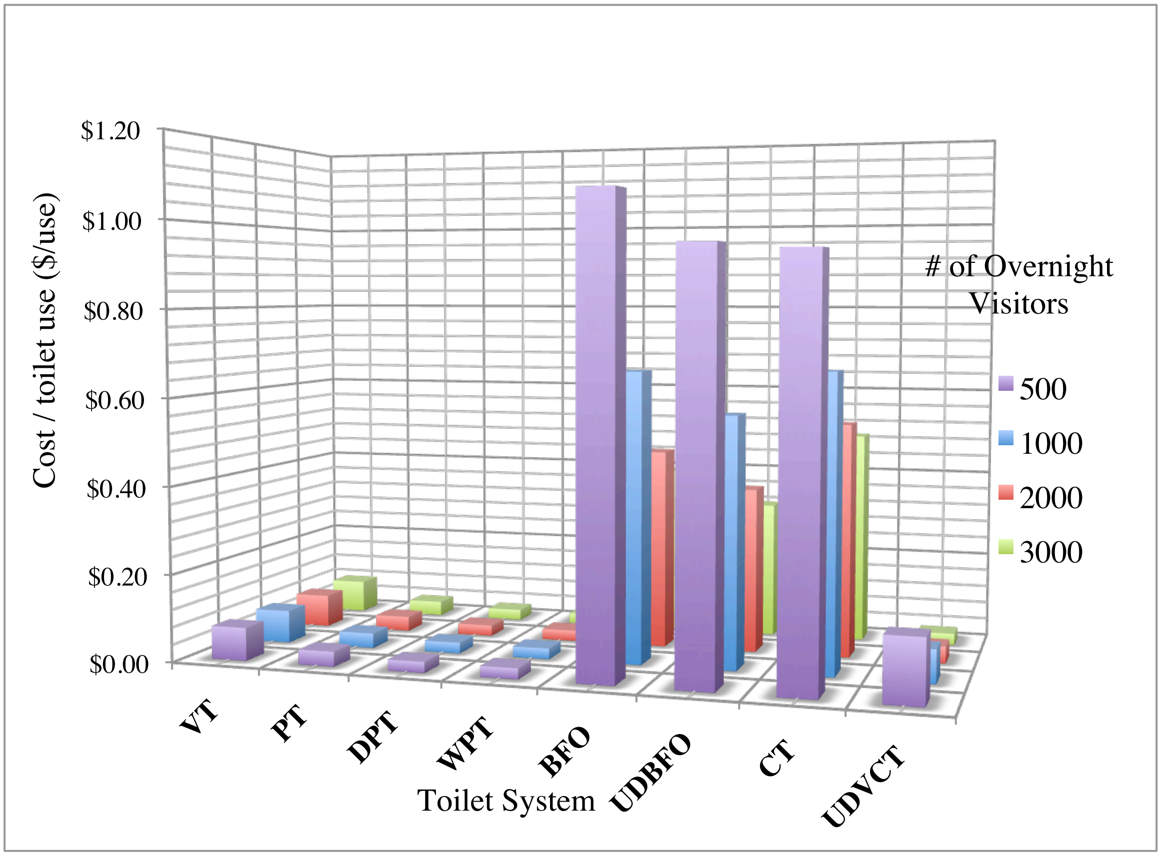


Figure 7.2. Comparisons of occupational exposure hazards associated with 20 years of maintenance of public remote site toilet systems at various visitation intensities.

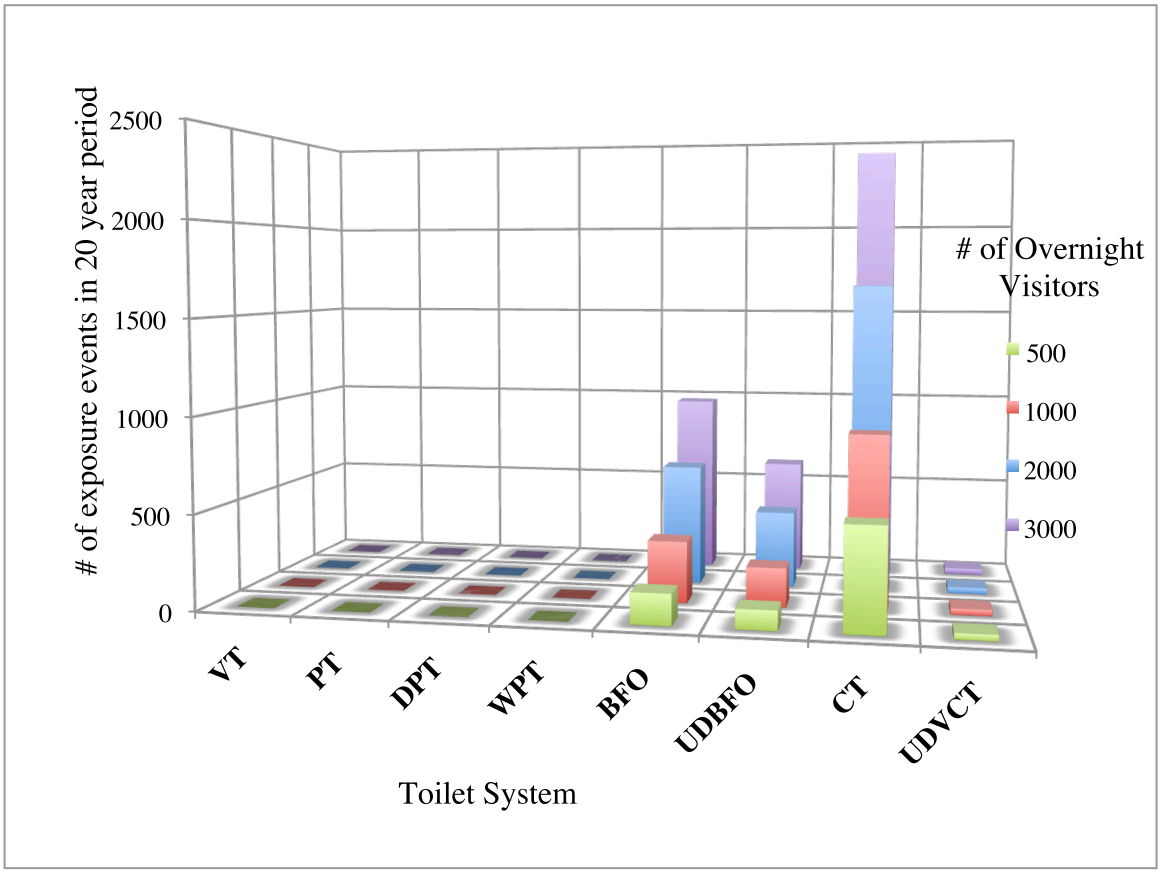
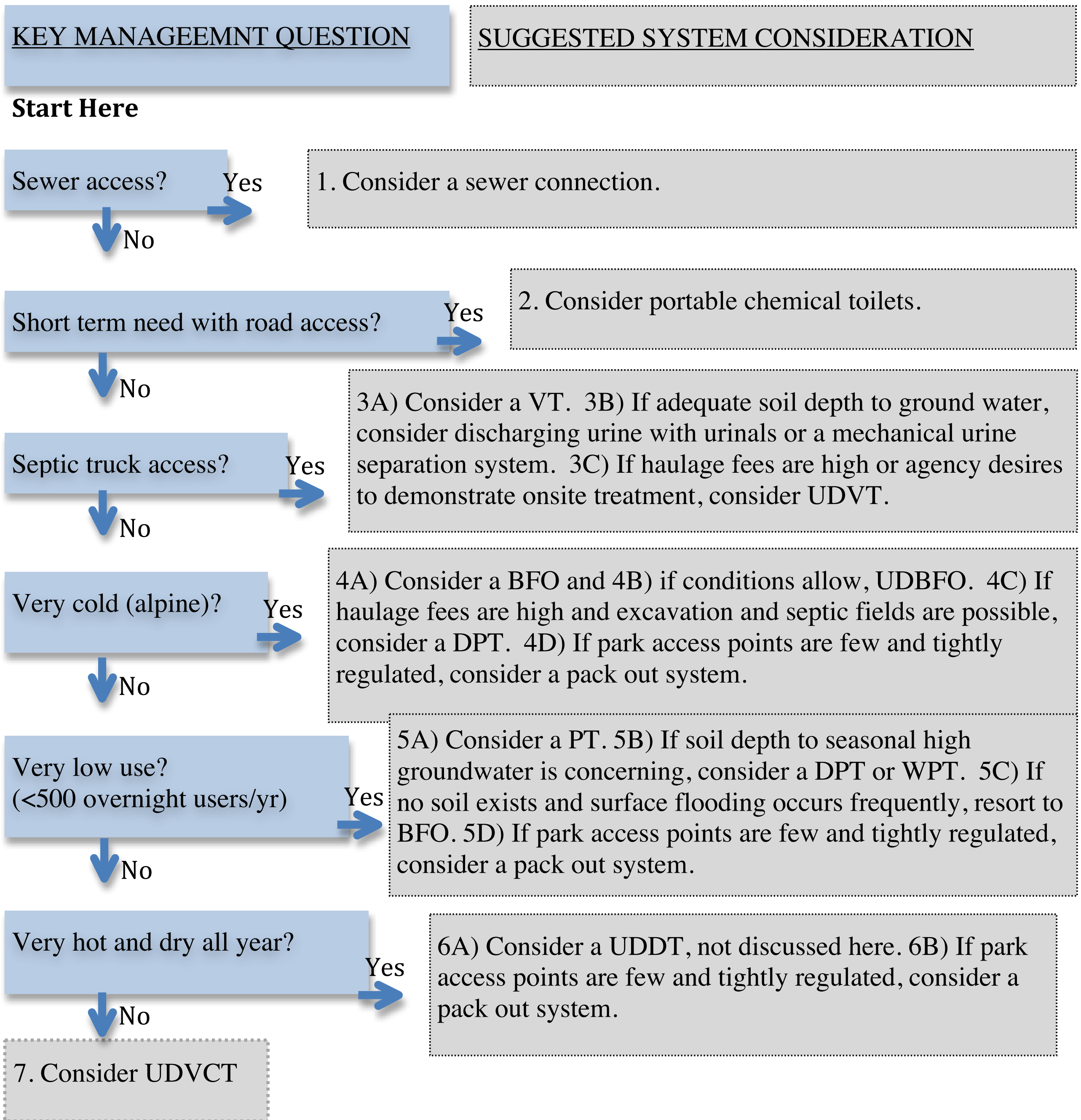


Figure 7.3. Remote site toilet system selection flow chart.



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Appendices

Appendix 1. List of contacts cited as personal communications:

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Frieda Schade, Metro Vancouver, Canada
Tom Hopkins, Department of Conservation, New Zealand
Eddie Espinosa, American Alpine Club

Appendix 2. Toilet system budget model

A model was created to benchmark the costs, hazards, and performance of each toilet system studied at various site useage intensities. It is included as an *.XLS file via CD.

Appendix 3. Additional PCA analyses

Appendix Table 3.1. Estimation of ammonia from compost pH and Solvita® NH_3 test result (modified from Woods End 2000).

Solvita® NH_3 value	1	3	3	4	5
Compost pH = 7.0	n/a	n/a	>10,000	8000	<4000
7.5	n/a	>15000	8000	4000	<2000
8.0	>20000	10000	4000	2000	<1000
8.5	>7000	3000	1500	600	<400
9.0	>4000	1500	700	300	<200

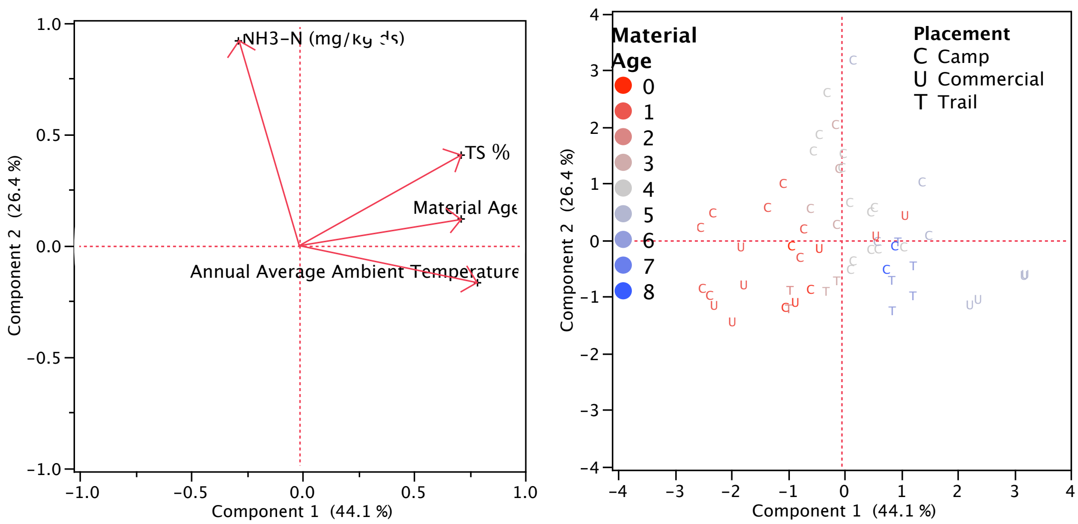
Estimated total ammoniacal-N in compost (mg/kg dry basis) (from Table #6 in Solvita® user manual).

Appendix Table 3.2. Estimation of Solvita® NH_3 from ammonium results.

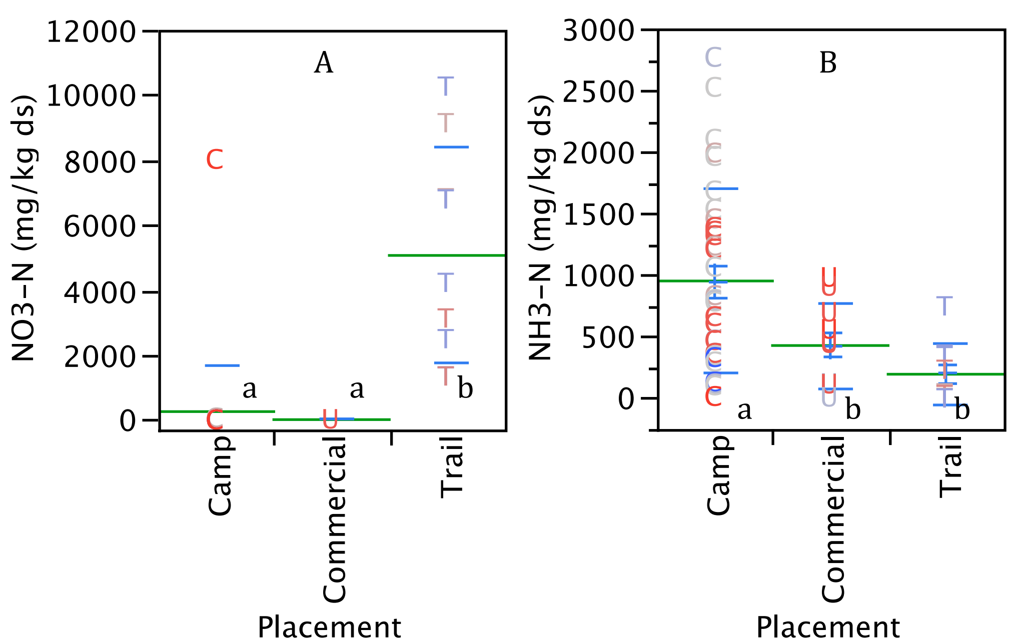
Random Sample #	Actual total Ammoniacal-N (mg/kg ds)	Solvita® NH_3 Value	pH	Predicted total ammoniacal-N (mg/kg)	Fit (yes/no)
1	7888	1	8.78	>4000	Yes
2	13121	1	8.45	>7000	Yes
3	4762	3	8.16	3000	Yes
4	1362	1	8.51	>7000	No
5	6936	1	8.66	>7000	Yes
6	12683	2	8.41	3000	No
7	9706	4	8.11	2000	No
8	25222	1	8.6	>7000	Yes
9	1572	4	7.1	8000	No
10	10181	1	8.59	>7000	Yes

Comparison between measured total ammoniacal nitrogen and estimated total ammoniacal nitrogen as provided by the Solvita® manual using 10 randomly selected samples.

Appendix Figure 3.1. PCA plots of only the dominant variables affecting end-product 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.



Appendix Figure 3.2. ANOVA tests on the placement of public CTs by (A) NO_3 and (B) NH_3 content in end-product samples. Significantly different placements denoted by different letters ($p<0.003$).



Appendix 4. **Pit toilets: a no-pollution re-design**

Introduction

In the preceding Chapters, all types of major remote site toilet waste management systems were reviewed, assessed for performance, and potential improvements tested where possible, with the exception of the PT. Pit toilets are one of the most widely used systems in the world (Franceys *et al*. 1992, Esrey *et al*. 1998) presumably because of their low capital construction costs (absence of containment or treatment tank) and low operational cost (liquids drain by gravity, solids are abandoned within the pit, and no process is maintained). The major expense with PTs occurs with the periodic excavation of a new pit and transfer of toilet structure over to the new hole. However, of all public toilet systems, PTs have the greatest risk of pathogen and nutrient transmission. The risk is the highest with thin soils and high groundwater. Soil has considerable ability to attenuate pathogens and facilitate the assimilation of nutrients through microbial mineralization and plant uptake (Moore *et al*. 2010), but these processes generally occur near the surface in unsaturated soil, which supports aerobic microorganisms (Cilimburg 2000, Sherlock *et al*. 2002, Meile *et al*. 2010). In Australia, New Zealand and the United States, public PT site selection and depth of pit relative to adequately drained, unsaturated soil is regulated to prevent ground water contamination. Australia/New Zealand require 600mm separation between the bottom of the pit and seasonal high ground water and Washington State requires 1220mm (4') (WSDOH 2007, AS/NZS 2001). Washington State also mandates a maximum pit depth of 1524mm (5'). Preliminary PT field surveys were conducted at Assiniboine Provincial Park, B.C. and through interviews with park operators, maintenance personnel and construction contractors. It was discovered that ground water could be found in PTs across the province of B.C. from Whistler (Russel personal communications July 2012) to the Rocky Mountains (McCrumb personal communications).

The prevalence of this can be traced to B.C. Parks Facility Standards (2005) where the most common PT system, having a bottom open to the soil, is excavated to a recommended depth of 2400mm, presumably to maximize holding capacity and minimize excavation frequency. However, this pit depth is 876mm (3') deeper than allowed in Washington State, which contains similar soils and biogeoclimate zones, presumably to prevent pollution and disease transmission risks, which are identified by the British Geological survey as significantly high when less than 5m of unsaturated soil exists below the bottom of a pit toilet (AGROSS 2001).

Soil depth and seasonal high ground water tables vary depending on region, elevation, and site. While the depth of unsaturated soil may be sufficiently thick enough to create an adequately safe waste treatment system below the standard PT in some regions, this will certainly not be the case at all sites, especially those at higher elevations.
At a depth of 2400mm in the B.C. backcountry saturated or rapidly draining gleyed soils, till, or parent material are frequently found (McCrumb personal communications June 2012), neither which would be permitted if PTs were regulated as septic fields by local health authorities (>610mm (24") above seasonal high ground water) (B.C. Health Appeal Board 2003). Relative to other B.C. regulations, such as >1m above seasonal high ground water tables required for farm animal burial (B.C. Ministry of Agriculture and Lands 2006), B.C. Parks PT standard construction depth appears overly deep, especially given the occurrence of seasonal high groundwater and shallower soils in the backcountry, especially at higher elevations. As was discussed in Chapter 1 and 8, saturated soil sustains pathogens increasing the risk of transmission to drinking water. Despite the commonly known risks, B.C. Parks Facility Standards (2005) state PTs, "perform quite satisfactorily in most areas, especially where the use is not too great and the groundwater level is not too high." There is considerable discrepancy between the risks associated with B.C. Parks standard PT design and B.C. Parks mandate for world-class protection and preservation of the environment (B.C. Parks 1999). However, exposing the gravity of the situation by conducting in-depth field analyses would be unlikely to induce change due to the lack of reasonable alternatives. As such, the development of an alternative PT design that minimized or eliminated pathogen

transmission, had similar capital and operating costs, and did not require frequent pumping out of the end-product (as is the case with VTs), is needed in order to move beyond the inexpensive yet polluting PT. In this Chapter, I incorporate principles of septic tank and septic field technology, which currently serve to treat ~25% of residential waste water in North America (CMHC 2012, Sherlock *et al*. 2002) into a framework for collection and treatment of waterless toilet waste, complete with critical design criteria and conclude with two designs theoretically capable of eliminating pollution of groundwater from pit toilets.

Methods

In order to maintain similarly low operating costs to PTs, the new design needed to utilize an onsite process for treatment and discharge of liquids and/or solid excrement, as the expense of collection in vaults and transport offsite for treatment of both waste streams is prohibitive (B.C. Parks Facility Standards 2005). An exhaustive literature search was made of onsite waste treatment systems that were robust, reliable, low cost, low tech, had a long history of utilization, and were founded upon a proven treatment process. Experts, technicians, and engineers familiar with these systems were interviewed with respect to the application of the design concepts to a waterless toilet environment and key criteria recorded. From this information, two designs were constructed to accommodate sites with a seasonally saturated vadose zone (ground water at or near the ground surface during visitation period) and sites with predominantly unsaturated vadose zone (infrequent saturation of near-surface soil during visitation period).

The only types of current onsite systems that meet the criteria for robust, reliable, low cost, low tech, and having a long history of utilization are septic tanks and septic fields, both of which are used for waste water treatment in 25% of North American residences (CMHC 2012, Sherlock *et al*. 2002). Septic tanks and septic fields are designed to manage and treat the waterborne effluent from residences, which include a dilute but wide array of constituents from excrement to kitchen waste and wash water. In general, septic tanks are sized based on residential occupancy ranging from 1800-3600L/residence and are designed to hold liquid this amount of effluent for a 24h residence time period

The primary objectives are to settle solids and float scum (oils and fats) preventing both from clogging down stream orifices. But, under certain conditions, they may be capable of anaerobically mineralizing a small fraction of organic matter into gas (CHMC 2012). In general, septic tanks are simple to maintain and operate requiring infrequent routine maintenance including sludge removal and filter cleaning/replacement every three years (CMHC 2012).

Septic fields are designed to disperse clarified effluent into natural, undisturbed, unsaturated and highly active soil in order to adsorb, assimilate and immobilize nutrients and attenuate pathogens (CMHC 2012, Sherlock *et al*. 2002). Septic field design and construction is regulated provincially, by State, and permitted and enforced by local health districts (CMHC 2012, Snohomish County 2004). Meile *et al*. (2010) report that fully functioning septic fields work well if installed into the appropriate type and depth of unsaturated soil. However, considerable evidence exists that when septic design, installation, and service regulations are not followed effluent, particularly nitrates, can be traced to adjacent groundwater or surface water bodies (Sherlock *et al*. 2002, Wilhelm *et al*. 1996, Werrick *et al*. 1998, Heisig 2000, Hatt *et al*. 2004). Horizontal separation of >60m and maximized residence time (months) in unsaturated vadose zone are two key important criteria of fully functional septic field systems (Meile *et al*. 2010, Sherlock *et al*. 2002). A septic field setback of 58m from surface water discharge zone eliminated 85% of input nitrate in a Georgia marshland (Meile *et al*. 2010). Residence times of effluent in the non-saturated vadose zone of glaciated podzolized hillside soil in New York varied from 200-500 days (Sherlock *et al*. 2002), which should provide adequate time for adsorbtion, immobilization, mineralization, and assimilation of nutrients (Jones and Murphy 2007) and attenuation of pathogens (Cilimburg *et al*. 2000). Similar results were found by Curry (1999) where the majority of tracer remained below septic field distribution lines for months.

Excrement deposited into pit toilets are a concentrated constituent of waste treated by residential septic tanks and septic fields and in theory should be able to be similarly treated by these systems with minor modifications. Such as system could be a low cost alternative to the pit toilet and could drastically reduces pollution to ground water such as septic tanks and fields have done for residential waste (Bianco personal communications September 2012).

Results and discussion

The list of important criteria to consider when integrating septic tank and septic field concepts into a waterless toilet environment are outlined in the Appendix. Two design prototypes emerged to meet the design criteria: 1) complete diversion of liquids to a septic field and isolation of remaining fecal matter in an inexpensive septic tank and 2) an excrement dilution system which passes rainwater through the tank in order to dilute blackwater, minimize odors by reducing residence time of ammonium salts, and increase longevity of the pit by moving dissolved or suspended solids through the filters to the septic field where they can be mineralized. The former will be called a dry pit vault toilet (DPVT) and the latter a wet pit vault (WPVT).

Septic fields are employed to uptake nutrients, which may result in fertilization impacts. Fertilization impacts based on estimated high-use site inputs are discussed in Chapter 7. At sites where septic fields will be of little improvement over the conventional pit toilet dug to 2400mm below ground surface due to year round high groundwater or frequent flooding, a total containment system should be utilized such as a BFO or VT.

The key feature of the DPVT, as described in the Appendix, is a robust and efficient urine diversion device integrated at the toilet riser. If the vast majority of urine can be diverted from the tank, the remaining fecal matter can be essentially isolated from the environment. As a backup, should the tank fill up with water or urine during the initial research and development stages, the containment tank can be connected to the septic field into which urine is discharged. The failure of the commercially available urine diverting seat was discussed in Chapter 2. The success of Ecosphere Technologies ™ (France) patented inclined treadmill was discussed in Chapter 3, but its cost is >$5000CDN making it an unrealistic option (Neau personal communications March 2012). There are no North American products currently available to fill this need as the urine diversion industry is in its infancy here (Shiskowski 2009). The reason this system has not been invented or tested to this point in time is most probably because of the lack of this key component. Urinals remove a considerable amount of urine (~40% of excrement stream, Chapter 2) but without this robust and efficient urine diversion system at the toilet seat able, capable of diverting the remaining urine (20-30% of excrement stream), the container will fill up more rapidly than required in order to compete with the equivalent standard open-bottom pit from which urine drains freely. If this device were invented and mass produced, it could provide a viable means for which pollution from standard open bottom pit toilets were eliminated, as fecal pathogens could be virtually isolated from the environment and risk of water bourne disease reduced. Nutrient leachate to water bodies from urine discharge would still need to be mitigated and will be discussed further in Chapter 7. If the urine separation system is efficient enough to keep ammonium and salt concentrations below toxic thresholds for invertebrates and the ambient moisture were high enough to prevent material from drying out, it is possible than the fecal matter in the DPVT could be consumed by detritivores or introduced earthworms. Decomposition and mineralization stimulated by vermicomposting discussed in Chapter 3 would further extend the useful life of the containment vessel. If / when this robust urine diversion mechanism is invented, and has been proven effective at diverting urine added to a toilet riser by men standing and men and women sitting, its reliability should be tested in a long-term study conducted in various climates to ensure its usefulness in the range of environments pit toilets are found (hot, dry, cold, wet). Fertilization and eutrophication impacts should also be studied as discussed in Chapter 7.

Alternatively, or until such a urine diversion device is available, the pit toilet can be transformed into a WPVT, copying the design of the septic tank and field treatment system in order to shift effluent discharge from subsoil, as is the case with standard PTs, to near surface soil (as is the case with septic fields). This will maximize nutrient uptake and pathogen attenuation and minimize ground water pollution. In this design, rainwater can be used as a replacement for flush water, diluting the effluent, settling solids, and eventually being discharged to a septic field after passage through an effluent filter. Other differences between this design and the residential septic system are: direct defecation into the septic tank and abandoned of the tank once full rather than evacuation and reuse due to the impossibility or prohibitive cost of septic haulage from remote sites. The ideal site for this would be one with periodic rainfall throughout the season of use yet sufficient and predominantly unsaturated soil so as to construct a high quality septic field, and considerable separation distance from surface water as human pathogens will be carried into the septic field in the diluted effluent (70-2000m depending on soil type and depth (Moore *et al*. 2010) (Appendix).

The WPT is very similar in design to the aqua-privy used in the developing world, with the main difference in that the WPT container is likely to be abandoned *in situ* rather than being drained and re-used as is the case with aqua-privies (and VTs in North America) (Biellik *et al*. 1982). Water was more commonly introduced into the aqua-privy by hand during anal cleansing or through deposition of laundry water and in general were found to function satisfactorily where effluent disposal fields or soakage pits were constructed properly (Biellik *et al*. 1982). By extending the excrement drop chute below the level of the effluent drain, submerging it below the level of liquid in the tank, odors and insect problems were reduced, as is similarly accomplished in the residential setting with p-traps (Biellik *et al*. 1982). A variety of instruments can be fashioned to check the level of solids at the bottom of the tank.

The main aspects of this system that would benefit from testing are the degree to which solids settle and the resulting longevity of the effluent discharge filter, the pathogen content of effluent (as the dilution effect will reduce ammonia concentrations which likely have some negative effect on pathogens in standard latrine waste), longevity of the septic field, and an overall operations and cost comparison to the standard PT.

Conclusion

Pit toilets are the most widely used backcountry waste management systems but when insufficient soil exists below the pit and above the seasonal high ground water level, as is commonly found in B.C., they can be highly polluting and a source of disease transmission. A mechanical urine diversion system is needed which can be placed between the toilet hole and the pit to divert urine into near-surface soil, preventing ground water contamination, and fecal matter into a lined pit, preventing transmission of pathogens.

If reliability were proven over longer term tests, this system could be retrofit in existing pit toilet structures during the excavation of new pits at a fraction of the capital cost compared to an alternative toilet structure such as a VT, BFO, or UDVCT, in order to maintain low costs per use but greatly reduce groundwater impacts and disease transmission risks.

Appendix Table 4.1. Criteria and recommendations for the integration of septic tank and septic field designs and components into waterless human waste management.

Waterless Septic Tank & Septic Field Criteria	Recommendations
Inexpensive impermeable resistant tank that ideally does not require cribbing yet eventually oxidizes or decomposes >3 years after abandonment (after pathogens have died).	Galvanized culvert stood on end (Z610 galvanizing for long term (15-50 years) rust prevention) (Canada Culvert).
Impermeable bottom cap to prevent pathogen transmission to subsoil/groundwater.	Watertight gasket used between culvert and end cap or welded seam (Canada Culvert 2012).
Optimize tank size and dimensions balancing capacity (maximize), excavation costs (minimize), tank cost (minimize), surface area to volume ratio (maximize settling).	More research is needed here. The standard culvert used in standard PT and VT applications is 1400mm diameter 2400mm tall.
Minimize blackwater effluent where soils in septic field are thin or periodically wet	If soil is saturated during the season of visitation and budget allows, a complete containment (VT or BFO) and offsite treatment should be conducted. If budget is not permitting or high ground water infrequent, and climate is not optimal for urine evaporation, divert all urine away from septic tank to septic field through urinal and robust urine diversion mechanism (Chapter 2) to minimize discharge of blackwater through septic tank filter to septic field. Prevent rainwater or groundwater from entering collection tank. Provide effluent discharge filter and port to septic field as a back-up system in case culvert floods with rainwater or groundwater. This system will be called a dry pit vault toilet (DPVT).
Minimize capital and operating costs, odors, and maximize containment capacity where septic field soil is predominantly unsaturated of adequate depth and texture.	Divert urine with a basic urinal direct to septic field and route rainwater into the septic tank to dilute remaining urine, minimize volatilization of ammonia, and treat residual effluent in the septic field. This system will be called a wet pit vault toilet (WPVT).

Waterless Septic Tank & Septic Field Criteria	Recommendations
Prevent septic tank effluent from clogging septic field infiltration orifices	Use modified septic tank filters that can be cleaned periodically due to higher TS% content or that are additionally protected by wrapping them with multiple layers of coarse mesh to prevent rapid clogging once the level of solids rises to the level of the filter (signally the near end of the pit's useful life
Be able to identify when containment vessel is full of solids and should be decommissioned	Liquid effluent (remaining urine and rainwater if rainwater if routed in) will drain from the septic tank through the filter to the septic field. If the pile of fecal matter extends above the outflow discharge level and changing the filter doesn't reduce the level, the tank will have filled and will need to be decommissioned.
Be able to safely cap and decommission containment tanks.	Use watertight gaskets and culvert caps similar to those used for the containment vessel's bottom. This will minimize the depth of soil needed to safely cover the abandoned tank.
Prevent groundwater hydrostatic forces from lifting containment tank	Incorporate ballast on the bottom and sides of the culvert as described in BC Parks Facility Standards (2005).
Reuse septic field after decommissioning containment tanks	Establish a long term site plan identifying a series of excavation and toilet sites that move uphill from the septic field so that the original septic field can be re-used with each subsequent toilet site.
Prevent freeze damage to piping	Freeze damage is most likely to occur where water completely fills a pipe or vessel. Design system to enable complete gravity drainage through over-sized pipes from tank to field. Use high capacity infiltration products (half barrels) to increase surface area in the septic field. Low flow volumes and high salt content will assist in preventing freeze damage.

Waterless Septic Tank & Septic Field Criteria	Recommendations
Minimize septic field size while ensuring sufficient area for treatment of concentrated nutrients in urine	More research is needed to evaluate the efficacy of near-surface soil in adsorbing, immobilizing, assimilating, and denitrifying nitrogen and attenuating pathogens in low flow urine only or urine + rainwater/blackwater effluent. Some important monitoring parameters are discussed in Chapter 7.
Maximize residence time in vadose zone of septic field	Minimize vertical flux of effluent below septic infiltrators. Consider testing the efficacy of an impermeable liner under an enlarged septic field to reduce percolation, maximize horizontal movement, and increase residence time in vadose zone, especially where soils are thin or wet. This naturally occurs where soil sits on an impermeable bedrock or permafrost layer.
Maximize distance from surface water bodies, especially where fecal effluent is discharged to the septic field.	Assuming at least 1m of unsaturated soil exists below the septic field, at least 60m separation distance are required for >85% nitrate removal (Meile *et al*. 2010) and in the case where rainwater is used to dilute fecal matter, bringing fecal effluent to the septic field, 70-2000m are required for 11log reduction in Rotavirus (depending on the soil type, see Moore et al. (2010) for a guide to estimate based on site conditions).

Appendix 5. *Ascaris suum* ova viability methodology.
Written and conducted by Benchmark Labs, Calgary, AB.

Adapted from

Kato, S., Fogarty, E., and Bowman, D. (2003) Effect of aerobic and anaerobic digestion on the viability of Crytosporidium parvum oocysts and Ascaris suum eggs. International Journal of Environmental Health Research. **13**, 169-179.

1. Summary

This protocol is designed for the determination of viability of *Ascaris Suum* eggs using Excelsior sentinel chambers, following exposure to various environmental conditions. *A. Suum* eggs are used as on indicator or proxy for the inactivation of parasite eggs in biosolids, compost, or other potentially contaminated medium. Conditions witch inactivate *A. Suum* eggs, will render them unable to emryonate and will arrest their development into mature, infectious larvae. Although *A. Suum* is not considered to be a human pathogen, care should be taken in working with the sentinel chambers as they may have come into contact with true human pathogens. Additionally, the parasite is a verified pathogen of swine. After ingestion of the eggs, the parasite hatch in the GI tract. They then penetrate through the intestinal wall, migrating to the lungs. Once in the lungs the worms can grow, cause severe inflammation and serious complications. Accordingly, all contaminated material must be autoclaved and disposed of following use. Work should be conducted in the bio-saftey cabinet as much as possible. Proper hand washing and decontamination of workspace must also be followed with working with this organism.

2. Equipment and Reagents

1. Centrifuge capable of 2000 g.
2. Conical centrifuge tubes capable of withstanding 2000 g
3. Mechanical shaker
4. 38 um Sieve
5. T25 Culture flasks
6. Incubator (28C)
7. Biosaftey cabinet
8. Autoclave
9. Vortex (Optional)
10. Microscope
11. Deionized water
12. Supersaturated magnesium sulphate solution (specific gravity = 1.2)
13. Formalin (37% Formaldehyde solution)

3. Procedure

3.1) Cut open screen on sentinel chamber with a scalpel and carefully wash the material into a 15 mL conical bottom centrifuge tube with ddl H_2O. Fill tube with ddl H_2O.
3.2) Place tube on mechanical shaker on high for 15 min to homogenize material.
3.3) Centrifuge the homogenized material at ~1800 x g for 1 min. Decant or aspirate off

the supernatant.

3.4) Fill tube with a supersaturated magnesium sulphate solution (specific gravity = 1.20) Shake vigorously or vortex to re-suspend the solution.

3.5) Centrifuge again at ~ 1800 x g. The Asaris eggs should float to the top.

3.6) Pour the solution over a number 400 (38um) sieve. This is a small enough pore size to restrict the eggs, effectively concentrating them on sieve surface. After the liquid has passed through the sieve, wash the eggs across the surface by holding the sieve at an angle. Wash the eggs onto the lower edge, and re-suspend in a small amount of ddl H_2O (< 10 mL).

3.7) Pour the egg suspension in to a T25 style culture flask. Fill to the 10 mL mark on the flask then add 0.05 mL of 37% formaldehyde.

3.8) Place in incubator at 28 degrees Celsius for 3 weeks to allow embryonation.

3.9) Modify a pipette by widening the opening at the tip. This can be done by simply cutting off the end of the tip with a pair of clean scissors. This will prevent you from restricting the flow of eggs into the pipette. Pipette a small amount of the suspension onto a clean microscope slide, being careful to prevent spilling of the liquid off the slide. DO NOT use a cover slip , as this will likely push the eggs toward the edges of the slip.

3.10) Observe the eggs immediately using the 10X ocular. Eggs with be clearly visible as small brownish coloured circles. Click this link the view images of the developmental stages of Ascari suum eggs.
http://www.stalosan.dk/stanosanf.php?action=visoth&id=12
Viable eggs with have a small larva clearly visible in the centre, while inactivated eggs will have a dense, shapeless mass inside with no clear structure. Record the total eggs counted and number of inactivated eggs.

4. Quality Control

4.1) Count greater than 50 total eggs from 3 separate slide preparations to ensure homogeneity and validity of results.