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The State of Technology and Community Driven Application of Distributed Wastewater Reuse, Nutrient Reclamation, and Energy Savings

Author
Gocke, Thomas Edward

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The State of Technology and Application of Distributed Wastewater Reuse, Nutrient Reclamation, and Energy Savings

THESIS

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MASTER OF SCIENCE

in Civil Engineering

by

Thomas Edward Gocke

Thesis Committee:
Professor Sunny Jiang
Professor David Feldman
Professor Diego Rosso, Chair

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ABSTRACT OF THE THESIS

The State of Technology and Community Driven Application of Distributed Wastewater Reuse, Nutrient Reclamation, and Energy Savings

By

Thomas Edward Gocke

Master of Science

in Civil Engineering

University of California, Irvine, 2014

Professor Diego Rosso, Chair

The security of clean water for urban communities is increasingly uncertain due to over usage, a shifting hydrosphere, and changes in development patterns. The wastewater treatment community has come to a turning point, where wastewater is increasingly being viewed as a valuable resource that can be transformed into commodities such as clean water, nutrients and energy. This document will discuss the current state of the industry for water reuse and nutrient reclamation and evaluate each practice based on the feasibility of implementation in communities of different sizes.

Planning research has focused on distributed wastewater treatment systems primarily in two settings. The first setting is in the “developing” world where communities often lack rigorous water conveyance and treatment systems. The second is in the developing fringe on the

outskirts of already developed urban centers. In both settings decentralized water treatment technologies have often become synonymous with low energy water and energy saving technologies.

Research on engineering solutions for the recovery of nutrients and clean water, either for local use or export, has made tremendous strides. However, the application of these technologies has been confined to conventional centralized wastewater treatment plants. This document draws upon existing wastewater planning practices and evaluates how communities could best implement a distributed network of reuse and reclamation facilities. It proposes that the scale of distributed wastewater treatment and reuse systems could open up the planning process to members of the community that have historically not had a say in the planning of their community.

INTRODUCTION

The State of Water – Sanitation and Potable Water

In 2008 the World Health Organization (WHO) and United Nations Children's Fund (UNICEF) issued a combined report from their Joint Monitoring Programme for Water Supply and Sanitation (JMP). In the report the WHO/UNICEF characterizes the large sanitation deficit in the world with 2.5 billion people lacking access to improved sanitation, and with 1.2 billion of those people lacking access to any facilities whatsoever (WHO/UNICEF. 2008). In this report, improved sanitation is defined as, "facilities that ensure hygienic separation of human excreta from human contact," (WHO/UNICEF, 2008, 14). There are remarkable trends in this report including the fact that a disproportionate amount of the populations of sub-Saharan Africa and South Asia are lacking sufficient sanitation. Also, there is a pronounced difference in the level of sanitation in urban and rural settings globally with an average coverage of 79% in urban settings and 45% average coverage in rural settings (see Figure 1). Of particular note in the report is that in developing countries access to improved sanitation for the highest economic quintile (85% coverage) is three times greater than the for the lowest quintile (28% coverage) (WHO/UNICEF, 2008). The concern with regard to lack of improved sanitation and its link to public safety is that that if even a small portion of a community is disposing of its waste without sanitation it can dramatically increase the risk of diarrhoeal diseases, hepatitis, and worm infections for the entire community.

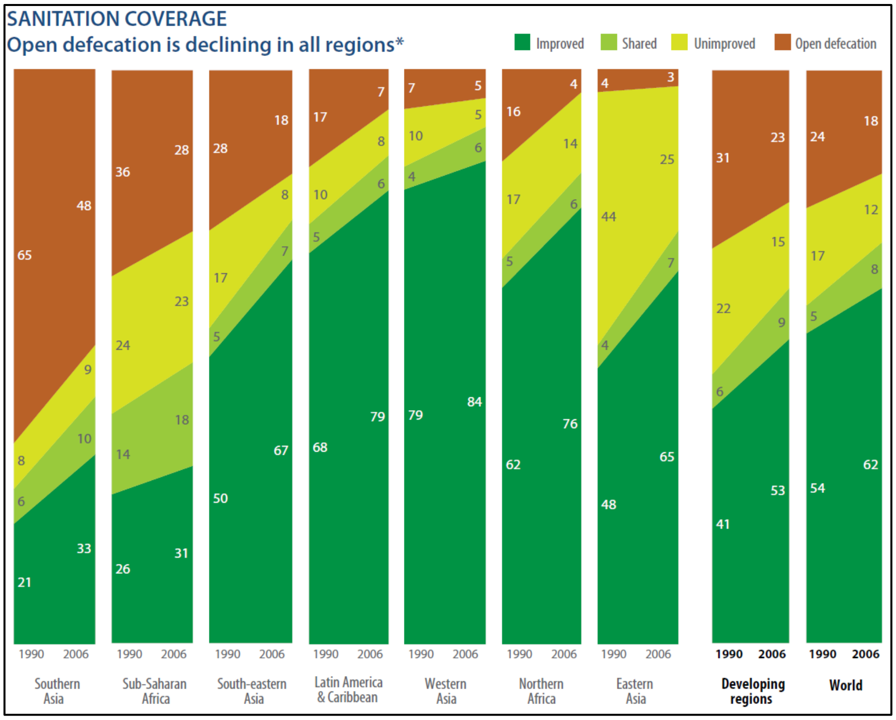

Figure 1: Composition of sanitation available to populations in MDG regions. Note: Ocean and Commonwealth of Independent States omitted due to incomplete data (WHO/UNICEF, 2008).

The state of potable water supply is dramatically different from sanitation, with up to 5.7 billion (87%) of the world's population having access to improved potable water sources in or near their homes. Of this 87%, there is 54% that uses piped connection and 33% from other sources (see Figure 2). Improved drinking water from piped connections is twice as likely in urban settings than in rural while the other improved sources (the 33%) are twice as likely in the rural setting (WHO/UNICEF, 2008). This is due to the nature of the other improved sources which include, "public taps or standpipes, tube wells or boreholes, protected dug wells, protected springs and rainwater collection," (WHO/UNICEF, 2008, 31). A problem that has become evident in both sanitation and potable water supply has been that although urban settings tend to be better equipped in comparison to rural settings, the urban coverage has been unable to scale up with the increase in urban population. This has meant that although the proportion of urban population that has access to improved sanitation and drinking water has increased the real number of people without access to improved sanitation and drinking water has also increased (WHO/UNICEF, 2008).

There are a great number of physical, biogeochemical, and cultural challenges facing planners and engineers as they attempt to provide sanitation and potable water systems to those communities that do not have them and improve those systems that already exist. These include the increasing uncertainty of the availability and reliability of water supply across geographical areas (Jackson *et al*, 2001). It also includes systemic problems with regard to integration and communication between the different types of water infrastructure at the local, regional, and global scale, as well as the potential damage that poorly designed water infrastructure can have on ecosystems when nutrients are released into the environment (Lienert, Schnetzer, and Ingold, 2013; Kalayrouziotis, 2011, Thawaba, 2006).

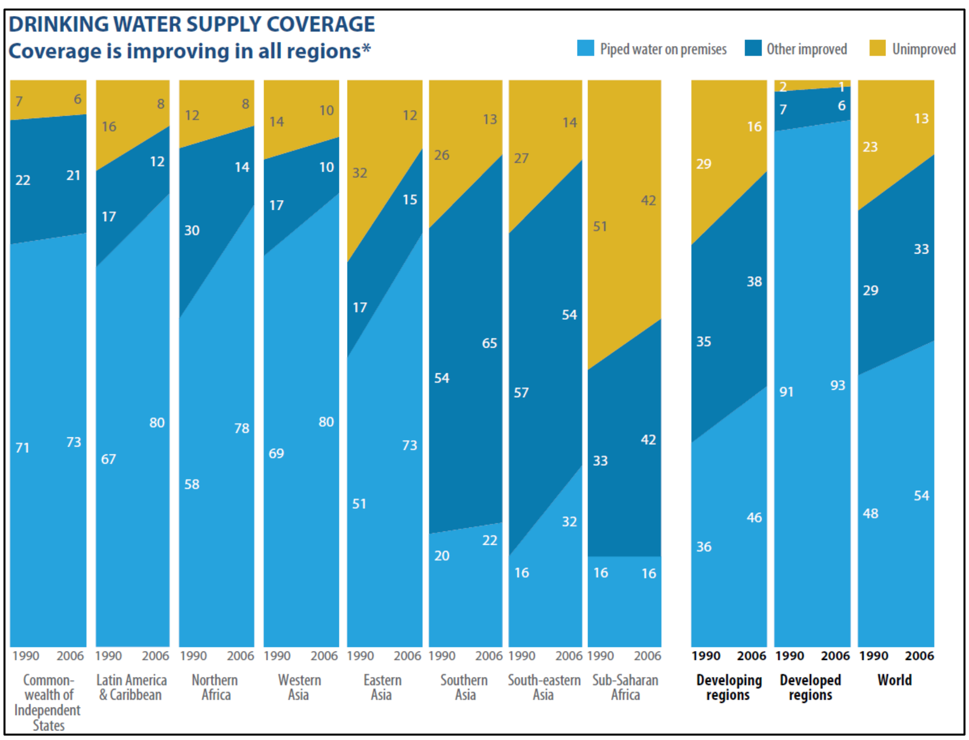

Figure 2: Composition of potable water sources available to populations in MDG regions. Note: Oceania omitted due to incomplete data (WHO/UNICEF, 2008)

One of the innovative ideas that is gradually taking greater and greater space in water policy is the idea of taking wastewater treatment plants and redesigning, retooling, and reclassifying them as water reuse plants; effectively taking the waste out of wastewater (Asano, 2002; Grant *et al*, 2012; Wallis-Lage *et al.*, 2011; Mintz *et al*, 2001). There are a great number of environmental reasons for doing this, and it can be beneficial to human health, wealth, and standard of living. In natural biological and physical systems the concept of waste doesn't exist (McDonough and Braungart, 2002). This is called a cradle-to-cradle natural philosophy where all components are in closed circuit loops, where the "waste" of one organism is the vital resource for another (McDonough and Braungart, 2002). Some view this as the natural state our water treatment systems should mimic (Verstraete and Vlaeminck, 2010). Through our sheer abundance and by our social proclivity to live in concentrated urban centers our production of human waste has the potential to cause environmental changes that are harmful to human health, wellbeing as well as ecosystems in general (Garnier *et al*, 2007; Grant *et al*, 2012; Seng et al, 2009). There are also a whole set of stressors on water security and biodiversity that are expected to become more prevalent within the coming decades due to changes in land use, global climate change, and regional contestation (Vorosmarty *et al*, 2010; United Nations, 2009).

In order to incentivize the idea of a resource plant, there has been extensive research in regards to how a plant might be designed. There are potential opportunities for municipalities to lower the cost associated with the disposal of their waste, or potentially make a financial gain from the exportation of reuse product (Wallis-Lage *et al.*¸2011). Others have proposed the idea of redefining water reuse programs so that they are designed and operated like potable water production facilities (Cushing *et al*, 2008). Questions have been raised regarding what would it take to build such a plant? Who should build it? And at what scale can they be built? Should they be the centralized massive treatment facilities of this past generation? Or can they be distributed systems of a small flux such that communities can optimize what they choose to optimize? Do community members truly have a say in their infrastructure and should they? These are all questions discussed throughout this paper in the context of the current reuse technologies and the linked history of planning and sanitation.

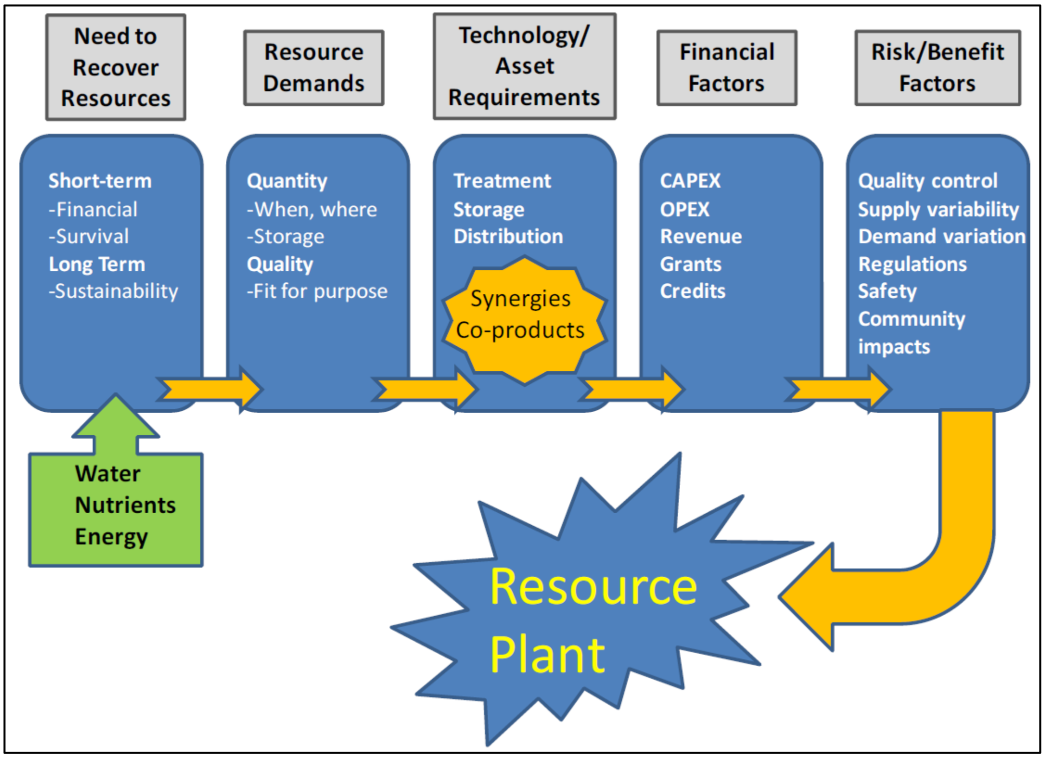

Figure 3: Development of a Resource Plant based off of a Business Case Philosophy. It should be noted that often the prioritization of one type of recovery precludes the recovery of another resource (Wallis-Lage *et al*, 2011).

THE PLANNING OF WATER INFRASTRUCTURE

Community Involvement in Planning

The history of urban and regional planning as a professional practice and as an academic discipline in the United States began with the utterly inadequate disposal of waste within the growing cities of the late nineteenth and early twentieth century (Hall, 2002; Krueckeberg, 2002). Chicago increased from around 500,000 residents to almost 1,700,000 residents in the twenty years between 1880 and 1900. New York grew by approximately 1.5 million residents, an increase of 78% (Krueckeberg, 2002). The cities became filled with enormous poorly constructed tenements, land uses were not regulated with respect to their adjacency to other land uses and there was little to no sanitation infrastructure in place for the removal, let alone the treatment of human waste (Hall, 2002; Krueckeberg, 2002). The planning profession grew out of this environment and it did so in large part through evolving from professionals in the engineering and landscape architecture professions (Hall, 2002). The earliest roles that the planner engineers took were in designing sewers for waste in the nation's cities. There was also tremendous effort put into the building of parts and open spaces for urban centers with Frederick Law Olmsted's Central Park in Manhattan being a formative example (Krueckeberg, 2002). The goal of these projects was to alleviate the serious public health problems associated with the rapid growth of these cities and to make them more livable.

The early history of planning in the United States and its links to early sanitation engineering and the goal of increasing public health has very large ramifications for the design, construction, and operation of water infrastructure today. The treatment of wastewater has been shown to significantly benefit the public health of communities and is seen as a basic necessity today (Naik and Stenstrom, 2012). The profession of planning developed from its early roots and became focused on designing cities as efficient mechanical machines with an emphasis on beautifying the city centers (Hall, 2002). The culture of professional planning became dominated by this devotion to input-output, top-down, nominally benign paternalism guiding zeitgeist. Professional planners have been able to use zoning of land use and eminent domain due to a series of landmark legal cases in the twentieth century (Selmi, 2008). The main concern with the application of zoning, eminent domain, and the building of infrastructure in the United States in the intervening years is that if any public participation was sought it was in the majority of cases sought at the tail end of decisions and projects (Hall, 2002). Despite a wide array of criticism of this planning philosophy, the philosophy persists, and is much embedded in planning departments to this day (Hall, 2002; Low, 2004; Wright, 1991). Also of note is that the seemingly benign paternalism of planning had a very colourful history of being exported to the colonies of European Powers and the United States in the late eighteenth century and early nineteenth century and continues to play a large role in the expression of power in the global south (Hall, 2002; Wright, 1991). The modern engineering professional community is very conservative and has a high aversion to risk, which can at times prevent innovation and the prudent adoption of beneficial management techniques and technologies (Farrelly and Brown, 2011; Shirley-Smith and Butler, 2008; Starkl *et al,* 2009).

Urban Centers and their Role in Water Management

There have been many attempts by professionals in both the planning and engineering industries to address these long term patterns within their disciplines. This has been in response to the perception that some planning and engineering practices cause preventable problems as urban spaces are developed. At the same time there has also been significant research that posits urban centers as being by their very nature the cause of conflict that occurs between stakeholders within the community. Current research has highlighted three major theories that account for how cities create difficulties for water management (Feldman, 2009).

The first of these theories is the city as a “growth machine” where the extraction of resources is used to maximize the production of goods for material gain of elites within a community. In this paradigm water is ultimately seen as a resource to be used to maximize the growth of wealth for elites and therefore decisions on how water is managed are made by those elites (Brannon, 2007). Such a system is shown to feature competition between different elites within a community. This competition is the root cause of the mismanagement of water and causes a lessening of financial efficiency for the community as a whole. Such a paradigm is in line with conceptions where non-elite stakeholders are not consulted until the end of development (Hall, 2002; Low, 2004).

A second theory of the city and its role in water management is the city as a “zero-sum conflict producer.” Under this theory any production process or use of natural resource (e.g. water or wastewater) within the city creates a loss for every corresponding gain (Feldman, 2009). This is complementary with the conceptualization of city as a “growth machine” discussed above since the competition among elites is to determine who among them accrues the gain and who else suffers the loss. This second theory of “zero-sum conflict producer” however describes such competition between elite and non-elite individuals as well as among other divisions such as class, race, ethnicity, and gender. This theory explains why cities place a greater emphasis on supply-side management over demand-side management. Under supply-side management cities and the people in them see water as a resource that should be obtained before someone else is able to (even if the resource itself is not needed yet). This excess in obtained supply can often lead to the minimization of conservation of water, energy, and money as there is less of an impetus to do so. Under such a paradigm of the city, growth is what is prioritized over community livability or equitable allocation among stakeholders.

A third conception of the city is as “metropolitan nature,” which differs from the “growth machine” and “zero-sum conflict producer” conceptions of urban centers. Under the theory of metropolitan nature cities grow as they do (and water is managed as it has been) is not due to failures in planning or engineering but is instead due to the natural dynamics inherent in human interaction (Feldman, 2009). Under this theory the environment and the people in it are co-adapted to each other in a dynamic process which creates the best system (city) at any given moment. Any perceived failures as the environment and natural resources are altered and harnessed is due to natural contention that occurs as “progress” is achieved. In this conception the environment is being mastered and controlled for the summed interest of the community. Ultimately this can lead to a less nuanced break down of interests discussed in the prior two theories which can lead to a lack of concern for community members lacking political or financial power. This can justify any lack of horizontal and vertical integration in the planning process as any “growth” or “development” in a community is used as a metric for community success in lieu of equity.

This third theory of the city ultimately is one in which the city is heralded as a natural and perfect order created by all the individuals within a community trying to maximize their own utility and thereby summing into what we call the city (Glaeser, 2011). Under such a theory individuals are naturally predisposed to create dense urban environments due to the wealth, social mobility and leisure at their center. Under this conception people seek to maximize their quality of life and therefore seek to move to urban centers (Glaeser, 2011). Such a conception fails to take into account the structural forces that are addressed in the other two theories of the city. These include the unequal force exerted on development agencies by elites that have vested interest in construction and amenities. It also glosses over changes that might be forced on people in the rural environments that cause people to move. Ultimately it is the responsibility of planning professionals and engineers to take into account these differences in power relations as they try to manage our natural resources for the common good.

More Inclusive Planning Methods

The field of urban planning and infrastructure development has had a long history of having top-down decision making, where planning imposed on whole populations of the urban settlement without their input or consent (Hall, 2002). However, with time, there has been a blossoming in emphasis placed on understanding who the stakeholders are within any given project, what they desire of their community, and how the project might affect their quality of life. It is vital that this focus on inclusive decision making and governance be based on the needs of each specific community, and be applied to the planning and operation of water infrastructure. The current state of water supply and treatment planning exhibits extensive horizontal and vertical fragmentation (Lienert, Schnetzer, and Ingold, 2013). Fragmentation on the horizontal level is indicative of low integration or strong divisions among water supply and wastewater sectors (Lienert, Schnetzer, and Ingold, 2013). Vertical fragmentation is where there is little collaboration between actors at the national, regional, and local levels (Lienert, Schnetzer, and Ingold, 2013). This fragmentation in infrastructure planning is a result of greater power being situated locally, specifically among engineers and politics. This fragmentation, both horizontal and vertical, has, by and large, undermined long-term planning with regard to sustainability (Lienert, Schnetzer, and Ingold, 2013). In order to facilitate integrated catchment planning, (planning of both water supply systems and waste treatment systems) it is therefore vital that the totality of stakeholders be involved in understanding, organizing, and communicating in the planning and management of water infrastructure.

In the Global South, or countries that are often considered to be developing in lieu of absent economic and infrastructure capital, a community based planning approach has been shown to be very successful. In this method, basic infrastructure can be developed at the community or municipality level by the concerted effort of citizen planners (Darrundono and Mulyadi, 1991). A classic case study is the Kampung Improvement Programme (KIP) project that was started in Jakarta, Indonesia in 1969 and has expanded to hundreds of municipalities (Darrundono and Mulyadi, 1991). In this case, the state encouraged the local citizenship to take on the development of infrastructure within their communities. Municipality residents were in charge of determining what if any infrastructure was necessary for the community. In this case citizens are intimately involved in deciding where the infrastructure should be located and how the maintenance would be ensured by user fees (Darrundono and Mulyadi, 1991).

There have been other cases when planning has attempted to facilitate public collaboration and collaboration between professionals across local and national boundaries. The United Nations, nation states, and nongovernment organizations have tried to address the development of water infrastructure in communities that have insufficient basic sanitation and/or potable water supply (United Nations, 2009). The majority of population growth in the next century will occur in the Global South or places that already have inadequate sanitation potable water supply (United Nations, 2009). Poverty and education are two key social drivers that have been identified as being necessary for community involvement in the planning process (United Nations, 2009). In poverty settings people do what they have to do to survive regardless of the environmental impact. An educated populace is able to better understand the costs and benefits of different water technologies and appreciate the environmental benefits accrued. One interdisciplinary approach that has been adopted to help water managers allocate water effectively in some developing countries is the Water Poverty Index (WPI) (Garriga and Foguet, 2010). The WPI addresses the link between potable water stress or scarcity and the socioeconomic drivers of poverty and it can help professionals by educating them on how poverty and water scarcity interact.

One of the major challenges in developing new infrastructure has been the large cost of traditional wastewater treatment plants. The traditional way of financing a project has been through governments using tariffs, taxes, or transfers through philanthropic aid (in developing regions) (United Nations, 2009). With tariffs, individuals pay for the service they receive, which is problematic as clean water is and should be a human right, and often households lack the ability to pay in the Global South. In developing countries there are often severe fiscal constraints that inhibit their ability to adequately fund development using their limited tax revenue (United Nations, 2009). The use of non-local and/or international sources of funding for water infrastructure projects in developing nations has often been a road block for community inclusion in the planning process. Some policy developments that could help address this might include the establishment of organizations that finance at the decentralized level, increasing the role of small-scale local water providers, or instituting tariff reform with cost recovery apparatus (United Nations, 2009).

Potential Costs and Roadblocks to Inclusiveness

Like any planning practice, community driven planning *is* context specific. It depends on social factors including the size of the community, its homogeneity in both culture and economics, and how the state manifests itself in resident's everyday lives. The capacity of the community to accomplish any given task also rests in the physical parameters of where a community lives and what physical capital is manifested. The view point of many in the planning professions is that community based planning has the potential to cost-effectively create much needed community infrastructure by empowering the individual members of a community (Friedmann, 2011). It, however, also has the potentiality of unduly burdening already impoverished communities by shifting the traditional role of the state as provider of infrastructure (Friedmann, 2011).

Other critiques of community based planning is that its reliance on local human resources and consent means that the scale of projects can only be small and are nearly impossible to scale up. There are two main answers to the concern over the ability to scale up projects and these are that some projects work just fine at their scale of deployment and have no need to be scaled up. The other answer is that with increasing horizontal integration of communities within a region, such as the expansion of KIP in Jakarta to hundreds of cities throughout Indonesia, can create a network that is a “scaled-up” representation of the original project (Lienert, Schnetzer, and Ingold, 2013). Simply claiming that a community is so site specific that community based planning can’t work in all communities negates the *very* foundation of the practice requiring community specific responses to community specific needs.

Another critique of implementing community based planning is that it is constricted to use only on technologically simple projects both in their construction and operation. The level of technology can be *too complex* for community members to engage with (Park *et al*, 2009). Although it is true that not every person who is part of a community obtained an engineering or professional planning degree, it is possible for members of both professions to engage the community and facilitate the educated community decision of what technology should be employed. In the United States, currently, the building of infrastructure and the planning of our cities remains a very undemocratic proposition with decisions being made by professional staff and appointed planning commissioners (Berke *et al,* 2006; Hall, 2002).

A final critique is that in observing community run programs that are for the common interest it has been observed that absent some kind of external coercion, whether it is cultural, economic, or punitive, individuals will not act to achieve the common goal (Olson, 1965). A problem that can occur is the free-rider problem, where individuals refuse to contribute toward a collective good that they draw upon. In the case of water services this nonpayment can undermine the financial health of water utilities and can lead to greater consumption (Aguilar-Benitez and Saphores, 2008; Bithas, 2008; Browning-Aiken *et al,* 2007). However in the case of the KIP the community in Jakarta was able to construct the infrastructure they needed in spite of the potential limitations of community driven planning.

A common perception among engineers and policy makers is that the public can be resistant to available policies because they simply don't understand the technology involved (du Pisani, 2006; Schnoor, 2009; Stenekes *et al,* 2006). This belief, that the lack of public acceptance of technologies, such as direct potable reuse or indirect potable reuse, is due to the "ick" factor alone or due to some other misunderstanding of the technology, represents a missed opportunity for communities (Dolnicar and Hurlimann, 2009). Often the pushing of one technology over another can be a reflection of the efforts of vested interests in the profession to implement a technology that they have predetermined as the optimal solution (Stenekes *et al,* 2006). This thought that people need to be simply "educated" on the engineering solution that is "best" for them can prevent the open dialogue that a community can have in deciding what is best for it as a whole (Stenekes *et al,* 2006).

There have been some new planning paradigms that have not been widely adopted and which try to get the public to collaborate in the planning and development processes. This is particularly evident when policy makers have tried to get the public to get behind water and infrastructure programs that are controversial (Brown *et al*, 2008). This research is similar to the conception of the city as "metropolitan nature" where cities progress toward what some researchers call the terminal "Water Sensitive City" (Brown *et al*, 2008). This paradigm aims to assist water managers in the urban environment to understand that there exists a "hydro-social" framework within their community and that capacity development and community engagement are needed to expedite transitions of the city through the different types displayed in Figure 4.

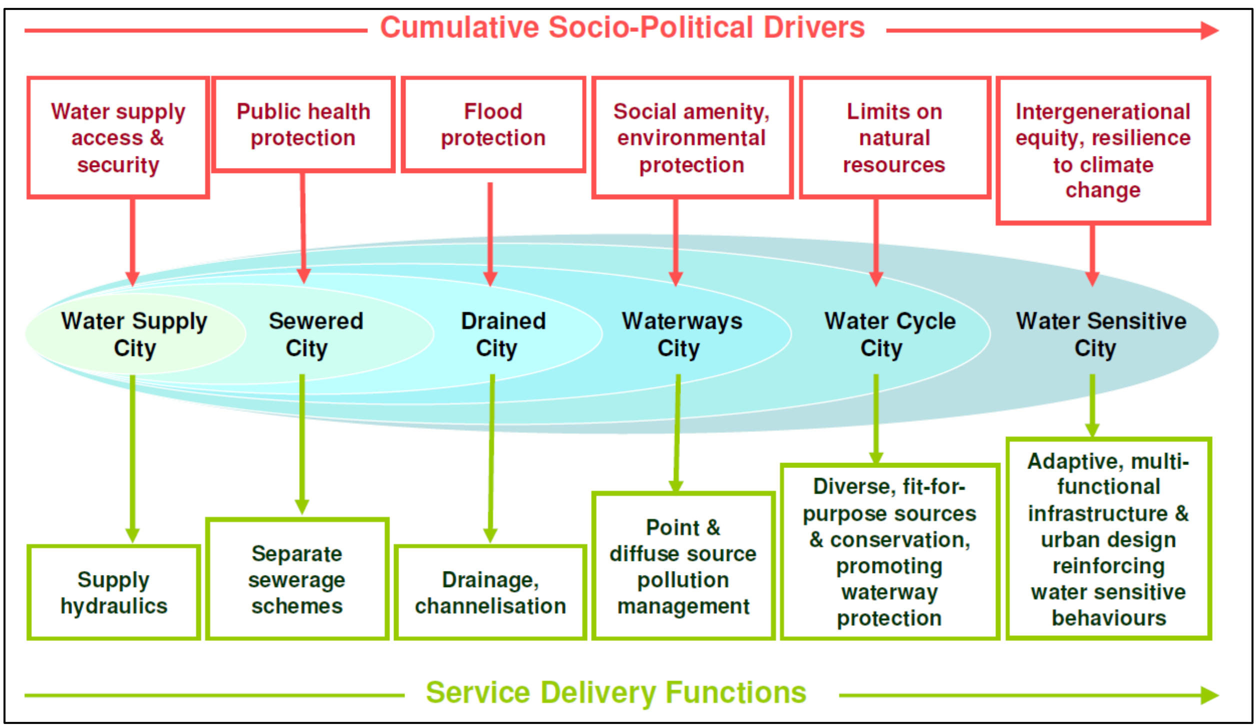

Figure 4: Urban Water Management Transitions Framework which conceptualizes as the city an entity that has a hydro-social framework. Thus the city goes through different transitions in response to socio-political drivers and water management responds in kind. This causes the city to progress toward a sustainable future (Brown *et al*, 2008).

There remains a thread within the planning and engineering fields which is resistant to the inclusion of the average community member into the process of planning. It is important to realize that there is right now throughout much of the world a *local planning monopoly*, where the professionals employed by municipalities have great control over the planning process (Gullstrand *et al*, 2003). Also the population size of a community is independent of the openness of a municipality's planning regulatory framework (Gullstrand *et al*, 2003). Ultimately the ability of the "scaling up" of community based planning of water infrastructure relies upon the ability of professionals with their technical expertise and community members to bridge the horizontal and vertical fragmentation when it comes to the planning of infrastructure.

Bridging Horizontal and Vertical Fragmentation

The bridging of the horizontal fragmentation of the water infrastructure community is ultimately the task of taking "distributed" processes of water conveyance, capture, public health, ecosystem health and addressing their process connectivity in a sustainable (or temporal) manner. Some of the research that has gone into trying to optimize regional water infrastructure has the potential to bridge horizontal fragmentation of water industry (Hyang *et al.,* 2013; Heusch *et al.,* 2010; Marropoulos *et al.*, 2008).

The bridging of the vertical fragmentation of the water infrastructure community is ultimately the task of seeing how to take complex systems across the spatial network that is composed of human communities at different contextual levels. Any given community member is a member of multiple communities across their life-span whether that be their family, their local municipality, and through regional, national and international communities. When community members interact with others at these various levels they practically overcome vertical fragmentation (Evans and Varma, 2009).

Future Water Available

A major problem with water availability in the future is that there is uncertainty of in its distribution. At the same time there is low interest in green technologies among private citizens, public officials, and private industry groups (Conte *et al.*, 2012). In order to better understand what infrastructure will be needed in the coming years, research has been performed that forecasts the site specific (regional) future demand for freshwater. In the case of the Abu Dhabi Emirate, the proposed solution for the expected scarcity of fresh water due to the changing abundance of water was to propose the regional commitment to creating the long-term storage capacity of freshwater that would meet demand for a year or greater (Al-Katheeri, 2008; United Nations, 2009).

Other research has attempted to create a formalized method for establishing a mass balance or total water cycle management system that quantifies all anthropogenic and natural water flows in an urban environment (Chanan and Woods, 2006; Duong *et al.*, 2011; Kenway, Gregory, and McMahon, 2011). In the preponderance of research that focuses on the future availability of water there is interest in integrated water management and how to optimize low-impact infrastructure (Burn, Maheepala, Sharma, 2012; Cunha *et al.*, 2009; Erbe and Schutze, 2005; Nhapi and Gijzch, 2005; van Roon, 2007; Wang *et al.*, 2006).

Infrastructure Efficiency

There are a host of techniques or methodologies that try to wholly capture the entire budget of water within urban communities. These range from methods that attempt to determine the productivity and efficiency of specific structures in the total water industry (Abbott and Cohen, 2009; Cardoso *et al.*, 2012; Lim *et al.*, 2010). Others include the quantitative benchmarking of treatment and disposal using influent characteristics, wastewater treatment alternatives, and location constraints (Aybar *et al*, 2007). A tremendous amount of research has gone into the assessment of integrated wastewater systems and how they affect the water supply over the life cycle of different types of technology applications (Munoz *et al,* 2010; Soares, Parkinson, Bernarders, 2005; Lundin, Bergtsson, Molander, 2000; Lundie, Peters, Beavis, 2005; Dixon *et al*, 2003; Foley *et al*, 2010). Some researchers call for integrated reuse systems where there is continuous monitoring of physical and chemical parameters throughout the urban landscape wherever reclaimed water was used (Hack and Wiese, 2006; Kalavrouziotis and Apostolopoulos, 2007; Rodriguez *et al.*, 2013; Schutze *et al.*, 2008; Sharma, Cook, Chong, 2013).

Others try to analyze environmental and cost effective performance of wastewater treatment plants and sewers through an integrated urban wastewater system by using individual sewer catchments that compose a total basin for a year time scale (Benedetti *et al*, 2008). There have been a number of analysis that model the cost-benefit relationship decentralized systems have in the water supply (Chen and Wang, 2009; Chung *et al.*, 2008; Schmitt and Huber, 2006).

Centralized v. Decentralized (and "Alternatives")

There has been a tremendous amount of work devoted toward comparing distributed (or satellite) treatment systems in substitution to centralized treatment of wastewater. This was done because for much of the time the choice of one over the other was simply deciding which was the more cost effective in meeting the community's required waste treatment standard.

One method that researchers have used to choose between centralized and distributed systems is by defining a set of sustainability indicators and assessing which option is more sustainable (Balkema *et al,* 2002). In evaluating sustainability a definition commonly used of sustainable development is that the development should bring the system to a more advanced/effective state and it should be able to keep that existence without diminishing it

(Bradley *et al*, 2002; Kerstens, de Mes, Lue, 2009; Hellstrom, Jeppsson, Karrman, 2000). Some others take social and economic criteria into their quantification of sustainability (Sahely, Kennedy, Adams, 2005). One of the chief arguments against urban growth at the peri-urban edge of existing communities is that it forced the construction of the infrastructure for a minority of individuals, thus causing disproportional harm via sprawl (Southerland, 2004). Sprawl is often seen as being very undesired by planners because communities run into inefficiencies in their infrastructure to user demand ratio. This viewpoint favored the continued growth around the existing centralized treatment systems or the use of the less intensive distributed systems to meet the peripheral demand (Maktropoulos and Butler, 2010; Massoud, Tarhini, Nasr, 2009; Woods *et al*, 2013).

One of the primary reasons why the engineering profession has in the past pushed for larger and fewer wastewater treatment plants or centralized systems has been due to perceived cost savings on capital investment due to economies of scale associated with centralized treatment (Hopkins, Xu, and Knaap, 2004). In the case of the metropolitan Chicago region there was a perception that a savings of $170 million USD in capital costs could be yielded were the region to pursue consolidation of wastewater treatment (Hopkins, Xu, and Knaap, 2004). In this case it was demonstrated that forcing development (through a regulatory framework) to spatially locate where existing capacity exists yields the highest savings (Hopkins, Xu, and Knaap, 2004).

In evaluating whether distributed systems are economically appropriate for a given community a metric that has been used is by use of a critical distance parameter (L_O) which is defined as the distance where the cost of construction of the pipeline connecting a community to centralized treatment is equal to the cost of construction decentralized systems (Wang *et al.*, 2008*)*. This cost of “infrastructure” is then combined with a operation and maintenance cost per unit of water treated in each case. The cost of construction of the pipeline to access the centralized treatment is dependent on linear distance of the pipeline (L) and the design flow (Q), while the cost of decentralized system installation only depends on the community's Q (Wang *et al.*,2008). If L is less than L_O then it is economically more efficient to incorporate the community's load through connecting to already existing centralized treatment capacity. If L is greater than L_O then it is more efficient to deploy the distributed systems (Wang *et al.,* 2008). This narrow spatial treatment of analysis has a plethora of physical factors that go into determining the cost of the distributed systems including: the cost of electricity, chemicals, labor salary, daily maintenance, major repairs, and annual value depreciation. What is key about this analysis, and others like it, is that it presupposes a system of mutual exclusivities, where either distributed infrastructure or centralized infrastructure is deployed, but not both. It is also based on a least cost optimization framework that dominates the majority of the literature, when discussing sewer and treatment expansion (Chang and Hernandez, 2008; Prihandrijanti, Malisie, Otterpohl 2008).

Some methods have been developed for evaluating which municipalities are *most open to or could benefit most* by using distributed treatment facilities (van Afferden *et al*, 2010). This spatial analysis determines locales based on their relative location in comparison to centralized systems and geographical features and it can be optimized (Dong, Zeng, Chen, 2012). Much of the research and practice around water infrastructure has focused on the modeling of the weighing of investment in either centralized or decentralized infrastructure (Wang *et al*, 2008; Nanninga *et al*, 2012). This dichotomy has also been criticized as being too simplistic. A viewpoint is that it doesn't adequately encapsulate the complexity of communities requiring mixed solutions to meet their water needs in the face of increasing water budget uncertainty (Spaargaren and Oosterveer, 2010).

Most recent studies have continued in this vein, studying the deployment of a mixture of onsite (distributed) treatment systems and their connection to centralized treatment system (Tsuzuki *et al*, 2013). Ultimately the benefits of centralized systems are associated with the economies of scale achieved through their use, but they also have large distances that separate any potential reuse water from their consumers (Bradley, 2002). The benefits of decentralized systems are that they can meet demand where it is cost prohibitive to connect to centralized systems, they decrease the water stress risks associated with global cycles, and they are less threatened by natural disasters or terrorist attacks (Gikas and Tchobanoglous, 2009; Tjandraatmadja *et al.*, 2013). Additionally, decentralized systems can be used to meet demand in rapidly developing cities where poor planning may have occurred (Weber, Cornel, Wagner, 20047; Strohschon *et al.*, 2013; Parkinson and Tayler, 2003; Suriyachan, Vitivattananon, Nurul Amin, 2012).

EXAMPLES OF THE POTENTIAL OF DECENTRALIZATION

Complete Decentralization and Reuse

Some research in Japan that excellently bridges the horizontal fragmentation of the water industry attempts to integrate the energy, water, and waste (EWW) cycles of communities (Anilir, Nelson, and Allen, 2008). This is in spite of the large regional treatment systems built to treat wastewater in the post war era (Brill and Nakamura, 1978), by using a life cycle assessment (LCA) where the objective is to, "completely close the solid, liquid and gas waste cycle of a house," and reuse or recycle all waste products (Anilir, Nelson, and Allen, 2008). In order to accomplish this they model over a 30 year period alternative integrated technology at the small community level (20 households) to see whether they can cut back on the need for centralized supply of resources to both urban and rural communities. The three scenarios they model are wastewater gardens, anaerobic digester gas system (ADGS), and a combined heat and power unit (CHP) (see Figure 5).

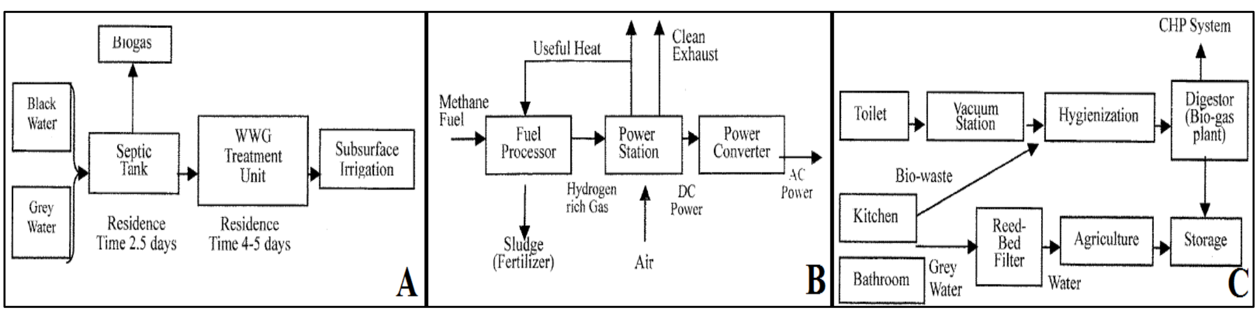

Figure 5: A is the wastewater gardens combined with biogas technology, B is the anaerobic digester gas system with integrate fuel cell technology, C is the combined heat and power unit with a bio-gas producing reed-bed system (Anilir, Nelson, and Allen, 2008).

In their analysis they were able to determine that a rural 20-community household composed of 50 residents was able to achieve equivalent treatment on-site (i.e. distributed) at between 25%-50% of the cost it would have taken for centralized treatment. Similarly the cost of these systems for an urban center of 5,000 to 100,000 residents would cost 33%-66% of what centralized treatment would cost (Anilir, Nelson, and Allen, 2008). Much of the savings in "cost" for this analysis lies in the concept of elimination of the concept of “waste” from the effluent systems and providing end products such as electricity, heat, nutrient rich irrigation, and potable water to consumers (Anilir, Nelson, and Allen, 2008). The main impediment to implementation is that the processes can be energy intensive, require high initial investment of resources, and require highly skilled maintenance staff or professional capital that many rural areas may lack (Anilir, Nelson, and Allen, 2008). The real unit of analysis for this project seems to be in unit of cost. This can be of great use as municipalities have to balance different costs when choosing their priorities. It could however be combined more with evaluating the desires of the members of *specific* communities as opposed to communities as bins of populations.

Centralized and Decentralized Combined Partial Reuse

There has been very recent research on the design of combined centralized and decentralized options for wastewater treatment and resource recovery. The key criteria that this case study uses in evaluating the quality of treatment design are the 1) initial investment cost, 2) operations and maintenance cost, 3) economic benefit from resource recovery, 4) net life cycle cost, 5) percent renewable energy (from the biogas use), and 6) resilience to water stress (fraction of water demand met by recycled water) (Lee *et al*, 2013). These criteria are then weighted and evaluated on a “better-than-others” performance standard rather than maximum performance in a single category. The goal for this is that it allows for the comparison of criteria that are monetary (USD for first four criterion) and non-monetary (resilience as a percentage in the final two criterion) (Lee *et al*, 2013).

This method (see Figure 6) could allow for specific communities to incorporate their own non-monetary criteria of that which they see as important and weigh them based on the specifics of their community. It should be noted that of the four system designs considered in this analysis, only energy and water recovery were evaluated (and not nutrients per se). This was done because the most economically beneficial resource recovered is water, due to the costs savings of needing to import less, or the revenue from exportation if possible. The second most beneficial resource is the energy savings from biogas usage (Lee *et al,* 2013). One of the main take away messages from this is that often the initial investment and operation and maintenance costs tend to go up when reuse plants are designed to recover a greater portion of resources, but that higher economic investment can be outweighed by the revenue and avoided costs of resource recovery (Lee *et al,* 2013). Iterative selection of indicators could be used in evaluating the environmental sustainability of the chosen urban water systems (Lundin and Morrison, 2002)

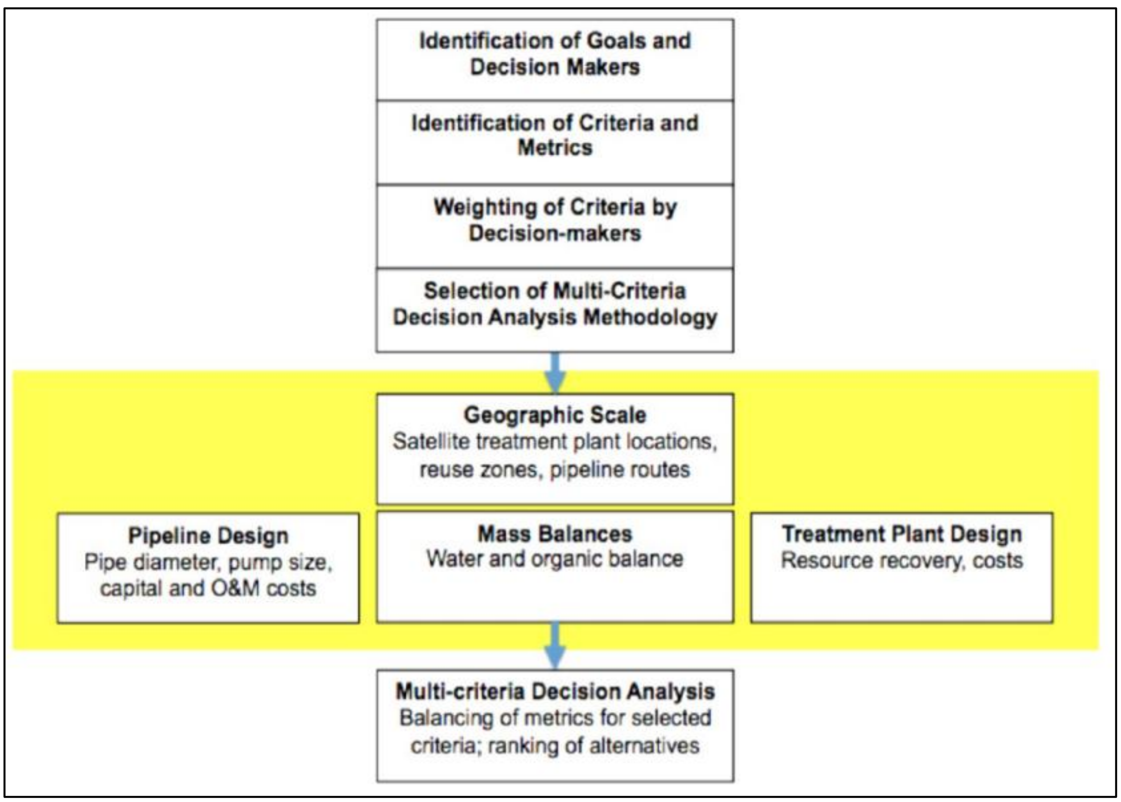

Figure 6: Outline of the steps of design of a resource recovery plant for different spatial scales. It should be noted that there are additional feedback loops that can be used for alternative designs (Lee *et al*, 2013).

High Community Involvement Reuse

There have been examples employed in the planning of distributed water infrastructure that have attempted to incorporate a participatory aspect where stakeholders were consulted through Future Workshops (Nanninga *et al.*, 2012). With this method, communities in the peri-urban outskirts of Mexico City evaluated different scenarios for regional development, evaluating existing problems and the technical feasibility of possible solutions (Nanninga *et al*., 2012). In this case, they tried to balance different stakeholders' desires with regard to economic development, environmental health, and maintaining local identity—desires that could in various ways be in either competition or in cooperation (Nanninga *et al.*, 2012).

From the three major goals of stakeholders listed above, key objectives and characteristics were defined. These objectives and characteristics were used in turn to identify treatment technologies that would facilitate the manner of use that would best accomplish their goals. The importance of this work, as stated by the researchers, is that it raises, "the awareness and interest of users and stakeholders to the issues at stake. Complex technical issues could successfully be discussed with a wide range of local users and stakeholders," (Nanninga *et al.,* 2012, 16). Fundamentally, this research asserts that decentralized treatment systems will require the acceptance of the technologies, not only in the tacit way that the community has embraced the vast majority of current water infrastructure. This is because the decentralized technology will be more evident to them, both through physical proximity, and potentially, through maintenance responsibilities being distributed more equally throughout the community. It has been observed that wastewater managers play a *crucial role* in the sustainability of these engineered systems (Cuppens, Smets, Wyseure, 2013). In the case of distributed infrastructure there is the potential that local residents will have to act in this capacity (Murray and Ray, 2010).

ENGINEERING TECHNOLOGY

Non-potable Water

The number one most sought after reuse product has been non-potable water reuse for one purpose or another. As mentioned previously (Lee *et al*, 2013) it has the greatest potential for savings in cost for a municipality due either to the diminishment in its use or due to the potential revenue that can be created by selling water. There is also the potential of using water treated to various levels of quality, which can be used for irrigation (and fertilization if nutrients are at a desired level). Included within this category are a broad set of low intensity, low energy water treatment techniques, including the use of constructed wetlands for reclaiming water and nutrients via harvesting aquatic vegetation grown in the wetlands (Greenway, 2004; Rousseau *et al*., 2008, Pastor *et al.*, 2003). In addition to the economic benefits that the constructed wetlands produce by removal of TSS, BOD, and NH_4-N they also provide aesthetic and environmental services to humans and animals respectively (Teng *et al*, 2012). One chief aspect of the reused water for urban landscape is that effluent water can be used in place of river water and can be sprayed on crops without increasing the risk to human health from pathogens (Brissaud *et al,* 2008; Dodic and Bizjak 2009; Fu, Butler, Khu, 2009).

However, one concern that has arisen is that irrigation long-term with solely treated effluent could potentially lead to the accumulation of trace contaminants (metals) in the upper levels of the soil applied on, potentially deteriorating the soil over time (Xu *et al,* 2010). In order to address this concern a Source Classification Framework to better model any potential trace contaminant pollution can be used (Lutzhoft *et al.,* 2012).

One of the most salient facts about the cost effectiveness of reclaimed water is that generally non-potable direct reuse is the most cost effective reuse of water, excluding inexpensive conventional sources like natural reservoirs or rivers (Ray, Kirshen, and Vogel, 2010). An additional source of non-potable water that is reused is by greywater recycling systems that reduce the quantity of water consumed by domestic users by as high as 40% (Al-Jayyousi, 2003; Henriques and Louis, 2011; Seng *et al*, 2009). An interesting application of this is how large skyscrapers could have on-site treatment and reuse of virtually all their water, potentially making buildings water neutral (Krogmann *et al*, 2007; Smith, 2009).

In addition to the low energy, low intensity treatment of water there are a great number of options for high intensity water treatment such as ultraviolet treatment, reverse osmosis, biofilm/membrane, membrane filtration, and adsorption. It is possible to create high-quality water with nutrients and organics using solely membrane filtration which can be used to increase crop yields (Ravazzini, van Nieuwenhuijzen, and vam der graaf, 2005; Hyun and Lee, 2009). It is also possible to mix and match natural treatment systems with the advanced technologies to produce the quality of water desired for specific applications (Brown, Jackson, Khalife, 2010; Sperlich *et al*, 2008).

Potable Water

There has been little application of direct potable reuse technology in urban infrastructure. This has in large part been due to a lack of trust or acceptance of direct potable water reuse technologies by the public or end-users (Grant, S.B. *et al.*, 2012). Those cases where reclamation and direct potable re-use are performed are seen as anomalous in nature and are also seen as being due to physical factors such as extreme aridity which have forced communities to pragmatic solutions. A seminal example of such a facility was the planning and construction of the Goreangab Water Reclamation plant for the City of Windhoek, Namibia in 1969 and in operation to date (du Pisani, 2006).

The case of the Goreangab Water Reclamation plant has three chief aspects that are of vital interest to this Thesis. The first is that the municipality was able to outflank public resistance to direct reuse by persistently informing the community of the importance of reuse through the media, newsletters, school visits to the plant, and active participation by elected representatives (du Pisani, 2006). The second was that in 1969, as a condition of securing a loan for the plant's construction from the European Investment Bank, there was a stipulation that operation and maintenance would need to include international, in this case private sector, direct involvement for 20 years (du Pisanai, 2006). This aspect of the plant highlights how Windhoek was able to overcome a human resources problem or a perceived lack of operational and technical expertise required to build advanced treatment facilities in "developing" countries (Parkinson and Tayler, 2003). The final, and perhaps most important aspect of the Goreangab Water Reclamation plant is that in order to meet demand of a growing population a new plant was commissioned and finished in 2002 which included the two new processes of ozonation and membrane ultrafiltration (see Figure 7). In the first year of operation the new plant produced 5,955,347 m^3 of high-quality water at an average cost of \$0.46 USD/m^3 for the treatment or \$0.77 USD/m^3 including capital costs. This compares incredibly favorably against water importation which is in excess of \$2.92 USD/m^3 with the value of ensuring security of supply included.

Although Goreangab Water Reclamation plant is the only truly successful large scale direct potable reuse plant in the world, there are multiple technologies that make direct potable reuse available in small-scale systems in the range of 1,000 L/day to 10,000 L/day (Peter-Varbanets *et al.*, 2009). These include but are not limited to reverse osmosis, microfiltration, and ultrafiltration.

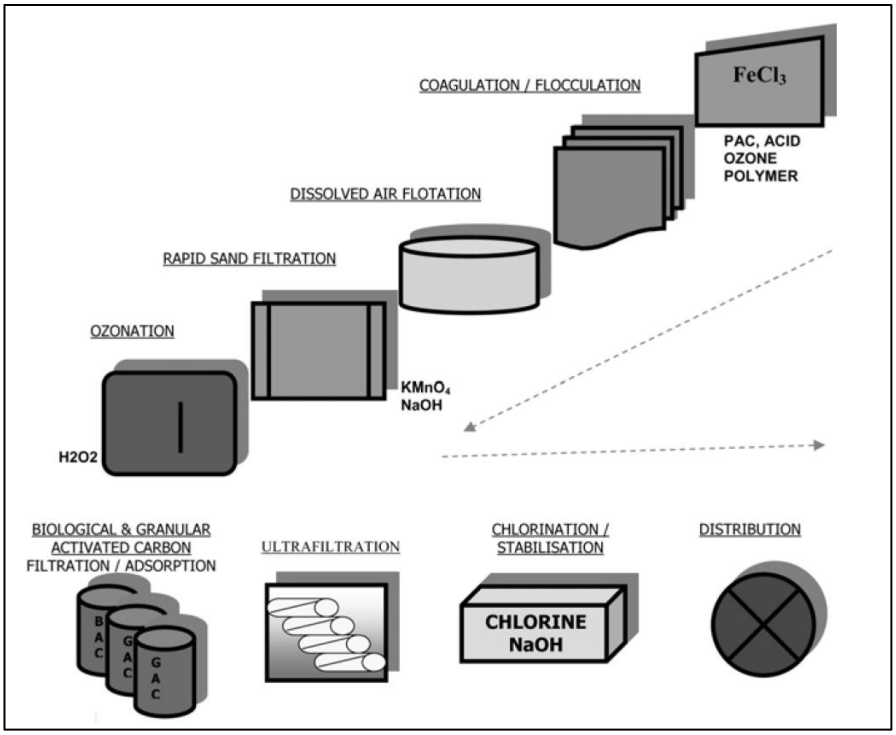

Figure 7: Example of treatment procedures used for direct potable reuse. Example is the process train of the new Goreangab Water Treatment plant (du Pisani *et al.*, 2006).

Energy and Economic Considerations in Water Treatment

A primary concern of reuse in treatment facilities has been the reduction in the footprint of energy used to treat water and the maximization of the amount recoverable from such sources as biogas production optimization (Assimakopoulos *et al.*, 2007). The design of early distributed effluent treatment systems was predicated on the idea that where there are multiple treatment processes required, and that the operator should seek to maximize the treatment load on the cheapest treatment processes (Wang and Smith, 1994). This design of distributed effluent treatment systems works by combining waste that requires the same processes thereby lowering costs and which would otherwise be done separate.

It is important to initially recognize that any net savings in water or nutrient usage directly correlates with increased economic savings as less water or nutrients need to be procured from off-site sources. In the case of distributed systems in the peri-urban setting this is readily apparent as the community does not need to pay as much for water treatment or nutrient removal by the centralized treatment facilities (Nanninga *et al.*, 2012).

As has been previously stated, there is substantial evidence that nonpayment for water services undermines the long term financial health of water utilities (Aguilar-Benitez and Saphores, 2008). Additionally, it has been observed that the usage of flat rates for water or a single cost regardless of amount of water consumed has been shown to disincentive conservation of water (Browning-Aiken *et al.*, 2007). There are other models of regulating the reuse and cost of water that encourage minimization of water usage. However in treatment facilities that focus on optimizing the facility for water reuse in agriculture there is a direct competition between increased profits and conservation of water (Murray and Ray, 2009).

In the coming decades there are expected to be increases in the amount of energy demanded for society to function. Typically this energy production requires water at one point or another in the production cycle and therefore an increase in energy demand requires a proportional increase in water demand. In the United States specifically this water demand is expected to scale up in a very unfavorable manner. The water consumption from energy-producing sectors within the United States is expected to increase by 70% between 2005 and 2030 (Elcock 2010). In comparison the total expected domestic water consumption is expected to increase by 7%, from 108.2 billion gallons per day in 2005 to 115.5 bgd in 2030 (Elcock, 2010). This shift, which is primarily due to increased water demand for biofuel production/irrigation, means that water for energy production will be equal to 10% of total water demand in the United States in 2030 compared to 6% in 2005 (Elcock, 2010). The use of water for irrigation in biofuel production will represent half of the water required for energy production or 5% of the total water demand in the country (Elcock, 2010).

Table 1: United States Water Consumption by Sectors, 2005-2030. All values in billion gallons per day (Elcock 2010).

Sector	2005	2010	2015	2020	2025	2030
Irrigation (other than biofuels)	80.0	79.5	79.2	78.9	78.7	78.4
Domestic and public	7.3	7.6	7.9	8.2	8.5	8.8
Thermoelectric	6.1	6.3	6.7	7.2	7.7	8.2
Energy production (includes biofuels)	6.1	9.1	9.7	10.2	10.2	10.2
Industrial and commercial	5.3	5.3	5.3	5.3	5.4	5.5
Livestock	3.5	3.7	3.8	4.0	4.1	4.3
Total	108.2	111.5	112.7	113.8	114.6	115.5

This increased demand could be heavily mitigated with the repeated use of distributed systems to recycle both the water and nutrients. In certain cases the cost neutral recovery from municipal wastewater in the form of treated grey water can have up to 75% efficiency (Nanninga *et al.*, 2012). This 75% of water from domestic sources could substantially augment the demand of water for energy production as 75% of the domestic and public demand in 2030 would be 6.6 bgd, while the demand for energy production is expected to be 10.2 bgd in 2030. This means that for certain municipalities, the exportation of unused grey water from reclamation could be used for irrigation of agriculture for biofuel production. One of the true limiting factors for this is how to make the cost of greywater competitive with the cost of water that would normally be used in energy production.

One of the chief facts to bear in mind when considering energy expenditures and savings associated with choosing one treatment technology or another can vary greatly for any number of reasons (see Figure 8). It is vital that deciding on one technology over another be based on the designed physical and social characteristics at the intended site of treatment (Rygaard, Binning, Albrechtsen, 2011).

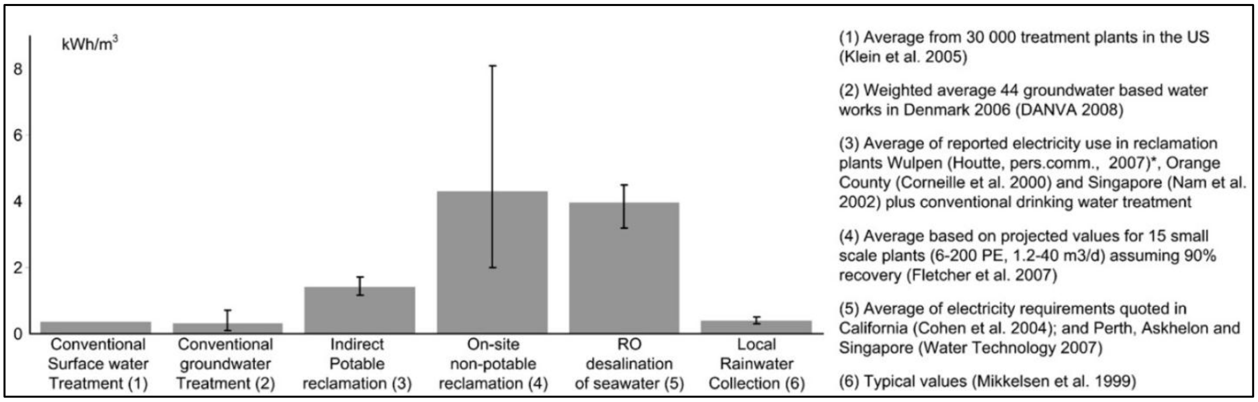

Figure 8: Electricity demand per unit m^3 of water produced by different treatments. It should be noted that the quality of each m^3 of water also varies and consequently so does their value (Rygaard, Binning, Albrechtsen, 2011).

Nutrient Reclamation

There is a tremendous amount of nutrients, both nitrogen and phosphorus embedded within domestic wastewater that could be potentially harvested. Within wastewater the mean daily contribution per capita in the United States is 10 g of Nitrogen (70% of which is embedded within urine) and 1.5 g of Phosphorus (60% of which is embedded in urine) (Phillips *et al.*, 2011). This source of resources, as well as its concentration in an easily separable form, allow for several different alternatives for nutrient recovery which are already available on the market. There are *many* processes that have been demonstrated as having the capability of removing and/or recovering these nutrients from wastewater (Berndtsson 2006; Bernhardt *et al.*, 2008; Bolzonella *et al.*, 2006; Chon, KyongShon, and Cho, 2010; Kinniburgh and Barnett, 2010; Kumar *et al.*, 2007; Phillips *et al.*, 2011; Samori *et al,* 2013). Some offer other services for other than simple removal of nutrients, such as algal biomass production for a low cost biofuel (Samori *et al*, 2013). However, those processes that are widely deployed for removal or deployed successfully for removal and recovery are a minority of the total, and reflect the state of the industry.

The state of the industry makes it cost ineffective to recover N (ammonia) from municipal wastewater and compete with the market value (Phillips *et al*, 2011). The primary prospect for nitrogen recovery is from the sludge stream and there is little prospect in recovering from the main stream (Phillips *et al*, 2011). Recovery of nitrogen can be accomplished via urine separation and used locally as a fertilizer in agriculture and landscaping settings (Phillips *et al*, 2011). But ultimately the current production of nitrogen fertilizer is dependent mostly on the natural gas and the cost of power, equivalent to 12 kWh per kg of ammonia-N (Phillips *et al*, 2011). As there have been substantial finds of natural gas within the contiguous United States it is likely that the supply will not be unduly affected in the normal planning horizon or 20 years to 50 years (Elcock 2010). This makes the recovery of nitrogen for exportation an unlikely possibility in the foreseeable future unless some other manner of recovery is developed. Nitrogen *can* still be recovered by urine separation which can provide a simple fertilizer for communities unable to afford to purchase nitrogen fertilizer from producers. This is especially true when one considers the cost of farm urea fertilizer (see Table 2).

The possibilities for phosphorus removal and recovery *are* economically viable. This is due to a combination of factors including the cost of phosphorus mining and the technology available for extraction (Phillips *et al.*, 2011). There is not at this time a scarcity of phosphorus reserves globally, but at the current level of projected usage, the known reserves are expected to be exhausted on the scale of one to two centuries (Kinniburgh and Barnett, 2010; Phillips *et al.*, 2011; Wallis-Lage *et al*, 2011). This potentiality makes it prudent for planners and communities to take steps to design systems that can recapture phosphorus rather than wasting it with no gain to the human or biological community. In planning for developing countries where the cost of fertilizer might be prohibitively expensive, it is important to model the link between sanitation, resource recovery of nutrients, and their effects on the agricultural production of communities (Meizinger, Kroger, and Otterpohl, 2009).

Table 2: Removal and/or capture costs for nitrogen render it economically prohibitive for exportation given the current value of nitrogen fertilizer. The cost per m^3 is based off a 10 g nitrogen daily production per capita and 0.38 m^3 (100 gallons) daily water usage per capita. The Cost ($/$m^3$) assumes total nitrogen removal (Phillips *et al.*, 2011)

Removal and/or Capture Process of Nitrogen	Cost ($/kg N)	Cost ($/$m^3$)
[1] Deammonification (Anammox)	0.15	0.0040
[2] Nitrification/Denitrification (Sharon®):	0.9	0.0238
[3] Cost of Farm Urea Fertilizer:	1.09	0.0288
[4] Mainstream Nitrification/Denitrification:	1.8	0.0476
[5] Cost of Urea Ferilizer:	4.5	0.1190
[6] CASTion Ammonia Recovery:	6.72	0.1778

Table 3: Other technologies for the removal and recapture of phosphorus and nitrogen (Bolzonella *et al.*, 2006; Chon *et al*, 2012; Kumar *et al.*, 2007; Phillips *et al.*, 2011)

Other Technologies	Recover and Remove excess	Remove
Air Stripping	Nitrogen (as NH_4NO_3)	-
Asahi	Phosphorus	-
Aquaculture	Phosphorus, Nitrogen	-
CASTion	Nitrogen (as $(NH_4)_2SO_4$)	-
Composting	Phosphorus	-
Crystallactor	Phosphorus (struvite)	Nitrogen
Incineration of biosolids	Phosphorus (with more processes)	-
Integrated Membrane System	Phosphorus (struvite)	-
Land Application of Biosolids	Phosphorus (apply at rate of uptake)	-
Ostara	Phosphorus (struvite)	Nitrogen
Phospaq	Sulfur, Phosphorus (struvite)	Nitrogen
Phosnix	Phosphorus, Nitrogen	-
Spray Irrigation of Effluent	Phosphorus, Nitrogen	-

METHODOLOGY OF FUTURE RESEARCH

Scenario Definition

BioWin is a wastewater treatment process simulator developed by EnviroSim that models biological, chemical, and physical processes. Using this platform it is possible to effectively model the "whole plant" or centralized wastewater treatment plant that engineers have been designing for communities. The use of this platform allows for the comparison of process efficiency given different treatment design choices and dynamic influent of oxygen demand, nitrogen, and phosphorus into different plant scenarios.

The two main scenario types are whether the processes of water treatment, nutrient reclamation, and energy reclamation are pursued in centralized systems or distributed systems. In effect, this centralized scenario and distributed scenario each describe the physical design of water treatment in a community and will each be modeled using the BioWin process simulator. Additionally, a null scenario where no treatment is pursued for a community can be modeled to act as a baseline against which the distributed and centralized systems can be evaluated. In creating this model, the removal and capture processes for nitrogen, phosphorus, and energy described in the above sections can each be placed in each scenario creating different configurations of distributed and centralized systems. Modeling each of these configurations for the two scenarios and comparing the scenarios against each other (as well as the null scenario) will allow for a normalized comparison of the different treatment technology available to communities. It will help a municipality or regional authority to evaluate how implementing certain reuse technologies at varying degrees of scale can lead to a certain level of environmental impact per unit of energy.

In order to evaluate the fidelity of this modelling of the different scenarios of centralized and distributed water treatment configurations it is vital that it be compared against real world treatment facilities that exist. Therefore the model will be compared against existing centralized treatment facilities that are in Orange County, CA and newly created distributed wastewater treatment facilities that have been built in Victoria, Australia that were built in response to the severe drought that the country faced. Victoria has outlined for the Melbourne Metropolitan a whole-of –water-cycle through their *Living Victoria* framework. It provides excellent indicators for evaluating local and regional water outcomes and benefits and provides a good real world example of how a community can be solicited in the planning phase (Living Victoria, 2014)

This study will provide a tool that professionals can use to see how different technology selection can abate peaks of power usage and minimize costs for their given dynamic conditions. Ultimately this model is an energy platform (which can be translated to economic cost) which will model configurations so as to provide the ideal configuration for the community. The conversion from energy to total cost is discussed below.

Life Cycle Assessment and Cost Analysis

Life Cycle Assessment (LCA) as a methodology has been applied to many fields of study in order to determine the sum total of inputs and the sum total of outputs that a system experiences (Azapagic 1999). Early application of life cycle assessment procedures were devoted to the selection of the indicators or variables that are most salient to understanding the life cycle of water systems in the urban environment (Lundin and Morrison, 2002). This methodology has been applied to understanding the management of both large scale wastewater treatment (Fagan *et* al, 2010) and of smaller scale distributed systems (Benetto *et al*, 2009). It is vital that this model evaluates the cost of each component that goes into water treatment infrastructure as they are needed to determine the total cost that a community will sustain. It can be a means of showing that the community is in fact achieving a net positive benefit when all reuse and recycling benefits are taken into account.

A common means used in the past to evaluate the cost of a wastewater treatment facility has been to evaluate the variable operating costs and the capital costs of constructing the wastewater treatment plant (Fagan *et al*, 2010; Brunner and Starkl, 2012; Devesa *et al*, 2009). These were priced out per population equivalent of waste treated or as a flow rate of per day and the least expensive was often deemed more efficient. The variable operating costs calculation would include such things as the costs of purchasing of energy and reagents, waste disposal, labor, and maintenance (Fagan *et al*, 2010; Benetto *et al*, 2009). The capital costs would include such things as infrastructure design, construction, and decommissioning costs (Fagan *et al*, 2010). A major concern highlighted in determining the true economic cost of these treatment options has been the uncertainty that has existed in the quantification of econometric variables and benchmarking (Benedetti *et al*, 2008; Corominas *et al*¸ 2013a; Corominas *et al*, 2013b; Wittmaier *et al*, 2009; Renous *et al*, 2008).

Many past life cycle assessment and econometric studies take this approach toward the "economic cost" of a wastewater treatment plant. They simplify (though the calculation are by no means simple) the economic cost as the operating costs of the facility combined with the capital costs of construction (and decommissioning in some cases). Calculated separately, often using different indices, are the costs associated with greenhouse gases (kg CO_2equiv), environmental impacts and services (ex. Eco-indicator'95 Pts), or the benefits from producing energy, nitrogen, and phosphorus on site (Dixon *et al*, 2003; Verstraete *et al*, 2009; Rietveld *et al*, 2009; Fagan *et al*, 2010).

In very many ways when treating water greenhouse emissions, energy, and cost can be interconverted as the removal of waste or extraction of resources in water reuse require energy which produces greenhouse gas emissions at a certain cost (Rosso and Stenstrom, 2008; Bani Shahabadi *et al*, 2010). The reason why the calculations are often done separately is due to the uncertainty in econometric (see above) and greenhouse gas emissions variables (Sweetapple *et* al, 2013; Rodriguez-Garcia *et al*, 2012). However, it is vital that the true cost of these infrastructure choices be evaluated as the decision making criteria for water management professionals can be complex as it stands (Brunner and Starkl, 2004; Starkl *et al*, 2009; van de Meene *et al*, 2011). In order to incorporate all members of a community in the decision making process the cost efficiency value of all infrastructure alternatives should be available. Therefore for each configuration of each of the scenarios modeled above the cost of the infrastructure will be determined using a modified cost estimate (from Fagan *et al*, 2010) that evaluates the total economic costs and benefits of greenhouse emissions, energy, and the production of clean water, nitrogen, and phosphorus. This value in cost or benefit per population equivalent can then be used by planners and engineers in making infrastructure choice easier to understand by members of their community (Brown, 2007).

The original equations (again from Fagan *et al*, 2010) used to evaluate the total costs and converted into an hourly equivalent based on the expected lifetime of the infrastructure are displayed below as Equation 1 for the operating costs and Equation 2 for the capital costs.

Equation 1: $P = P_0 + \sum_u k_{PF,u} * F_u * n_u$ (operating costs)
Equation 2: $p = p_0 + \sum_u p_u n_u$ (capital costs)

In the above equations P is defined as the total operation cost ($/hr), P_0 is the initial operating cost ($/hr), $k_{PF,u}$ is the constant gain over time of the operation cost per unit flowrate for each unit operation u ($/hr), F is the volumetric flowrate of the influent (ML/hr), and n is defined as the number of units for each unit operation (Fagan *et al*, 2010). The equation for capital costs uses the same definitions as those for operational costs but the p is the capital cost in hourly equivalent of operation for all the life cycle phases of the infrastructure ($/hr).

An example of a unit operation that can vary the total cost is using different nitrogen removal and capture processes. In Table 2 the cost of removing nitrogen displayed is the average industry wide combined operating cost and capital cost that is required for each process listed. Each of these would constitute a unit operation (u) in Equations 1 and 2. As these listed prices are indicative of the average cost of removal and/or capture at both centralized and decentralized levels it is vital that evaluations of the cost of unit operations be modulated by the cost difference that each process may have at the centralized or distributed level. This can render a more accurate evaluation of the cost as treatment facilities take advantage of economies of scale and novel technologies.

Each of these operations can be cycled through, as can the currently available technologies for nitrogen removal, water reuse, and energy savings in order to produce a compendium that evaluates different technologies for communities to select. However, in effect calculations using the equations as they stand produce an incomplete economic impact evaluation as they do not take into account to cost carried by conventional environmental degradation or those of greenhouse gas effects. Therefore it is vital that all future calculations take these considerations into account so as to best serve the public good.

There is an expected cost for emitting carbon in the coming years given probable federal regulatory legislation that will hold people responsible for their emissions (Luckow *et al*, 2014). A mid-range forecasted price for carbon is $15 USD/ton CO_2 in the year 2020 and $60 USD/ton CO_2 in the year 2040 (all in 2012 USD). This means that the Equations 3 and 4 (from Fagan *et* *al*¸ 2010), which are used to determine the cost of greenhouse gas emissions in units of mass $CO_{2\text{-equivalent}}$ emitted from operating costs and capital costs respectively, can incorporated into the total cost.

Equation 3: $G = G_0 + \sum_u k_{GF,u} * F_u * n_u$ (operating costs)
Equation 4: $g = g_0 + \sum_u g_u n_u$ (capital costs)

In Equations 3 and 4 the G is defined as the total operation greenhouse gas emissions (in mass $CO_{2\text{-equivalent}}$/hr), G_0 is the initial operating greenhouse gas emissions (in mass $CO_{2\text{-equivalent}}$/hr), $k_{GF,u}$ is the constant gain over time of the operation greenhouse gas emissions per unit flowrate for each unit operation u (in mass $CO_{2\text{-equivalent}}$/hr), F is the volumetric flowrate of the influent (ML/hr), and n is defined as the number of units for each unit operation (Fagan *et al*, 2010). The equation for greenhouse gas emissions from the infrastructure creation uses the same definitions as those for operational greenhouse emissions but the g is the capital greenhouse gas emissions in hourly equivalent of operation for all the life cycle phases of the infrastructure (in mass $CO_{2\text{-equivalent}}$/hr). It should be noted that in other literature (Fagan, *et al*, 2010) the emissions in units of mass $CO_{2\text{-equivalent}}$ are converted to Eco-indicator'95 Pts/hr. Instead the mass of mass $CO_{2\text{-equivalent}}$ should be converted into the uniform unit of account for community decision making their infrastructure decisions.

A sample calculation that models different configurations of infrastructure centralization is demonstrated in Tables 4 and 5. In Table 4 a comparison between the typical costs of electricity associated with each treatment type is compared against the cost of the technology deployed. The cost of choosing different technologies for nitrogen removal and reclamation is then incorporated into the model in Table 5. This very basic version of the model will be expanded upon to include both the carbon and environmental costs in subsequent research.

Table 4: Evaluation of the cost electricity and infrastructure by treating water at different levels of centralization. It should be noted that the electricity demand ($Cost_E$) both in kWh/m^3 and $\$/m^3$ are based off the values in Figure 8 and the minimum and maximum cost of technologies (capital costs $Cost_C$) were adapted from (Rygaard, Binning, Albrechtsen, 2011). The Total Cost ($Cost_T$) is the summation of these capital and operational costs. The C/E ratio is the ratio of capital costs divided by the electricity (operating) costs.

Water Production by Treatment Type	$Cost_E$ (kWh/m^3)	$Cost_E$ ($\$/m^3$)	$Cost_{C,min}$ ($\$/m^3$)	$Cost_{C,max}$ ($\$/m^3$)	$Cost_{Tmin}$ ($\$/m^3$)	$Cost_{Tmax}$ ($\$/m^3$)	C/E Ratio
[A] Local Rainwater Collection	0.40	0.04	0.10	1.10	0.14	1.14	2.5-27.5
[B] Conventional Groundwater Treatment	0.30	0.03	0.20	1.20	0.23	1.23	6.7-40
[C] Conventional Surface Water Treatment	0.40	0.04	0.10	2.20	0.14	2.24	2.5-55
[D] Indirect Potable Reclamation	1.50	0.15	1.30	2.00	1.45	2.15	8.7-13.3
[E] RO desalination of seawater	3.60	0.36	0.90	2.20	1.26	2.56	2.5-6.1
[F] On-site non-potable reclamation	4.00	0.40	1.30	4.20	1.70	4.60	3.3-10.5

Table 5: The scenario described combines the different total cost values ($\$/m^3$) obtained for different levels of water treatment centralization with the cost of different ammonia removal/reclamation processes (from Table 2).

	Water Production by Treatment Type	[1]	[2]	[3]	[4]	[5]	[6]
$Cost_{Tmin}$ ($\$/m^3$)	[A] Local Rainwater Collection	0.14	0.16	0.17	0.19	0.26	0.32
	[B] Conventional Groundwater Treatment	0.23	0.25	0.26	0.28	0.35	0.41
	[C] Conventional Surface Water Treatment	0.14	0.16	0.17	0.19	0.26	0.32
	[D] Indirect Potable Reclamation	1.45	1.47	1.48	1.50	1.57	1.63
	[E] RO desalination of seawater	1.26	1.28	1.29	1.31	1.38	1.44
	[F] On-site non-potable reclamation	1.70	1.72	1.73	1.75	1.82	1.88
$Cost_{Tmax}$ ($\$/m^3$)	[A] Local Rainwater Collection	1.14	1.16	1.17	1.19	1.26	1.32
	[B] Conventional Groundwater Treatment	1.23	1.25	1.26	1.28	1.35	1.41
	[C] Conventional Surface Water Treatment	2.24	2.26	2.27	2.29	2.36	2.42
	[D] Indirect Potable Reclamation	2.15	2.17	2.18	2.20	2.27	2.33
	[E] RO desalination of seawater	2.56	2.58	2.59	2.61	2.68	2.74
	[F] On-site non-potable reclamation	4.60	4.62	4.63	4.65	4.72	4.78

CONCLUSIONS

The professional communities of planners and engineers involved in water infrastructure over more than a century have done a tremendous job by any measure of bettering the public health and quality of life of communities they have acted on their behalf. Improved drinking water is supplied to 87% of the world's population. The planning profession began in the United States in response to the lack of sanitation and its threat to public health. The sanitation engineers and proto-planners of the day built the gravity fed sewer systems that still exist throughout much of the country to remove the waste of humans. With the growing awareness of the damage that concentrated untreated human waste could cause to the ecosystems, there was an increased effort placed on cleaning up water prior to discharge. In prior eras energy was not seen as the limiting constraint and the idea of a wholly exhaustible resource was foreign to engineers. The rise in the cost of energy over the decades has changed how we design, as it reflects this new reality of scarcity. The spectre of both the cost of water and of phosphorus increasing due to eventual depletion in the latter and due to increased uncertainty in the former should likewise influence how we build the vital systems that sustain our quality of life.

The new emphasis in the design and planning of water infrastructure has been in utilizing resources to their maximum potential—often evaluated in terms of cost. What amount of water can be recycled and how much money can be saved (or made) by not having to import as much (or by being able to export more). The new goal for water infrastructure is the reuse of all things that are currently wasted. However, choosing one reuse process over another one ends up maximizing the reuse of the chosen at the expense of the other. Does a community decide that potable water reuse is its first priority as was done by Windhoek, Namibia? Or is the cost of nitrogen and phosphorus fertilizer outside the means of an agricultural community to purchase? In an ideal world engineers and planners would be the perfect diagnosticians of the physical needs of communities; able to determine exactly what a community desires or needs and tailoring the infrastructure so it can have the highest level of health and well-being. Unfortunately, the history of planning and engineering is filled to the brim with failed attempts at forcing communities to live, work, and play in one fashion or another. This has been perceived as a *benign* paternalism that persists to this day in professional planning with many communities having their communities planned out by appointed environmental commissioners, planning staff, and engineers who act as technical specialists.

There are other ways of planning communities and building infrastructure that can be sanctioned by the state. Ultimately the distinguishing aspect of the community driven planning discussed in this paper is that it is nominally democratic in nature. It allows for a site specific evaluation of what *communities members* want in their community. The prevailing theory of public participation in engineering has been in educating the public on the appropriate solution. All too often the appropriate solutions are those that are pushed by vested interests in the field whether it is for economic reasons or ease of use reasons and not based on the fact that the solution is the best fit for a community. The work outlined by Victoria, Australia though their *Living Victoria* strategy is a fabulous example of a municipality soliciting the public for submissions so that the public may educate engineers and planners on what they hold to be important.

In the past, larger centralized water treatment facilities were seen as the way to go because they took advantage of economies of scale in treatment processes and allowed for the concentration of an undesired land use (neighbor wise) away from the rest of the community. In contrast, the field of distributed infrastructure was viewed as the low cost, low energy, alternative that could be used in lieu of the cost of connecting distal communities to centralized treatment. Therefore, distributed treatment was seen as involving technologies such as rain barrels, constructed wetlands, some filtration, and greywater reuse. The argument for distributed systems was that they could *meet the water treatment demand* of communities either at equivalent or lower energy, water, and monetary cost. It has been proposed that the new centralized wastewater treatment plants of the future be called water reuse plants, and that they could be run on a business model. It is perhaps even more important that this paradigm shift be extended to distributed systems. The new paradigm would allow for the individuals in communities to take charge of the decision making process when it comes to how energy, water, and nutrients are managed in their community.

Future research will include the use of a process simulator to model both a centralized water reuse scenario and a distributed water reuse scenario. Both scenarios will also be compared against a null scenario (where no treatment is performed). The two scenarios will use the water treatment and resource reclamation technologies discussed in this paper. The technologies used will vary across iterations of the centralized and distributed scenarios. The process simulator will model the energy and mass transfer through each system. This will be combined with a life cycle assessment approach that normalizes the total economic and environmental costs and benefits of each scenario into a single economic utility function. The accuracy of the model will be tested by using it to model both a centralized water treatment facility in Orange County, CA and a distributed water treatment network that exists in Australia.

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