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The role of Green Infrastructure in Water, Energy and Food Security in Latin America and the Caribbean

Experiences, Opportunities and Challenges

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THE ROLE OF GREEN INFRASTRUCTURE IN WATER, ENERGY AND FOOD SECURITY IN LATIN AMERICA AND THE CARIBBEAN:

EXPERIENCES, OPPORTUNITIES AND CHALLENGES

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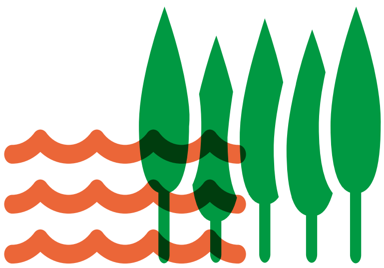
ABSTRACT

The nexus of water, energy and food encompasses the three interactive factors that are vital to both human communities and ecosystems. Traditional engineering approaches (known as *gray infrastructure* because they typically involve concrete) are unable to meet the challenges to nexus sustainability posed by evolving human demographics and climate change in Latin America and the Caribbean (LAC) due to planning and implementation time constraints. On the other hand, *green infrastructure*, utilizing ecosystems or their structural and functional components, can be implemented rapidly and provide a cost-effective treatment equivalent to gray infrastructure. Green infrastructure has historically focused on constructed wetlands for wastewater treatment, but has recently evolved into a broader approach involving multiple, integrative technologies to address urban sustainability in water, energy and food. With changing water availability across the region due to climate change and with growing demand impacting overall water resources downstream, increased attention has been paid to its conservation, storage and reuse based on green infrastructure. While full implementation at both community and governmental levels poses major challenges, there is no question that there is increased recognition of the economic importance of green infrastructure projects –progressively more developed as hybrid systems with gray infrastructure– to local economies and nexus security. Biomes are proposed as fundamental units for planning and implementing green infrastructure.



INTRODUCTION

The three dominant factors structuring human communities globally are often linked within the context of the nexus of water, energy and food (Figure 1). The nexus is a versatile means of adjusting each factor to meet changes in human demographics, land use, climate and water availability in the face of shifting water resources (temporally and spatially) and increased sectoral demand. It is important to determine how much, when and for how long water is needed for each sector to address current conditions and develop adaptive management plans that can respond to both short and long term projections and extreme weather events to sustain equitable water distribution to meet human and natural ecosystems' needs.

The United Nations University began to push for the integration of water-food, water-energy and food-energy into landscape management in the early 1980s (Scott et al., 2015), but it was not until the early 21st Century that formal integration of these sectors into the water-energy-food (WEF) nexus was proposed at a series of international meetings, including the World Economic of 2011 and the Bonn 2011 Nexus Conference: The Water Energy and Food Security Nexus (Leck et al., 2015, Hoff, 2011, Simpson & Jewitt, 2019). Justification of a focus on the WEF nexus follows three general lines: 1) increasing resource interlinks due to growing scarcities, 2) recent resource supply crises and 3) failures of sector-driven management strategies (Al-Saidi & Elagib, 2017). Yet, among the more than 300 nexus publications between 2009 and 2017, there is no consistent view of integration within the nexus.

Given that water and energy resource utilization by societies has not considered awareness of scarcity or value (Allan et al., 2015), there is a critical need to develop universal metrics that incorporate societal values at different geographical and temporal scales as part of WEF analyses (Tevar et al., 2016, Miralles-Wilhelm, 2016). An extensive literature review by Albrecht et al. (2018) noted that: 1) reproducible methods for nexus assessment are rare, 2) methods fail to capture the linkages within WEF, 3) most methods are quantitative, 4) use of social sciences methodologies is limited, and 5) most methodologies are confined to disciplinary silos and narrow in scope.

Faced with increasing populations and fast urbanization, diminishing resources and climate change, Miralles-Wilhelm (2014) proposed WEF as the best approach for tackling environmental, economic and societal sustainability in LAC based on the following factors:

1. Water is needed for food production. Approximately 90% of LAC agriculture is rain fed.
2. Water is needed for energy generation. Hydropower supplies 46% of LAC electricity.
3. Energy is needed for food production. Poorly understood but a significant issue.
4. Energy is needed for access to water sources. Desalinization is important in the Caribbean.
5. LAC is a net water exporter. Water footprints vary widely throughout LAC.

Miralles-Wilhelm (2016) proposed a research agenda for WEF nexus planning in the region based on integrative modeling tools with an interdisciplinary approach that can be tailored over different geographical areas and temporal scales. Such an approach must incorporate economic and social trade-offs among competing WEB priorities.

The current nexus approach identifies human health as the common linkage of the three traditional components –water, energy and food– and circumscribes them all within the framework of economics. This recognizes that humans (individuals and communities) are part of the ecology, and that any potential bias for one of the three traditional components is largely dictated by both availability and economics. In turn, policy is largely driven by human health and economics.

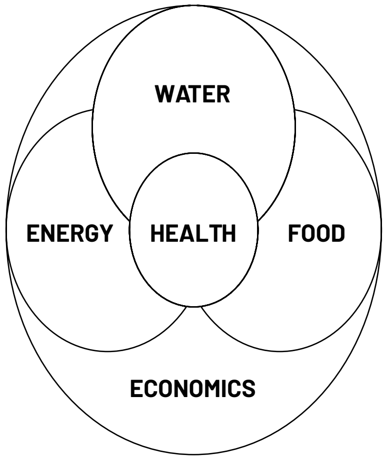


Figure 1. Nexus approach to environmental management recognizing the overarching importance of human health and economics.

Latin America and the Caribbean is experiencing landscape alteration, urbanization, and climate change faster than infrastructure can be built or modified to address them. Engineered solutions (gray infrastructure) require lengthy periods for planning and implementation, are expensive to build and maintain, are difficult to modify to meet changing conditions, and usually have a finite operational lifetime. In contrast, the green infrastructure approach uses natural processes or ecosystems to perform similar actions, yet it can be implemented quickly and cost effectively to meet changing baseline conditions. Additionally, it can be designed either to complement existing gray infrastructure or as an independent treatment technology.

First proposed (MacKinnon et al., 2008, Mittermeier et al., 2008) as a pathway to meet climate change challenges while protecting biodiversity and sustainable human communities, the International Union for the Conservation of Nature (IUCN) has defined nature-based solutions (NBS) as "actions to protect, sustainably manage, and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham et al., 2016). Green infrastructure is a subset of NBS that utilizes ecosystems or their structural and functional aspects within the nexus of water and food to promote societal sustainability. Both provide benefits to humans from properly-functioning ecosystems (ecosystem services) that can be grouped into four broad categories: provisioning (food and water production), regulating (control of climate and disease), supporting (nutrient cycles and oxygen production) and cultural (spiritual and recreational benefits) (Millennium Ecosystem Assessment 2005). Humans are not simply the recipients of ecosystem services, but are critical components in local and regional ecology and management.

Although NBS in general, and green infrastructure in particular, can often operate as stand-alone technologies, they are increasingly integrated with gray infrastructure to produce hybrid solutions to meet increasing demands from individual sectors of the nexus, promote interactions among sectors, and improve overall management efficiency. Green infrastructure plays an important role in both water storage (water supply and flood control), treatment, and reuse and multipurpose projects promoting water, energy and food security in rapidly urbanizing areas of LAC. Increased emphasis is being placed on decentralized, small scale solutions near the source of individual problems. While traditional gray engineering approaches are top-down from international agencies and national governments, green infrastructure typically promotes a bottom-up approach and stresses local community ownership from project planning to implementation and operation. In addition, green infrastructure can often provide economic return to local communities, with declining product output serving as an early warning of declining system treatment efficiency necessitating initiating adaptive management plans to address the problem.

The current paper reviews green infrastructure approaches to the water, energy and agriculture nexus in LAC and identifies opportunities and challenges for their implementation. Biomes are proposed as the framework within which the current watershed approach for water management and project implementation can be compared with other geographical areas of similar physical and biological characteristics that may influence the performance of green infrastructure approaches. Throughout, particular emphasis is placed on the role of human communities to address fast-changing baseline conditions for urbanization, landscape alteration and climate change and the role of adaptive management to promote sustainability of the structure and function of natural and created systems.

GREEN INFRASTRUCTURE AND THE NEXUS IN LAC WATER

Water Supply. South America delivers more freshwater discharge to the sea per land area than any other continent, with the largest contributor being the Andes (Harden, 2006). Most rivers in LAC originate in high mountains (montane biome) as headwater streams fed by glacial meltwater (Guido et al., 2016). These streams are the main contributors to major rivers, including 50% of the Amazon River flow (Andersson et al., 2018), and are key sources of potable water for large cities like La Paz, Lima and Quito (Chevallier et al., 2011), as well as for hydropower generation and agriculture in high mountain communities.

Water resources in the high Andes are limited as a result of pronounced climate gradients and extremely steep topography (Buytaert & De Bievre, 2012) and dependent on long-term storage in glaciers and seasonal meltwater and precipitation. The progressive loss of glaciers over the past 50 years, which accelerated during 1976-2010 (Rabatel et al., 2013), is a clear indication that climate change is having and will continue to have a strong impact on the high Andes (Bradley et al., 2006). Soruco et al. (2015) noted that although there has been a 50% loss in total area of the 70 glaciers within the La Paz watershed between 1963 and 2006, runoff has not changed significantly. However, such seemingly stable hydrology simply reflects increased meltwater discharge and will soon revert to considerably reduced runoff as glaciers reach the tipping point of no return (Chevallier et al., 2011, Soruco et al., 2015). The greatest impacts on water supplies will be during the dry season, with a 24% drop in availability for La Paz, Bolivia (Soruco et al., 2015) and a 40% decline in discharge of the Rio Santa in Peru (Bradley et al., 2006). Reduced river flows, plus periods of intense drought, have contributed to ongoing water shortages in La Paz (Martinez, 2017), but Buytaert and De Bievre (2012) cautioned that human demographics are likely to outpace the impact of climate change on water availability. Overall, reduced water discharge from the Andes will affect all sectors of the nexus, including a shift from hydroelectricity to greater reliance on fossil fuels for energy and disruption of irrigated agriculture (Bradley et al., 2006).

As overall Andean water resources decline, there must be greater emphasis on water storage near its source, whereas controlled release is needed downstream. As suggested by Bradley et al. (2006), there has been increased emphasis on construction of new highland reservoirs. Constructed reservoirs –a key component of the comprehensive plan to supply irrigation to local communities and drinking water to the rapidly expanding city of El Alto and metropolitan La Paz– have been coupled with water storage in *bofedales* wetlands immediately below glacial outflow. Bofedales have strong water retention capacity due to their highly organic soils (Harden, 2006, Buytaert et al., 2006), and have been manipulated to provide water storage and pasture for livestock since prehispanic times (Fonken, 2014). Bofedales feeding into reservoirs provide a green infrastructure maximizing water storage and timed release for nexus needs. A similar role can be played by *páramo* wetlands in the northern Andes (Buytaert et al., 2006).

Two innovative programs are developing networks to provide baseline hydrological data for all watersheds in LAC against which green infrastructure potential can be assessed and implemented: The National Water Reserves Program of Mexico and Hydro-BID.

Developed in cooperation with the Inter-American Development Bank, the World Wildlife Fund (WWF), CONAGUA and Fundación Gonzalo Río Arronte, Mexico's Water Reserves Program **(http://awsassets.panda.org/downloads/wwf_mex_water_reserves_program.pdf)** is mapping and characterizing 189 river basins (watersheds) for integrated planning and management of both groundwater and surface water to reach water security through long-term planning and quantity and quality monitoring. The project emphasizes maintenance of ecological flows and biological diversity, while supplying domestic needs. Initial results for environmental flows and allocation of water to ecosystem and human sectors are very encouraging (Ordonez et al., 2015).

Hydro-BID **(http://Hydro-BIDlac.org/)** is a simulation tool launched by the IDB to estimate water budgets for all watersheds in LAC to help predict short- and long-term responses of water resources to demand, climate change and both floods and droughts. From initial projects in Argentina, Peru, Ecuador, Brazil, and Haiti, the Hydro-BID Technical Support Center (CeSH) is working towards a long-term goal of capacity building of water resource management and water supply institutions at national, subnational, and basin levels throughout the region.

The two programs focus on integrating natural flow regimes (Poff et al., 1997), critical water flows, and levels that must be maintained to ensure river connectivity from source to discharge for both biodiversity protection and minimum water quantities to meet human needs. These programs are considered critical due to their use of adaptive management, in combination with NBS and green infrastructure, to address natural and human-induced changes in water resources and their allocation.

Wastewater Treatment. Rural populations in LAC have long separated blackwater from graywater household wastes, with the latter discharged directly into the environment. But with fast urbanization, household wastes are combined and overwhelm treatment plants. Most nations in the region treat only 10-15% of wastewater (Reynolds, 2002, Noyala et al., 2012, Wilk & Altafin, 2018). Chile is the best performer, treating close to 100% of its wastewater, while Mexico, Nicaragua, Brazil, and Uruguay treat 35-50%.

Noyola et al. (2012) surveyed 2,734 municipal wastewater treatment plants in six countries throughout the region (Brazil, Chile, Colombia, Dominican Republic, Guatemala, and Mexico). They found that 67% of the plants were small (influent flow <25L/s) to very small (<5L/s), especially in Mexico and Brazil, and that 80% were stabilization ponds, activated sludge and upflow anaerobic sludge blanket reactors. It was concluded that such small facilities likely resulted in low energy efficiency and noncompliance with discharge standards.

Conventional wastewater treatment plants have a useful life expectancy of 60-70 years for concrete structures and 15-25 years for mechanical and electrical components (U.S. Environmental Protection Agency, EPA 2002). Constructed wetlands, however, do not have a finite lifespan, and with proper monitoring and adaptive management to maintain peak operational efficiency, can provide services indefinitely. Wetlands take up nutrients and other contaminants via macrophyte or algal photosynthesis and sediment microbial processes and can store them either short term (algae or herbaceous marshes) or long term (woody vegetation in swamps and sediments). For them to operate at peak efficiency, vegetation must be harvested and/or sediments removed periodically. Where large wetlands can be constructed and contaminant loading rates are low to moderate, swamps dominated by trees are favored because long term nutrient storage in wood maximizes the time between harvests needed to maintain system efficiency. In areas where terrain or urban densities dictate small footprints for constructed wetlands, fast-growing herbaceous macrophytes are favored, but they must be harvested frequently both to encourage new vegetative growth and to circumvent contaminant release from decaying vegetation. Whenever possible, only native species of plants should be used in constructed wetlands

Vymazal (2011) noted that the three essential design criteria for the four major constructed wetland designs are hydrology (surface, subsurface flow), vegetation type, and flow (horizontal, vertical). Surface flow wetlands slowly pass water through macrophyte stands for effective sedimentation of solids and uptake of nitrogen and phosphorus. They have been used most commonly for tertiary treatment of domestic wastewater, stormwater and mine discharge, and function well in all climates.

Subsurface flow wetlands consist of both horizontal and vertical flow treatment designs. In horizontal flow systems, water passes through a porous substrate upon which macrophytes are planted. Nutrient and contaminant removal takes place via both aerobic and anaerobic processes, which are very efficient at eliminating organic matter, suspended solids and heavy metals. Such systems are mostly gravity flow-based, leading to low operation and maintenance costs, and can treat diluted wastewaters from combined sewer systems better than conventional activated-treatment technologies. System longevity is strongly influenced by the relationship between substrate porosity and clogging. Overall, horizontal flow wetlands are more effective than surface flow wetlands in colder climates.

Vertical flow wetlands are of two basic designs: downward and upward flow. Both use a coarse substrate of sand and gravel upon which macrophytes are planted. In downward flow systems, wastewater is fed to the surface in batches and percolates through the substrate. Between batches, water is drained completely, facilitating oxygen movement into the substrate for nitrogen treatment. These systems have lower land requirements than surface flow wetlands to achieve effective waste treatment. Upward flow wetlands require higher energy inputs and maintenance than downward flow methods. In them, wastewater is pumped in batches into the bottom of the wetland and allowed to percolate upward to pond on the surface, where it remains static for a set time to support microbial growth and nutrient transformations before being drained downward through the substrate, where it is collected and discharged.

There is extensive literature on the operation of constructed wetlands (Zi and Ji 2012) and removal efficiency for nutrients (Moshiri 1993), heavy metals (Kadlec and Knight 1996), acid mine drainage (Johnson & Hallberg, 2005, Nyquist & Greger, 2009) and, most recently, synthetic compounds in cosmetics and personal care products linked to endocrine disruption in aquatic environments (Hijosa-Valsero et al., 2010, Toro-Velez et al., 2016). Treatment efficiencies of constructed wetlands are similar to their gray infrastructure counterparts (Kadlec & Wallace, 2009). Although less extensive than for temperate systems, treatment efficiencies in the tropics equal or exceed those of conventional systems, and construction and operating costs of constructed wetlands in China are approximately 30% and 50% those of gray infrastructure, respectively. Similarly, Arias and Brown (2009) estimated the annual cost for operating a constructed wetland near Bogota, Colombia as comparable to a stabilization pond but about 20% that of a batch reactor. The first publication on constructed wetlands in LAC was for a site in Brazil (Salati & Rodrigues, 1982). Between 1991 and 2011, Brazil and Mexico published the largest number of studies on constructed wetlands (31-40), followed by Argentina and Colombia (11-20) (Zhi & Ji, 2012). Seven other LAC nations were just beginning to consider constructed wetlands during this period (Costa Rica, Panama, Venezuela, Chile, Uruguay, Jamaica, and Cuba) resulting in few refereed publications (1-10). It is important to note that Jamaica and Cuba were the only Caribbean nations represented. It was apparent from the review of mostly constructed wetland projects in Bolivia, Brazil, Mexico, Nicaragua, and Peru by Wilk and Altafin (2018) that data from individual projects are either not published or are issued as gray literature or internal reports. The same is true throughout LAC, but perhaps most evident in the Caribbean basin, where publications are scant and few overview websites exist aside from those in St. Lucia **(https://www.caribbean-sea.org/projects/st-lucia/)** and Antigua **(www.bvsde.paho.org/bvsAIDIS/PuertoRico29/morris.pdf)** and university theses in Jamaica (Stewart, 2005).

Although the great potential of constructed wetlands for wastewater treatment in the tropics has long been recognized because of their use of natural processes, low building and maintenance costs and comparable treatment efficiency to conventional gray infrastructure (Denny, 1997, Haberl, 1999, Kivaisi, 2001), Noyola et al. (2012) documented only 137 active constructed wetlands in LAC. Most are scale pilot operations in small rural villages (Whitney et al., 2003, Dallas et al., 2004, Rios et al., 2009, Kaplan et al., 2011, Zurita et al., 2012) and universities (Mitsch et al., 2008).

There is a critical need to scale up operations, especially of hybrid gray-green systems, to meet dynamic urban demographics, complicated waste streams, and climate change (WSP 2008). Currently, studies are testing the best design and macrophyte species composition to address specialized waste streams from dairy farms and processing plants, banana paper plants, landfills (Nahlik & Mitsch, 2006), swine farms (Gonzalez et al., 2009) and slaughterhouses (Rivera et al., 1997). In addition, removal of both pathogens (Dallas et al., 2004) and synthetic compounds (Belmont et al., 2006, Toro-Velez et al., 2016) that are partially treated in conventional wastewater plants has received growing attention. Linking constructed wetlands with small conventional treatment plants offers a cushion to treat peak loads, vastly increasing the efficiency of phosphorus removal (Arias & Brown, 2009). For both standalone and hybrid uses of green infrastructure, adaptive management is required to keep systems operating at peak efficiency, and increasingly is focused on product development to remove accumulated sediment and nutrients and provide economic return and project acceptance to local communities.

ENERGY

Hydroelectric power. Although estimates vary, mountainous nations in LAC get most of their energy needs from hydroelectric plants. Costa Rica leads the pack with nearly 100%, followed by Ecuador, Peru, and Colombia (>70% of demand). Even low-lying Brazil is currently 80% hydroelectric (Tundisi et al., 2014). Peru has the greatest number of dams, and together with Bolivia is dominated by small dams located high in the mountains (Anderson et al., 2018). Brazil's oldest dams are in the eastern and southeastern parts of the country, but as all favorable sites for dam construction have been occupied, construction has begun in the Amazon basin (da Silva Soito et al., 2011, Tundisi et al., 2014).

While the number of small dams high in the Andes is on the rise, most new hydroelectric reservoirs are situated lower in the Andean Amazon and the lowland tributaries of the Amazon River. Small Andean reservoirs have limited potential for hydropower due to their size and depth, but they are critical for both irrigating alpaca pastures and quinoa fields and as a water supply for large cities including La Paz, Lima and Quito (Chevallier et al., 2011).

To take advantage of high rainfall and striking relief patterns, 151 large dams on 6 of 8 major tributaries of the Amazon are planned or are being built in the next 20 years for a 300% increase in power generation (Finer & Jenkins, 2012, Andersson et al., 2018). Most are located in the Amazonas high Andes ecoregion and are larger than any current facility in any Andean nation, with the largest number of dams planned for Peru. The Andes contribute 50% of the Amazon River flow, 93% of its sediment load and most of the nitrogen and phosphorus to drive riverine productivity (Andersson et al., 2018). However, there is great concern that these new Andean dams will disrupt the connectivity of the Andes with the lowland Amazon, trap 100% of sediment needed for diverse channel morphological types and a significant portion of both nitrogen and phosphorus, and isolate conservation areas higher up in the mountains from the lower river (Tundisi et al., 2014, Finer & Jenkins, 2012, Andersson et al., 2018). Forsberg et al. (2017) estimated that together the six largest dams planned for the region will lower the supply sediment, phosphorus and nitrogen of the entire Amazon basin by 64, 51 and 23%, respectively. Furthermore, associated lowered river productivity will significantly reduce fish yields, the primary protein source for local communities along the river.

As the best sites for dams have already been occupied in eastern and southeastern Brazil, the Amazon basin is viewed as the "new hydroelectric frontier," with 100 dams in operation and an additional 137 planned (Tundisi et al., 2014, da Silva Soito & Freitas, 2011). Because of generally low relief, dams in the Amazon basin tend to be wide and shallow, causing large forest areas to be flooded by reservoirs that trap large quantities of river-laden sediment critical for lower Amazon alluvial floodplains and channels (Manyari & de Carvalho, 2007, Finer & Jenkins, 2014). These huge reservoirs tend to become anoxic over large areas due to decomposition of rainforest vegetation left in place during construction, posing water quality problems from excess nutrients that foster algal blooms. They also are major contributors to atmospheric sulphur and carbon dioxide during low water phases (Tundisi et al., 1998, Rosa et al., 2003, Forsberg et al., 2017).

In addition to trapping sediments and nutrients, hydroelectric reservoirs can have a major impact on water security due to surface-water evaporation—their blue water footprint. The larger the lake area and the higher the temperature, the greater the release into the atmosphere (Hogeboom et al., 2018). Because of the topography-dictated need for vast surface reservoirs, Brazil has the largest blue water footprint of any nation globally, drawing into question the positive benefits of hydroelectric power.

Dams can have significant social and ecological impacts upstream and downstream (Finer & Jenkins, 2012). Rivers evolve in their structure and function from headwaters to discharge in a predictable manner (stream continuum), that if interrupted by dams (serial discontinuity) can dramatically affect upstream movements of commercially important and endangered fish species (Ward & Stanford, 1983). For this reason, Tundisi et al. (2014) recommended spacing out dams along river lengths to allow sufficient distance for the river to recover before encountering another reservoir.

Perhaps dams' most significant impact is on ecological flows, "the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems" (Brisbane Declaration 2007 **http://www.watercentre.org/news/declaration**). In addition to maintaining a minimum flow in river channels, recommended flows need to adjust to a number of variables such as high flow pulses throughout the year and floods, along with monthly and inter-annual variability (Richter et al., 2006). Annual flooding of floodplains is especially critical for forest productivity and breeding of commercially important fish species in the Amazon basin (Tundisi et al., 2014).

In order to adjust to sustainable freshwater management standards, environmental flows must shift their focus from restoration to adaptation to climate and landscape changes, expand from single sites to whole river basins, and include social-ecological sustainability in all water management scenarios (Poff & Matthews, 2013). In addition, they need to recognize differences in riverine structure and function dictated by key processes operating at the biome and ecoregion levels and incorporate adaptive management that considers short- and long-term changes in the extent of individual biomes throughout LAC.

While hydroelectric dams have a major impact on watersheds via water storage and altered flows downstream, their contribution to atmospheric water loss by way of evapotranspiration has largely been ignored (Hogeboom et al., 2018). Although the relationship between water availability and human uses is readily recognized (water footprints), reservoirs have either been ignored from previous studies (Hoekstra & Mekonnen 2012, FAO 2016) or their contribution to evapotranspiration water loss to the atmosphere overlooked (IEA 2016). Hogeboom et al. (2018) proposed the Blue Water Footprint to calculate water consumption of blue water resources (surface water and groundwater), including evaporation, to provide a realistic picture of the influence of reservoirs on watershed water budgets. Brazil has the largest water footprint of reservoirs globally as a result of its relatively flat topography and the resulting need of large reservoirs for power production. This can dramatically affect water security downstream. In contrast, small dams and those located in montane areas usually have a much smaller impact. While addressing the broader impact of reservoirs on water security is a promising start, there is a need to conduct a detailed analysis of dam size relative to climatic conditions in all of LAC's biomes.

The recent incorporation of environmental flows into potential hydropower projects and evidence of dams' negative impacts both upstream and downstream point to a need to consider alternative power generation technologies that do not impede river flow, such as run-of-river (ROR) and tidal systems. ROR generates power by diverting flow from a river through a parallel channel equipped with a turbine. While pondage ROR designs include small dams on rivers to ensure stable water availability, the preferred design is subject to seasonal flow variation and is considered an intermittent energy source. Benefits of the latter include limited impact on natural flow regimes and unimpeded upstream movements of fish (Jager & Bevelhimer, 2007). The vast majority of ROR operations are in Asia, whereas in LAC Brazil tops the list (**https://en.wikipedia.org/wiki/List_of_run-of-the-river_hydroelectric_power_stations**).

Tidal power generation is based on both tidal magnitudes and currents from river discharge into ocean-run turbines. While it is considered a significant source of renewable energy, its environmental impacts are poorly known (Frid et al., 2012). Both ROR and tidal power generation have potential applications to LAC, but a detailed analysis of potential sites, operational problems and environmental impacts is needed.

Biofuels. Biofuels are touted as a viable alternative, particularly for nations lacking significant access to fossil fuels (Mosedale, 2008). They are viewed as a renewable resource that can help reduce greenhouse gas emissions and provide an alternative profitable use of crops and their residues beyond the food sector. (Sorda et al., 2010, Kraxner et al., 2013). Based primarily on sugarcane, Brazil is both the second largest producer and consumer and largest exporter of ethanol globally (Carneiro et al., 2014). Colombia, Venezuela, Costa Rica, and Guatemala are following the Brazil model in their emerging biofuel industries. Argentina has based its biofuel industry on soybeans and currently is the fifth largest producer of biodiesel globally (Janssen & Rutz, 2011). Additional emerging or potential sources of biofuel production in LAC include palm oil, maize, and coffee (Huttunen & Lampinen, 2005, Ludena et al., 2007, Adams & Ghaly, 2007, Graefe et al., 2011).

Biofuels are fully integrated within the framework of the nexus of water, energy and food and provide a unique opportunity to develop strategies for implementing sustainable approaches to landscape management from watersheds to biomes. But they can also have impacts associated with agrochemicals, erosion and soil carbon loss, deforestation, greenhouse gas emissions, altered microclimates, wildlife loss and conversion of agricultural lands from food to biofuel production (Janssen & Rutz, 2011, Cerri et al., 2011, Verdade et al., 2012, Sandhu et al., 2013, Ferreira et al., 2015, Muñoz Castillo et al., 2017, 2019).

In addition, there is serious concern that increased production of sugarcane for biofuel will affect food security, especially in LAC's poorest and driest regions (Janssen & Rutz, 2011). Noting that most studies on biofuel production have focused on either water or land and few have considered both, Muñoz Castillo et al. (2017, 2019) used an environmentally extended multiregional input-output (MRIO) model to quantify tradeoffs and synergies between land and water use for bioethanol production in Brazil. Bioethanol accounts for one third of the country's sugarcane water footprint and has less impact in water stressed states when rain fed rather than irrigated. While northeastern Brazil's water stressed states are likely to suffer from a larger bioethanol output, Sao Paulo will garner most of the economic benefits from its production and export. Similar detailed analyses are needed throughout LAC, but the global sustainable energy project (GSB) that started in 2009 is beginning to provide intercontinental guidelines for feasibility of biofuel production and sustainability (Lynd et al., 2011).

FOOD

Rural Agriculture. The high Andes are dominated by grasslands and unique wetlands, bofedales or páramos. For centuries, camelid herders have constructed canal networks to capture seasonal runoff and direct it to bofedales wetlands as a dependable water supply and pasture land (Verzijl & Quispe, 2013). Bofedales have expanded over time and are important for storing water high in the mountains. With a stable water supply, alpaca herds have expanded rapidly, and overgrazing is extensive in most high elevation areas and promoting serious erosion in 50-60% of the high Andes (Millones, 1982). Although an important grain for Andean peoples since at least 5000 B.C., quinoa production almost completely disappeared by the mid-20th Century (Pedersen, 2015). Since its "rediscovery" in the late 20th Century, quinoa production in Bolivia, Peru and Ecuador combined doubled between 2001 and 2011, with prices rising 300% between 2007 and 2008. Water security of the Altiplano, however, has become very unstable as more herders have enlarged their alpaca herds with profits from quinoa sales, thus increasing pressure on water stored in bofedales to replace that lost via climate change and progressive shrinking of local glaciers. In el Alto, Bolivia, additional reservoirs are planned to collect water more effectively from the area and supply La Paz with potable water.

LAC's Amazon rainforest, dry forests and subtropical grasslands are also severely threatened by agricultural conversion (Grau & Aide, 2008). Only 20% of the original Amazon forest remains, and at current deforestation rates, 27% of the current forest will be lost by 2030, especially in the Andean Amazon of Bolivia and Peru. And yet, these rates pale by comparison with that of Brazil (**https://en.wikipedia.org/wiki/Deforestation_of_the_Amazon_rainforest**).

Deforestation of *cerrado*, the savanna bordering the eastern and southern Amazon, has dramatically accelerated, with 30% of the remaining biome likely to be lost by 2050 (**https://www.worldwildlife.org/stories/saving-the-cerrado-brazil-s-vital-savanna**). While historically most cleared land in both biomes was for cattle production, currently most is for soybean. Brazil is the top soybean producer, having quadrupled crops in the past 20 years to meet China's growing demand (**https://www.producer.com/2018/05/brazil-could-take-soybean-production-crown-from-u-s/**). Increased output has resulted in massive erosion and transfer of increased sediment loads to rivers of the Amazon system to the north as well as to those flowing south into the Pantanal, the world's largest wetland.

Traditionally, western Costa Rica's dry forest biome, Guanacaste, was used for cattle grazing during the dry season and suspended when the area became wetlands during the wet season, especially in the Tempesque River basin. The Arenal-Tempesque Irrigation Project (PRAT) was initiated in 1980 to provide water continuously to overcome constraints imposed by wet-dry seasonal rainfall and periodic droughts (Daniels & Cumming, 2008, de Szoeke et al., 2016). Water is conveyed from Lake Arenal in the highlands via the 7 m wide Canal Oeste to encourage conversion of dry forests to cattle grazing and crop production. The newly irrigated land quickly contributed 50% of Costa Rica's total rice and sugar output. While the original canal was concrete-lined with underpasses at stream crossings, 20 km of a hastily constructed canal extension in the 1990s to boost production was sediment-lined and lacked culverts for stream passage. Stream hydrology changed completely from intermittent to permanent flow as a result of canal leakage. Such quick responses of governments to meet changing agricultural market trends ignored concerns for water security in the region. Wetlands were turned into rice paddies, and many were subsequently switched to sugarcane to support biofuel markets. But sugarcane requires land drainage rather than irrigation, so the water brought by the canal goes to waste. There is a critical need to balance infrastructure construction with high value crops throughout the region (Ringler et al., 2000).

Urban Agriculture. Food security has always been a factor in urban sustainability. Urban agriculture was a major activity in Classic Mayan cities and Byzantine Constantinople (Barthel Isendahl, 2013). In northern Europe it morphed into Schrebergarten or garden colonies in most German cities beginning as early as 1826, when lands with limited value, including floodplains and those bordering railways, were leased to urban dwellers both to promote outside activities and to produce food. Such gardens are still extremely popular in Austria, Germany and Switzerland. Artmann and Sartison (2018) noted that while peri-urban agriculture supports ten key societal challenges facing urban populations –climate change, food security, biodiversity and ecosystem services, agricultural intensification, resource efficiency, urban renewal and regeneration, land management, public health, social cohesion, and economic growth–, such urban agricultural schemes are under threat from development. Urban agriculture can promote long-term food security in times of energy shortages and is an important component of water management contributing to resilience of cities (Barthel & Isendahl, 2013).

At least 800 million people globally are engaged in urban agriculture, accounting for roughly 15% of food supplies (Kisner, 2008). Local production reduces postharvest food losses from inadequate preservation and transportation to market (Qiu et al., 2013). Mexico City currently meets about 20% of its food demand via rooftop gardens and hopes to expand them throughout the city (Dieleman, 2014). Rooftop gardens, vertical gardens and living walls show great potential for reducing air pollution and temperature (Qiu et al., 2013), as well as water treatment and reuse during dry periods (Rowe, 2010) in LAC cities. Rooftop gardens (green roofs) are also effective at both reducing stormwater runoff (30 to >40%) via plant and soil uptake and evapotranspiration, resulting in improved water quality (Stovin 2010, Hashemi & Mahmud, 2015, Feng et al., 2016). Mexico City has implemented highly innovative projects using rooftop gardens and vertical gardens (living walls) to capture stormwater runoff and to reduce air pollution (Qiu et al., 2013, Dieleman, 2017). Although limited, data clearly point to the fact that these are promising technologies to reduce runoff and promote water reuse in LAC.

Rooftop gardens are favored in dense urban cores with limited space, while hydroponic gardens are common in peri-urban areas with fewer space limitations and readily available gray water sources. In some impoverished peri-urban areas of Lima, hydroponic gardens have played a key role, helping combat malnutrition and poverty (Orsini et al., 2010). The production potential of high value crops is great provided markets can be established (Schnitzler, 2012). There is a critical need, however, for analytical data to evaluate operational efficiencies of both rooftop and hydroponics in LAC. There is little doubt that these operations can become important components of adaptive management to meet both flooding and drought in urban areas with impressive economic and social returns.

Local stakeholders and communities must be encouraged to take ownership of green urban infrastructure. There are great opportunities for local communities to become stewards of green projects via product development that is also part of the adaptive management needed to keep the system operating at peak efficiency. In addition to providing irrigation water for crops such as rice (Salati et al., 1999) and select vegetables (Martinez-Cruz et al., 2006), high value plants, especially flowers, can be planted directly in the constructed wetlands, especially subsurface, horizontal flow systems (Belmont et al., 2004, Zurita et al., 2009, Zurita et al., 2011). There are additional wetland products that support local economies, as seen in Uganda: fish, vegetables, livestock grazing, and building materials for houses and furniture (Kakuru et al., 2013). Increased emphasis needs to be placed on compiling wetland, floodplain and urban habitat economic activities supporting the nexus's water, energy and food sectors.

OPPORTUNITIES FOR GREEN INFRASTRUCTURE FOR NEXUS APPLICATIONS IN LAC

High Mountains. As Andean glaciers disappear, management strategies must emphasize water storage as close to the headwater source as possible for controlled release downstream to meet water supply, energy and food production needs. Bofedales wetlands of the central Andes and páramo grasslands/wetlands of the northern Andes have high water retention capacity due to their highly organic soils (Harden, 2006, Buytaert et al., 2006, Fonken, 2014). These permanently wet ecosystems also display significant carbon sequestration for long term storage, a relationship that is positively related to increasing elevation (Pena et al., 2009, Muñoz et al., 2015). When linked with storage reservoirs immediately downstream –as implemented by the rapidly expanding city of El Alto and metropolitan La Paz–, bofedales appear to significantly enlarge glacier meltwater storage. However, such wetlands are extremely sensitive to overgrazing, so additional research is needed to maximize their water storage capacity in the face of climate change.

Glacial meltwater is the only source of irrigation for approximately 80% of farmers around Ladakh, India. In response to dwindling and unpredictable snowfall, retreating glaciers, shorter winters, drought, and landslides, communities used local knowledge and institutional arrangements to resort to water-harvesting green infrastructure in order to ensure they had sufficient water storage levels to support farming (Nusser & Baghel, 2016, Clouse, 2016, Clouse et al., 2017). Three innovative frozen landscape interventions were employed: artificial glaciers, ice *stupas* and snow barrier bands (Clouse, 2016). All are built as close to the glacial water source as possible and capture meltwater for longer term storage and release as needed for agriculture in the valley below. The first artificial glacier was designed by a Ladakhi engineer in 1987 and diverts meltwater in October and November via gravity to freeze in a series of cascading rectilinear pools spread across low sloping, shaded sites surrounded by stone walls. Ice stupas are named after dome-shaped Buddhist shrines and are formed by gravity feeding water from the stream in a pipe that ends in a freestanding position on level ground. Water slowly freezes as it exits the pipe, forming freestanding domes up to two stories high. With volumes exceeding 150,000 liters, the larger stupas remain intact long after annual snowmelt. Snow barrier bands are masonry walls that trap snow that would otherwise be blown away over the mountains. The snow melts early in the spring to support farmers during the planting season. All these techniques provide only temporary relief from source-water decline (Clouse, 2014, Clouse et al., 2017), but their importance is often overlooked in the face of pressing socioeconomic, cultural and political concerns (Barrett & Bosak, 2018). Still, artificial glaciers have been recognized as one of the top fifteen features that can have a major impact on global biodiversity in the future (Sutherland et al., 2016), and could be a viable option for use in the high Andes.

The ancient Egyptians depended on annual Nile River floods to supply their floodplain farms with sediment and nutrients. Floodplains are one of the most threatened ecosystems globally, with 90% of European and US systems considered functionally extinct due to intensive agriculture (Tockner & Stanford, 2002, Opperman et al., 2009), and with rivers bordered by high modified floodplains displaying unpredictable, short-lived flood pulses (Junk et al., 1989). Although disappearing rapidly through altered hydrology, numerous tropical floodplains are still relatively intact (Tockner & Stanford, 2002). Reconnected floodplain-river systems enhance river basin resilience to absorb human and climate impacts, reduce flood risk and boost overall ecosystem goods and services availability (Opperman et al., 2009, Kiedrzynska et al., 2015).

Large-scale reconnection of rivers and floodplains increases flexibility and resilience of water management infrastructure by reducing the need to lower water levels in reservoirs in anticipation of pending floods, thus minimizing impacts of downstream environmental flows (Opperman et al., 2009). Grygoruk et al. (2013) calculated the value of water storage in a Polish floodplain during floods at EUR 5.49 million per year, and Costanza et al. (1997) noted that the numerous services provided per acre of floodplain are surpassed only by those of estuaries.

Given that floodplains are extremely effective at trapping nitrate in floodwaters (28-47%) but less so for phosphorus (4-7%) (Kronvang et al., 2007), Mitsch and Day (2006) have suggested reconnecting drained floodplain wetlands along the Mississippi, Ohio and Missouri to prevent nitrogen from polluting the Gulf of Mexico, while simultaneously restoring ecosystem structure and functions. Such an approach also has great potential in LAC, where there are many large floodplains capable of significantly reducing nutrient loading to coastal cities and beyond.

Urban Areas. The Urban Stream Syndrome has evolved to catalog the negative physical, chemical and biological impacts of urbanization of streams including flashier hydrology, high nutrient and contaminant concentrations, altered channel morphology, and reduced biodiversity (Walsh et al., 2005, Booth et al., 2016). Although similar impacts have been reported in cities globally, the top controlling factor for the intensity of each is the climate physiographical region (i.e., the biome in which the city is located) (Booth et al., 2016, Hale et al., 2016). Hughes et al. (2014) noted that urban streams respond to both latitudinal and longitudinal impacts at scales from watershed to stream reach, but Ramirez et al. (2012) found that island streams, including those of Puerto Rico, are most responsive to longitudinal connectivity to the sea for diadromous species.

Streams can provide numerous ecosystem services to cities (Everard & Moggridge, 2012, Delibas & Tezer, 2017), including water storage and nutrient uptake. The Urban Watershed Continuum (Kaushal & Belt, 2012) stresses that connectivity of streams from headwaters to discharge must be maximized to support peak efficiency stressing a continuum of engineered and natural hydrologic flowpaths to transform and transport materials and energy based on hydrologic residence times. Headwater streams are critical for watershed functioning because of their major contribution to stream length and proximity to local pollution inputs. However, they have been buried all around the world at a disproportionate rate because of their small size. Elmore and Kaushal (2008) documented that 66% of all headwater streams in central Baltimore, Maryland and 70% of streams in watersheds smaller than 260 ha had been buried during urbanization. In addition to serving as a constriction point leading up chaotic upstream flooding, buried streams disrupt nutrient retention upstream and Beaulieu et al. (2015) found that nitrate travelled on average 18 times farther downstream in buried streams before biological uptake in the stream channel.

Pinkham (2000) coined the term "daylighting" to describe the process of exposing previously buried streams. Collapse of a street over the buried Ilisos River in Athens, Greece has led to a plan to daylight the river channel as both a lower cost alternative and green infrastructure for stormwater (BBC News 27 February 2019). Elsewhere, stream daylighting has gained wide acceptance in numerous cities of the temperate zone (Buchholz & Younos, 2007). Stream daylighting is not well advanced in LAC, with one of the earliest projects under consideration for Medellin, Colombia (Calderon, 2013). Everard and Moggridge (2012) stressed the need for planning tools to implement stream restoration technologies, and Hale et al. (2016) suggested controlled, large-scale research at select sites globally to fine-tune urban climate, effectively a biome base approach. Finally, of the 26 key identified research needs requiring immediate attention in urban stream ecology, most focus on the need for further understanding stressors of streams within the context of the Urban Stream Syndrome (Wenger et al., 2009).

Biomes for Evaluating Green Infrastructure. Environmental management has recently been based at the watershed scale. This is totally appropriate as watersheds are a single hydrological unit from headwaters to discharge downstream and includes both surface and groundwater. Both Hydro-BID and Nature Reserves programs have demonstrated the value of watershed approaches to water security issues, but they have also shown how significant interbasin differences can be, even for seemingly similar adjacent basins (Crisman 2014). The types of questions that can be addressed effectively depend on watershed size. For massive watersheds like the Amazon and Orinoco, regional issues like climate change and massive deforestation can be addressed, but only small to intermediate sub-watersheds are appropriate for understanding impacts associated with individual development projects. No standardized watershed exists that can serve as a single model at national and regional levels.

Ecoregions, first proposed by Loucks (1962) and adopted by the U.S. EPA in the late 1980s (Omernik, 2004), are macroscale terrestrial and aquatic ecosystems that interact in a predictable fashion with a set of abiotic and biotic factors (Bailey, 2004). They are usually considered subsets of biomes and share a geographical distribution with comparable life zones and biogeographic provinces. WWF has divided the earth into 867 terrestrial, 426 freshwater and 232 coastal and continental shelf ecoregions based on their distinct biota. Of these, 238 single or combined ecoregions are considered of conservation priority. Ecoregions are clustered into complexes and bioregions defined by similar biogeographic history and strong affinity at higher taxonomic levels, 42 of which are located in LAC. This extremely complex classification hinders the development of standard practices for addressing green infrastructure project applications throughout LAC. In addition, there is no direct relationship between ecoregions and the extent of individual watersheds and greatest emphasis is placed on biodiversity, often out of the context of the physical environment and human interactions. Ecoregions are embraced by several conservation organizations, including WWF and The Nature Conservancy, as the baseline for conservation strategies and ecosystem management, including LAC (Olson et al., 2001, Dinerstein et al., 1995). Detailed ecoregion maps for both terrestrial and aquatic systems have been published for all of LAC and most nations.

A biome is a zone of common physical environment and climate with distinct plants and animals adapted to that set of conditions. Because they are controlled by physical parameters, they share common characteristics throughout their global extent. It is proposed that biomes become the basic geographic unit for developing regional and national action plans for green infrastructure implementation. Individual biomes often have numerous ecoregions and habitats, but they provide a baseline for developing principles that recognize the overriding importance of the hydrological cycle, local geology, elevation and common land use practices in water resource management.

It is suggested that six biomes in LAC become the basis of guidelines for assessing pending and current green infrastructure projects relative to physical and biological environmental parameters in the context of landscape and water management, urbanization and climate change (Figure 2). Although not a "natural" biome, Urban has been included as a seventh category because of the cross-cutting similarities in structure and function of all major urban areas as well as their individuality driven in large part by their biome location.

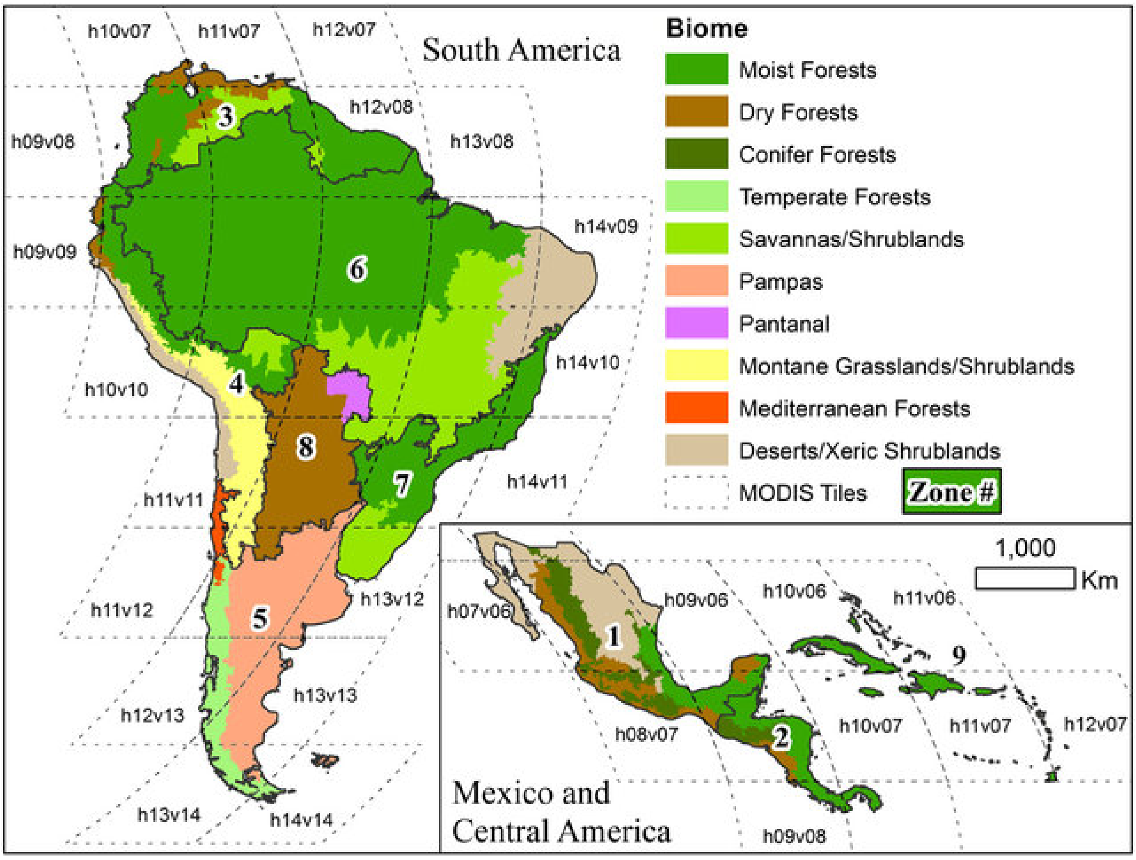


Figure 2. Major biomes in LAC. Adapted from Olson et al., 2001.

Montane. This biome includes both the high mountains above treeline and high mountain forests. The dominant ecosystem types in these areas are extremely fragile páramos and bofedales, whereas glaciers, the main water supply, are quickly disappearing.

Rain Forests. Rain forests are characterized by high rainfall, plant production and biodiversity, but are extremely sensitive to disturbance. In addition to the Amazon basin, rain forests are found throughout Central America and as far north as southern Mexico.

Savannas. Extensive grasslands found in Venezuela, Brazil and southeastern South America. They typically have a high natural fire frequency, which has been expanded by humans to provide high quality pasture for cattle.

Dry Forests. Dry forests extend along the Pacific coast from southern Mexico through Central America and into northern South America and eastern and central Brazil (Cerrado). They are also the dominant biome on the southern portion of Caribbean islands with mountains high enough to reduce rainfall on their leeward flanks. Dry forests experience pronounced annual wet and dry seasons and generally exhibit high fire frequencies.

Deserts. The most extensive deserts in LAC are in northern Mexico, the Pacific coast from Peru to northern Chile and in Argentina (Patagonian Desert). A few southern Caribbean islands are also dominated by desert biomes as are some parts of the high Andes. Desert biomes are extremely fragile systems with slow recovery rates following disturbances.

Coastal. Coastal areas are often considered a separate biome based on shared interactions between terrestrial and marine habitats that are universal globally. LAC is characterized by numerous coastal terrestrial biomes from Mata Atlantica forests in Brazil to deserts along the Pacific coast of South America, dry forests of Pacific Central America and rainforests of Caribbean Central America. Recognition of a distinct coastal biome is also a unifying concept critical to management of Caribbean islands.

Urban. Ellis and Ramankutty (2008) and Pincetl (2015) suggested that urban areas be considered as an anthropogenic biome, with a suite of functions and processes comparable to those of any natural biome. Given LAC's rapid urbanization, common features and problems exhibited by all large metropolitan areas as well as linkages with and dependence on surrounding biomes, it is logical that water security be considered in both natural and anthropogenic biomes.

Since the structure and function of green infrastructure depend on local biology and climate, differences in key physical/chemical and biological parameters determine overall project efficiency (Table 1). Understanding a biome's conditions is critical to determine green infrastructure technologies suitability and their resilience to climate change. Each biome will have a suite of aquatic plants adapted to local temperature, hydrology, length of growing season, and sunlight conditions. These factors must be considered when selecting those plants with the best survival and productivity prospects and with the greatest potential for product development by local communities.

Table 1. Biome Influenced Wetland Parameters

Physical and Chemical Parameters

Precipitation: total and annual distribution of precipitation

Evaporation-transpiration rates

Groundwater and surface water resources

Local geology: dominant rock and soil type, weathering rates and levels of nutrients, conductivity and heavy metals

Elevation and temperature: length of growing season, daily and annual temperature ranges

Solar insolation: total and annual distribution, wavelengths

Biological Parameters

Macrophyte species presence and availability

Productivity of plant species under local conditions

Potential for product development from candidate wetland and swamp plants

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The biome approach to green infrastructure and NBS is recognized as an emerging approach for evaluating climate change impacts and human alterations to landscapes. Both de Groot et al. (2012) and Costanza et al. (2014) calculated ecological services' values for 10 of the 12 global biomes defined by de Groot et al. (2010). The most critical biome affecting water security in LAC, montane, was left out of analyses likely due to a scarcity of data points. Costanza et al. (2014) stressed that it is critical to factor a biome's geographical changes into all valuations, as boundaries are dynamic and responsive to climate control and human landscape alterations. Finally, Boumans et al. (2002) used GUMBO, a metamodel of the biosphere to simulate services and ecosystem goods for 11 biomes. The model is a synthesis of several dynamic global models of intermediate complexity for both social and natural sciences. They estimated that global ecosystem services were 4.5 times greater than gross world product in 2000.

The geographic extent of LAC's biomes is changing in response to climate change. The high montane Andes currently accounts for 9.5% of the world's freshwater resources, but it is experiencing rapid warming and the retreat and loss of glaciers, especially in the central Andes, threatening the water security of cities like La Paz (Russell et al., 2017). Similar changes in the extent of savanna and rainforest biomes are expected (Salazar et al., 2007). In addition, there are major metropolitan areas in all biomes of LAC, which might be considered Anthropocene biomes. There is a critical need to develop broad principles governing the three nexus sectors that can be applied to similar areas throughout LAC. Adaptive management strategies must be put in place to account for water resources changes associated with shifts between biome types as a result of climate change.

CHALLENGES FOR GREEN INFRASTRUCTURE AND THE NEXUS

Convincing Stakeholders. While ecologists have long separated humans from nature, they recently began to recognize the dynamics of tightly linked social-ecological systems (Alessa & Chapin, 2008). Brink et al. (2016) suggested that ecosystem-based adaptation (EbA) (i.e., nature-based solutions) are too narrowly focused and lack interdisciplinary rigor, with few studies including stakeholders participation. The authors raised three challenges to future research: 1) integrating systems perspectives of EbAs within a socio-economic context, 2) identifying human winners and losers in EbA projects and 3) placing more emphasis on EbA's role in transformative adaptation. Yocum (2014) stressed that project success should be gauged by both socio-economic and ecological performance and must include community, educational and participatory considerations. Moreover, Pickett et al. (2001) noted that urban ecosystems studies are full of contrasts of urban versus biological planning, ecology versus ecology of cities, and disciplinary versus interdisciplinary, and posited that they must be reformulated into a human ecosystem framework.

Quintero (2012) stressed the need to move away from a project-to-project approach and embrace a broader temporal and regional perspective. Pincetl (2015) proposed that urban ecosystem services be evaluated within the framework of the urban biome, within which cities will be classified by city-type and characterized by climate zone.

Governments and Policy. Although the United Kingdom has created a legal framework for green infrastructure to promote quality environments via integrative design, social inclusion and public participation, there remains a big gap between goals and project implementation (Mell, 2009, Roe & Mell, 2013). The result is "institutional schizophrenia", a fragmented approach that affects stakeholder collaboration and confidence and is associated with restrictions placed on local authorities. Additionally, Kabisch et al. (2016) identified the challenges that science and policy face to implement nature-based solutions: 1) produce stronger evidence of their value through concrete results, 2) address governance issues by establishing networks of society, nature-based supporters and practitioners, and 3) consider integrated governance approaches that include all stakeholders to address socio-environmental justice and social cohesion issues.

Economic Feasibility and Project Funding. The UNDP, WWF, TNC and World Bank embrace emerging economic arguments that green infrastructure can provide the same or better benefits than gray infrastructure and with cheaper investments (Green et al., 2015). Its benefits can be direct (sale of goods, clean water, cultural services that can be traded on the market) or indirect (positive externalities including lower risk of flooding or altered climate) and are extremely hard to assess in traditional economic models (Jaffe, 2010, Bockarjova & Botzen, 2017, Barbier, 2011). Ecosystem services are broadly considered to be human benefits derived from processes reflecting ecosystem structure and function. Costanza et al. (2014) pointed out that valuations of ecoservices is not the same as commodification or privatization and that many ecoservices are best considered public goods or common pool resources. Regardless, if nature contributes significantly to human well-being, then it is a major contributor to the real economy. Although valuation is usually driven by short-term human preferences without considering the value of avoiding catastrophic ecosystem change (Limburg et al., 2002), the European Community has tried to address this bias by actively investing in NBS to develop ecosystem services regionally and globally (Lafortezza et al., 2018, Nesshover et al., 2017).

Patterson (2002), using ecological pricing theory, estimated the ecosystem services global value at 25 trillion USD in 1994, and this value rose rapidly from 33 trillion USD in 1997 to 46 trillion USD in 2007 and 125 trillion USD in 2011 (Costanza et al., 2014). Noting that ecosystem goods and services data are scattered or unpublished and difficult to interpret due to the use of incompatible scales and classification systems, de Groot et al. (2002) proposed a standardized classification scheme consisting of 23 functions. An Ecosystem Service Value Database (ESVD) was then developed based on over 1251 value data points (Van de Ploeg et al., 2010).

There is no standardization in the spatial scale for the calculation of ecosystem services. Costanza et al. (2014) suggested that regional aggregates of individual sites and multiple ecosystem services are useful for assessing land use change scenarios, national level aggregates for revising national income accounts and globally for raising awareness of the importance of ecosystem services. Naturvation, an ongoing European Union project, is developing a classification scheme for ecosystem services to categorize NBS at ecosystem and landscape levels (Bockarjova & Botzen, 2017). The next step in this project is to develop an alternative benefit transfer method to apply value to goods from different contexts and to estimate benefit transfer functions for different NBS. It is recognized that values for different ecosystem services vary among regions, related in part to how stakeholders value services at different spatial scales (Brander et al., 2013, Hein et al., 2006).

As pointed out earlier in this report, the biome approach to water security and NBS is being recognized as an emerging method for evaluating both climate change impacts and human alterations to landscapes.

Although 94% of LAC households have access to improved water, 17% lack wastewater treatment services (Fay et al., 2017). Overall, the region spends only 3% of GNP on infrastructure, compared to 4-7% elsewhere. Data on the sewer networks are lacking for most cities in LAC, and most infrastructure management has focused on water supply and quality, paying little attention to flood control in cities (Tellman et al., 2018). Green infrastructure and NBS can be as effective or more so than gray infrastructure as treatment options in both rural and urban areas. Direct comparison of the economics of green vs. gray is extremely complicated because enhanced ecosystem services are often indirect and difficult to put into economic terms (Jaffe, 2010). It is clear, however, that green and gray infrastructures are synergistic, but while gray is designed to meet a set of design criteria, green is adaptable to changes in environmental, economic and social baselines and often undervalued when this added dimension is not considered (Palmer et al., 2015).

Several tools have been proposed to place economic value on green infrastructure and NBS for comparison with gray infrastructure. Examples include the U.S. EPA's cost-benefit analysis with focus on urban areas (**https://www.epa.gov/green-infrastructure/green-infrastructure-cost-benefit-resources**), the Center for Neighborhood Technology Green Values Calculator (**http://greenvalues.cnt.org/national/benefits_detail.php#reduced-treatment**), and the Green Infrastructure Valuation Toolkit from the Natural Economy Northwest program (UK) **http://www.greeninfrastructurenw.co.uk/html/index.php?page=projects&GreenInfrastructureValuationToolkit=true)**. However, it is extremely difficult to apply tools developed for the north temperate zone to the diversity of conditions present in LAC's biomes and urban areas. While greater investment is needed to develop specific tools for the region, Hydro-BID is an excellent start.

There are several approaches for valuing and funding watershed level NBS and green infrastructure. Recognizing that water supplied to 82% of the world's population from upstream sources is highly impaired, Green et al. (2015) developed the Freshwater Provisioning Index for Humans (FPI_h) to map upstream source areas for water utilized downstream. Far too often headwaters are ignored, as are cumulative pollution inputs along a river course, to focus only on water quantity and quality at the point of use downstream. Green et al. (2015) also recognized the importance of water footprint and life cycle analysis and the water poverty index that links human and ecosystem uses of water.

Tellman et al. (2018) emphasized that water conservation in the upper watershed is critical for both water security for LAC cities and for reducing threats from catastrophic floods. They noted that there are significant opportunities for these components of water security for 42 of the largest cities in LAC and their 96 million inhabitants. Most, if not all, cities are likely to require integrated green and gray infrastructure designs to provide adequate protection, but given the limited funding available, cities must be prioritized according to risk threat and the number of people likely to be affected by any actions.

Water funds are organizations that bring together private and civil society stakeholders to promote water security through NBS and sustainable watershed management, fostering long-term watershed conservation for both humans and biodiversity. As of 2014, there were 30 funds in operation or development in LAC (Bremer et al., 2016). Such programs can promote water security in cities and reduce flood risks (Tellman et al., 2018), but there is a current lack of sufficient information to develop NBS that are city and watershed specific. To date, the most effective use of NBS has been for small watersheds, where reduced conservation areas are needed and lower financial input can be expected (Tellman et al., 2018, Palmer et al., 2015). Finally, while assessment of societal values from ecosystem services is well developed in LAC, payments for services have received considerable attention but have received limited attention throughout the region (Balvanera, 2012).

The nexus is the best baseline to develop models integrating water, energy and food into an expandable and adaptable system that can include additional interacting parameters with multiple time and spatial scales. Watersheds and biomes are the ideal landscape units for comparative studies throughout LAC. There is a need, however, to standardize terminology and to apply and fine-tune analytical tools such as MRIO models using real-time data collection from Earth observation systems in order to implement adaptative management scenarios to meet challenges from climate change, catastrophic events, and changes in human demographics and landscapes (Muñoz Castillo et al., 2017, 2019). Given LAC's rapid urbanization, it is critical to recognize that cities are urban biomes with both common structures and functions that must be fine-tuned within the WEF nexus of the "natural" biome where they are situated. Nature based solutions can provide swift and highly effective alternatives to traditional engineering solutions that are costly, have finite life expectancy and are too slow to tackle fast-changing human and environmental conditions. In particular, NBS show great promise for managing the quick-changing nexus of cities of all scales throughout the region.

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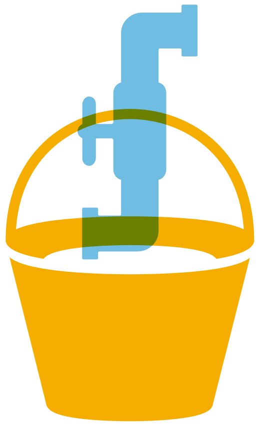

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