Title: Consumer-driven nutrient dynamics in urban environments: the stoichiometry of human diets and waste management

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Abstract

Studies in both terrestrial and aquatic ecosystems have documented the potential importance of consumers on ecosystem-level nutrient dynamics. This is especially true when aggregations of organisms create biogeochemical hotspots through nutrient consumption, assimilation, and remineralization via excretion and egestion. Here, we focused on aggregations of humans in cities to examine how diet and waste management interact to drive nitrogen (N) and phosphorus (P) fluxes into nutrient pollution, inert forms, and nutrient recycling. We constructed six diet patterns (five US-based and one developing nation) to examine N and P consumption and excretion, and explored their implications for human health. Next, we constructed six waste-management patterns (three US and three for developing nations) to model how decisions at household and city scales determine the eventual fates of N and P. When compared to the US Recommended Daily Intake, all US diet patterns exceeded N and P requirements. Other than the "enriched CO_2 environment scenario" diet, the typical US omnivore had the greatest excess (37% N and 62% P). Notably, P from food additives could account for all of the excess P found in US omnivore and vegetarian diets. Across all waste-management approaches, a greater proportion of P was stored or recycled (0 to >100% more P than N) and a greater proportion of N was released as effluent (20 to >100% more N than P) resulting in pollution enriched with N and a recycling stream enriched with P. In developing nations, 60% of N and 50% of P from excreta entered the environment as pollution because of a lack of sanitation infrastructure. Our study demonstrates a novel addition to modeling sustainable scenarios for urban N and P budgets by linking human diets and waste management through socio-ecological systems.

Key words: nitrogen, phosphorus, protein, additives, Ecological Stoichiometry, the Geometric Framework for Nutrition, diet, waste, socio-ecological systems

Introduction

Research in both terrestrial and aquatic ecosystems has documented the important role consumers can play in biogeochemical cycling (Augustine and McNaughton 2006, Vanni et al. 2006, Vaughn 2010, Coetsee et al. 2011). Consumer-driven nutrient dynamics are influenced by organism-specific characteristics, such as diet, and by the physicochemical attributes of an environment, such as ecosystem nutrient demand (McIntyre, et al. 2008). Though myriad studies have documented the profound effects food production and agriculture can have on biogeochemical cycles (e.g., Vitousek, et al. 1997), the influence of human consumers on nutrient dynamics has yet to be explored.

Using the approach of Ecological Stoichiometry (ES), recent work has demonstrated how aggregations of organisms can generate areas of enhanced nitrogen (N) and phosphorus (P) remineralization rates via excretion and egestion, or biogeochemical hotspots, that affect net ecosystem productivity (McClain, et al. 2003, McIntyre, et al. 2008, Meyer, et al. 1983). Ecological Stoichiometry focuses on how organisms and their environment interact to mediate the flow of elements within ecosystems; it considers both the total amount of elements and the ratios of elements in ecological processes (Sterner and Elser 2002). The focus on the flux of N and P through consumers is relevant to both consumer physiology and to ecosystem function because N and P are critical to support individual growth and maintenance, they commonly limit ecosystem productivity, and they can be significant pollutants in terrestrial and aquatic environments. For example, sewage effluent from urban centers can be a primary source of nitrogen (N) and phosphorus (P) to inland and coastal waters; inputs that frequently enhance aquatic productivity, but ultimately degrade aquatic ecosystems and alter the services these systems provide (Gucker, et al. 2006).

Currently, more than 50% of the global population has aggregated in cities and this proportion is growing (Grimm, et al. 2008, Shen, et al. 2008). The accelerated rate of urbanization over the past century has catalyzed changes in biogeochemical processes; many of which are not well understood (Wu 2014). For instance, human diet choices influence the amount and ratio of N and P ingested (Messina, et al. 2004) and waste-management decisions at both the household and city scales alter the proportions of N and P that reach and pollute urban receiving waters (Fissore, et al. 2011, Van Drecht, et al. 2009). However, we do not fully understand the integrated effects of diet and waste management on the flux and stoichiometry of waste. Employing the lens of consumer-driven nutrient dynamics may be useful to elucidate complex relationships among human dietary decisions, waste-management decisions, and urban biogeochemistry.

To survive and reproduce, humans are required to consume complex resources to balance vitamins, minerals, amino acids, carbohydrates, and lipids (Simpson and Raubenheimer 2012). A preponderance of research conducted by nutritionists has produced detailed dietary recommendations for human health (e.g., the Recommended Daily Intake). Studies using the Geometric Framework of Nutrition (GF) have illustrated importance of balancing the acquisition of different nutrients in an organism's diet and how life history traits influence foraging behavior and physiological outcomes when organisms are faced with imbalanced diets (Simpson and Raubenheimer 2012). Of the macronutrients, protein is the most satiating for humans; hence, when confronted with an imbalanced diet, humans will prioritize protein consumption (Gosby, et al. 2011, Gosby, et al. 2014). Consuming meat or animal products, in particular, influences a person's protein intake, and subsequently alters the stoichiometry of their diet and their excreta (Fissore, et al. 2011, Rizzo, et al. 2013). For instance, N is found in high concentrations in protein as amino acids—the main source of N in all foods (Messina, et al. 2004). Phosphorus intake is influenced by consumption of P-rich foods such as cheese, seeds, fish, shellfish, nuts, pork, and poultry (USDA 2013, Wirsenius, et al. 2010). Additionally, P is found in higher concentrations in processed foods that contain phosphate-based preservatives; thus, more P is consumed in diets containing more processed foods (Kalantar-Zadeh, et al. 2010, León, et al. 2013, USDA 2013).

Though the metabolic absorption of N and P in humans varies somewhat by food source, large proportions of N and P are excreted after consumption by adults, especially when eaten in excess (Messina, et al. 2004). The fate of human excretion depends on waste-management decisions at both household and city scales, which in turn are influenced by local environmental conditions, socioeconomic status, and the regulatory environment. In developed regions of the globe, centralized waste management systems (sewers) are common in urban areas, whereas decentralized sanitation systems, such as septic systems, are common in suburban and rural areas. Conversely, in developing regions, 90% of wastewater is still untreated and discharged directly into surface waters (Corcoran, et al. 2010). The way in which waste is managed will influence where and how waste is discharged and the physical form and stoichiometry of waste products that are released into the environment. Therefore, the ability to model the influence of dietary N and P on the abundance and stoichiometry of waste discharged under various management scenarios may be particularly important to predict the potential influence of urbanization and development on the trophic state of aquatic ecosystems.

To promote global economic and ecological sustainability, policies that reflect the complexity of coupled human and natural systems must be developed, tested, and employed (Liu, et al. 2007). Numerous studies have examined either differences in human diet patterns (e.g., Sharkey, et al. 2010, Sisk, et al. 2010) or the influence of anthropogenic activities, such as wastewater disposal or crop production, on the flow of N and P (e.g., Holeton, et al. 2011, Jankowski, et al. 2012); however, few studies have integrated diet patterns and anthropogenic activities to examine consumer-driven nutrient dynamics of humans. In this study we interface published work with modeling approaches to: investigate how human appetite for protein can drive consumption of other nutrients, study how the nutrient content and composition of diets affect the nutrient content of human waste; and, understand how waste-management patterns affect stoichiometry of nutrient pollution and recycling.

METHODS:

2.1 Calculating human diet patterns

Diet patterns describe the type of foods eaten by an individual and the corresponding nutrient levels consumed. Nutrient consumption associated with a diet pattern can be plotted in multidimensional nutrient space. In addition, a diet pattern can be represented within the nutrient space with a line or 'rail' that denotes the ratio of the plotted macronutrients (Simpson and Raubenheimer 2005). Predominant global diet patterns include non-vegetarian (meat > once/wk), semi-vegetarian (meat < once/wk), pesco-vegetarian (fish, shellfish, and dairy/eggs), lacto-ovo vegetarian (dairy/eggs), and vegan (no animal products) (Le and Sabate 2014, Messina, et al. 2004). The amount and type of protein consumed within these diet patterns varies widely around the world (Fig. 1A; Appendix 1 Table A1; FAO 2003, FAOSTAT 2011, Messina, et al. 2004) and influences N and P intake (Fig. 1B).

To understand how the nutrient contents of major food groups differ and might drive differences among different dietary patterns, we obtained nutrient contents of US food groups from the US Department of Agriculture (USDA) National Nutrient Database for Standard

Reference (http://ndb.nal.usda.gov/), hereafter referred to as the USDA Nutrient Database. We used the USDA query function to group food items into major food groups from the abridged list of foods containing a subset of 1000 commonly consumed food items from the entire database (n = 8463; Appendix 1 Table A2). This dataset is a realistic sampling of the American diet and includes processed forms of each food type. For example, the fruit section includes raw fruits, canned fruits, juices, and dried preserved fruits. Nutrient contents of the major food groups are displayed in Fig. 1B and detailed calculations are in Appendix 1 Table A3.1- A3.12. The KFC™ biscuit (USDA item # 21419) data, shown in Fig. 1B, was estimated using a pre-2007 recipe.

We represent the diversity of diet patterns in the US and developing regions by defining the N and P contents of six diet patterns: RDI diet, US omnivore diet, US vegetarian diet, US omnivore diet minus P additives, CO_2-enriched environment scenario diet, and a diet typical of a developing country (the Arba Minch diet discussed below) (Fig. 1C; Appendix 1 Table A4). To determine dietary N content, we converted protein to N using conversion factors provided in the USDA Nutrient Database. When no conversion value was provided, we assumed a 6.25 conversion (16% N by mass).

We assumed that rates of N and P excretion match rates of N and P intake (similar to Baker, et al. 2007, Fissore, et al. 2011). However, this only holds strictly true when people are neither gaining nor losing mass and human populations are not likely to be in perfect steady state. Therefore, we quantified potential sources of error due to this assumption. Because our analysis focused on annual estimates, we do not account for long-term demographic changes such as increasing or decreasing population size, or mortality and replacement. We estimated the amount of N and P accumulated in human biomass each year due to 1) adult weight gain, and 2) growth of youths. Both of these would cause N and P excretion rates to be lower than N and P intake rates. For this exercise, with the goal of assessing the potential error due to assuming steady state, we focused on the United States where all necessary data are readily available. Compared with total dietary intake, N and P accumulation in the US due to adult weight gain plus growing youths is small (1-2%). As a result, we chose not to decrease our estimates of human excretion because other sources of uncertainty in our analyses are much larger, especially in our estimates for developing nations. For a complete description and calculations, see Appendix 1, Table A6.

2.1.1 RDI diet

The Food and Nutrition Board of the National Academies of Science and the USDA set recommendations for human diets by assessing the average daily level of intake that is sufficient to meet nutrient requirements for a majority of healthy people (97-98%; NRC 2006). These Recommended Daily Intake (RDI) values vary by age and gender based on studies of human nutrition and physiology. We used the RDI values as a base diet (hereafter referred to as the RDI diet) to which we compare other, more realistic diets. While the RDI diet is a recommendation that may not represent actual diet patterns in the US, it has been in use in various forms since the 1960s as a benchmark.

2.1.2 Standard US omnivore and vegetarian diets

The predominant US diet pattern is omnivorous and includes protein in excess of daily recommended requirements (FDA 2003, Messina, et al. 2004, Rizzo, et al. 2013). Approximately 93-97% of Americans have a diet pattern that includes meat while the remaining have a diet pattern consistent with one form of vegetarian eating (Haddad and Tanzman 2003, Stahler 2006). The most common vegetarian diet pattern in the US is lacto-ovo which includes animal products, but no animal flesh (US and Canada; Messina, et al. 2004, Rizzo, et al. 2013). The inclusion of animal products in the lacto-ovo diet can increase protein consumption over more restrictive vegetarian diets such as vegan (Clarys, et al. 2014), but protein consumption by lacto-ovo vegetarians and vegans in the US is similar and somewhat lower than that of omnivores (Messina, et al. 2004). In general, the US omnivore diet includes between 50%-100% more protein than the RDI diet and the lacto-ovo vegetarian diet includes between 30-45% more protein (FDA recommended diet is 56 g protein/day for men and 46 g/day for women; Messina, et al. 2004). Adult US females consume an average of 1128 mg P/day and adult US males an average of 1602 mg P/day, while the RDI diet recommends 700 mg P/day for adults (Messina, et al. 2004).

We calculated diet pattern composition and stoichiometry for the US omnivore diet by combining the FDA Total Diet Study 2003 food consumption data (Appendix 1 Table A5.1) with nutrient data from the USDA Nutrient Database (Appendix 1 Table A5.2). The Total Diet Study is an ongoing program that monitors the amount of target nutrients and toxins in the average American diet by combining detailed diet surveys with analysis of store purchased products. Food consumption amounts for each Total Diet Study food were taken from the 'Total US' population calculations in the Total Diet Study (a composite average intake for all ages, male, and female). Consumption values for each item in the Total Diet Study were then matched to nutrient data from the USDA Nutrient Database to obtain protein, phosphorus, carbohydrate, and lipid content. When a Total Diet Study food list item had more than one listing in the USDA Nutrient Database (e.g. duplicate brands), the product with the least amount of additives or fortifications was used, and generic was always selected over name brand when available. Two foods were removed from the Total Diet Study food list that were not found in the USDA

Nutrient Database (Tuna noodle casserole homemade TDS # 272, Macaroni salad grocery/deli TDS # 346).

To estimate the nutrient content of a standard US vegetarian diet, we replaced 65 items in the Total Diet Study food list that contained meat with vegetarian alternatives (Appendix 1 Table A5.3 and A5.4). Comparable vegetarian items in the USDA Nutrient Database were used when available (e.g. chili con carne replaced by chili with beans). Vegetarian meat-style products were used as a substitute for meat when available (e.g. pork bacon replaced by veggie bacon). There were 12 meat items that consisted of unprocessed meat with no vegetarian alternative that were replaced by soft and firm tofu products (six of each type).

To check the accuracy of our phosphorus content calculations in the US omnivore diet, we compared them to the phosphorus content as estimated by the Total Diet Study Market Basket analysis (Appendix 1 Table A5.5).

2.1.3 US Omnivore diet without P additives

The average US diet contains a high proportion of processed foods (> 70%; USDA 2003), many of which contain phosphate-based preservatives and synthetic sweeteners high in P content (León, et al. 2013). Phosphorus additives are used for leavening, color and moisture retention, caking prevention, and flavor enhancement (Kalantar-Zadeh, et al. 2010, León, et al. 2013). They are found disproportionately among processed foods and are more prevalent in prepared frozen foods, dry food mixes, and packaged meat (León, et al. 2013). Much of the P found in additives is in the form of inorganic salts that more readily dissociate and absorb in the intestine than organic protein bound forms of phosphorus (Kalantar-Zadeh, et al. 2010), but there is currently no reliable method for distinguishing phosphorus content from organic and inorganic sources in food (Cupisti, et al. 2004, Murphy-Gutekunst and Uribarri 2005)

To estimate the contribution of P to the standard US omnivore diet by additives, we used calculations from León et al. (2013). The authors determined the prevalence of P-containing additives in top selling foods in grocery stores in northeast Ohio. They prepared sample meals of foods with and without P additives and calculated that the average daily P consumption would decrease to 59% of original consumption if meals without P additives were selected. We used this estimate to reduce dietary P in the US omnivore diet (section 2.1.2), but kept dietary N constant.

2.1.4 CO_2-enriched environment scenario diet

We developed the CO_2-enriched environment scenario diet (hereafter referred to as the CO_2 diet) to model the increase in atmospheric CO_2 associated with climate change that is expected to alter the nutritional quality of plants and animals and affect human diet (Robinson, et al. 2012). We used an estimated 54.4% increase of the carbohydrate: protein ratio of plant products in human diets caused by increases in atmospheric CO_2 from Robinson et al. (2012) and adapted by Raubenheimer et al. (2014). Although elevated CO_2 may also affect plant P content, relatively few studies have examined this (none, to our knowledge, in food crops), and those studies have shown divergent results (Gifford 2000). We therefore kept the P content of the diet unchanged in this scenario. We excluded animal products because there is not a good estimate for how their stoichiometry will change as plant stoichiometry changes, and because the protein and P contents of animal feed are explicitly managed in most commercial animal-feeding operations.

To calculate how the CO_2-enriched atmosphere would change the US omnivore diet (section 2.1.2), we used the ratio of protein and carbohydrate from plant products as calculated from the FAOSTAT 2011 US diet (35% and 94%, respectively). We then determined that an increase in the carbohydrate:protein ratio of 54.4% in plant products would result in an overall decrease of dietary protein from 14.4% to 12.7%. The protein leverage hypothesis predicts tight regulation of dietary protein by humans such that when faced with protein-poor diets, humans will over-consume carbohydrates and lipids to maintain a similar protein intake as when eating balanced diets (Simpson and Raubenheimer 2005). Gosby et al. (2014) estimated that, in diets below 20.9 % protein, the slope of the relationship between dietary protein and total energy intake is -0.47 MJ % protein^{-1}. Using this relationship, we calculated that on the CO_2 diet, total food consumption would increase by 789 kJ day^{-1}. Assuming the same P concentration as the US omnivore diet, this resulted in an increase to 0.46 kg P capita^{-1} year^{-1}. Dietary N decreased to 3.87 kg N capita^{-1} year^{-1}. Because consumption of protein-diluted foods will likely be exacerbated by the rising cost of protein-rich foods, this is likely a conservative estimate (Brooks et al. 2010).

2.1.5 Arba Minch, Ethiopia diet

We based a small-city, developing country diet pattern on Arba Minch, Ethiopia because it was one of the only cities in a developing region with a complete dataset for dietary N and P in addition to N and P fluxes through the city waste management (Meinzinger, et al. 2009). That study used protein (and hence N) content reported directly by FAOSTAT (2008), and calculated an estimate of dietary P using protein:P relationships for animal and vegetable protein reported in Jönsson & Vinneras (2004).

2.2 Calculating waste-management patterns

Similar to diet patterns, we defined and compared different patterns of waste management. We defined three waste-management patterns based on the US, starting with the current US national average. To focus on hotspots of nutrient loading in urban areas, and investigate opportunities for improvement, the second US pattern is for a hypothetical metropolitan area that employs some of the most common current waste-management practices. The third US pattern is a future scenario that envisions a high-tech environmentally-focused metropolitan area. In exploring waste management for developing nations, we found a wide variety of different practices. Rather than attempt to collapse these diverse practices into a single average pattern representing all developing nations, we developed three patterns as examples to illustrate some of this diversity. These patterns are based on data for current conditions in: Arba Minch, Ethiopia (population 80,000); Jakarta, Indonesia (population 9.6 million); and rural Indonesia.

To calculate the fates of N and P under each waste management pattern, we developed a model that tracks N and P fluxes from human excretion, through various transformations in the waste streams, to their eventual fates as pollution (N and P loading to ground or surface waters), inert forms, or recycling (deliberate return of N and P to the food-production system, usually to cropland or gardens). To parameterize each waste-management pattern, we combined the best available values from peer-reviewed literature and publications by governmental and non-governmental organizations to represent general conditions. We describe the waste-management patterns and model briefly here, with additional details, rationale, and literature background presented in Appendix 2.

2.2.1: Current US average

Because the best empirical data for human diet, waste management, and nutrient loading are available at the national scale, we developed a pattern describing the current US average practices to compare our results with prior work and put them in context. In the US, we assumed that all people have access to sanitation infrastructure: 75% of the population is served by waste water treatment plants (WWTPs) and 25% by septic systems (USEPA 2002). Of those served by WWTPs, 1% are served by primary treatment (which removes solids and 10% of sewage N and P), 35% by secondary treatment (which utilizes microorganisms to digest sewage, and removes 35% of N and 45% of P), and 64% by tertiary treatment (which uses various additional technologies, and removes 80% of N and 90% of P; USEPA 2008, Van Drecht, et al. 2009). Some N removed in secondary or tertiary treatment is removed by microbial processes to gaseous forms (N_2 or N_2O), and N and P are also both removed in solids that are landfilled (37%), incinerated (22%), or processed into biosolids and land-applied (41%; Idris, et al. 2010). The N and P not removed by WWTPs remains in the sewage effluent and is either discharged to ground or surface waters (92.5%) or reused to irrigate cropland, turf grass, or other vegetation (7.5%; USEPA 2012).

We parameterized all current septic systems as conventional systems consisting of a septic tank and a drainfield. In the septic tank, 15% of sewage N and 25% of P are retained in solids (also called septage), and the remainder is in septic effluent that is discharged to the drainfield (Viers, et al. 2012, WERF 2009). Septic solids are pumped out periodically and transported to a WWTP or land-applied; we assigned half of septic solids to each disposal pathway (see Appendix 2). Some N and P in septic effluent are attenuated in the soil beneath the drainfield by denitrification, biological uptake, or soil adsorption, and the remainder leaches into groundwater. Attenuation rates differ greatly depending on soil types, seasonal water tables, climate, and other biophysical factors as well as the design and condition of the septic system (Withers, et al. 2014). We were unable to find empirical data estimating an overall average attenuation rate; for all waste-management patterns, we assigned 50% of N and 75% of P in septic (or latrine) effluent to soil attenuation (see Appendix 2).

2.2.2: Hypothetical US metro area

To focus on hotspots of dietary N and P loading to on urban receiving waters, we created a hypothetical waste-management pattern more typical of a current US city and its immediate metropolitan area. We chose some of the most common urban waste-management technologies and practices to examine opportunities for improvement. We began by setting the boundary for our hypothetical metropolitan area such that 10% of the population is in peri-urban areas served by septic systems and 90% in the sewershed, served half by a secondary-treatment WWTP and half by a tertiary-treatment WWTP. For both WWTPs, we assigned WWTP solids half to landfill and half to land application, since these are the two most common disposal methods in the US, and assigned all WWTP effluent to discharge. For septic solids, we assigned 75% to WWTPs and 25% to land application, since the septic-served peri-urban areas are likely to be relatively densely populated with less access to farmland for land-applying solids (USEPA 2000).

2.2.3: High-tech environmentally-focused US metro area

To explore the possibilities for improving urban waste management, we also envisioned a future scenario with widespread adoption of current technologies that reduce N and P pollution and increase N and P recycling. We kept the same geographic structure as in the hypothetical metropolitan area, with 90% of the population served by WWTPs and 10% by septic systems. However, we upgraded all WWTPs to tertiary treatment, assigned all WWTP solids to land application, and assigned 92% of WWTP effluent to irrigating cropland (the same reuse rate as is currently practiced in Phoenix, Arizona, USA; Lauver and Baker 2000). We upgraded all septic systems to use a recirculating gravel bed (the highest published denitrification rate) between the septic tank and the drainfield. This system still removes 15% of N and 25% of P in solids, which we assigned entirely to land application, and additionally denitrifies 92% of N remaining in in septic effluent (78% of total septic N inputs) to N_2 (WADEQ 2014).

2.2.4: Arba Minch, Ethiopia

We based a small-city waste-management pattern on Arba Minch, Ethiopia (population 80,000) because it is one of the only cities in a developing region for which a complete N and P budget has been published (Meinzinger, et al. 2009). In Arba Minch, 16% of the population lacks access to sanitation infrastructure and practices open defecation (which we assigned to direct discharge into the environment), 6% are served by septic systems, and 78% are served by pit latrines. In septic systems, 8% of N and 17% of P are retained in solids; in latrines, 17% of N and 28% of P are retained in solids. However, all solids pumped out of septic tanks or latrines are trucked out of town and illegally dumped into roadside ditches and other environments (Meinzinger, et al. 2009).

2.2.5: Jakarta, Indonesia:

We based a large-city waste-management pattern on data for Jakarta, Indonesia (population 9.6 million). One quarter of the population lacks access to sanitation infrastructure, 3% are served by a primary-treatment WWTP, and 72% are served by septic systems (Corcoran, et al. 2010, WHO/UNICEF 2008). For people with no sanitation, we assigned 10% of the waste to direct reuse in urban gardens and the remaining 90% to direct discharge (e.g. toilets connected to storm drains or discharging onto ground areas; Strauss 2000). Because data for solid waste disposal were not available for Indonesia, we used a study for China, which found that 3% percent of WWTP solids are incinerated, 49% are landfilled, and 48% are land-applied (Chen, et al. 2012). For septic systems, 5% of septic solids are transported to the WWTP while the remaining 95% are dumped illegally (Moersid 1998).

2.2.6: Rural Indonesia

To investigate rural villages in developing nations as smaller local hotspots of dietary N and P loading, we based a village waste-management pattern on data for rural Indonesia, where 39% lack access to sanitation infrastructure and practice open defecation, and 61% are served by pit latrines (WHO/UNICEF 2008). Because many villagers report a strong cultural preference for defecating directly in the river (Robinson 2005), we assigned all people with no sanitation to direct discharge. For latrines, we assigned N and P to solids in the same proportion as for Arba Minch. As data for septic solids disposal were not available for rural Indonesia, we used a study for rural Vietnam, which found that 90% of latrine solids are land-applied to cropland (Jensen, et al. 2008). Because many rural areas do not have a septage management facility or WWTP nearby, we assigned the remaining 10% to direct dumping/discharge.

RESULTS:

In comparison to the RDI diet, all US diet patterns had an excess of N and P (Fig. 1C), with the typical US diet (US omnivore) having the greatest excess for current diets (1.1 kg N capita^{-1} y^{-1} (37%) and 0.16 kg P capita^{-1} y^{-1} (62%)). The US omnivore diet is a weighted average of all age groups and the RDI recommendations are for adults; therefore, the actual differences are slightly larger than our estimates. Replacing meat with vegetarian substitutes reduced nitrogen to only 17% excess of the RDI diet, but maintained similarly high levels of P. Conversely, removing P additives from the US omnivore diet had strong effects on dietary P. The “No P additives” diet reduced dietary P by almost 0.16 kg capita^{-1} y^{-1} or 50% relative to standard US omnivore and vegetarian diets and to a similar level recommended by the RDI diet (700 mg/day). The diet pattern we employed to examine diets in developing countries, the Arba Minch diet, had a similar P content to the US omnivore and vegetarian diets (0.41 kg vs. 0.42 kg and 0.41 kg P capita^{-1} y^{-1}, respectively), but was deficient in N relative to the RDI diet (2.5 v 2.9 kg N capita^{-1} y^{-1}). The future-scenario CO_2 diet had a similar N:P ratio as the US vegetarian diet, but had a 9.5% higher per capita P consumption than any of the other diet patterns and was also the diet with the greatest overall consumption of calories.

Waste-management patterns showed greater variability in N and P fluxes to water pollution and nutrient recycling (Fig. 2, Table 1). In the current US average, we estimated that 37% of N and 24% of P from human diet contributed to water pollution, and 11% of N and 28% of P were recycled. Our hypothetical metropolitan area had somewhat higher pollution fluxes than the US average (43% of N and 32% of P) despite having a lower proportion of the population on septic systems (10% vs. 25%), because we split the WWTP-served population evenly between a secondary-treatment and a tertiary-treatment WWTP. Recycling was similar to the US average, at 12% of N and 32% of P. The high-tech metropolitan area scenario substantially reduced both N and P pollution (2% of N and 3% of P from diet), and tripled recycling to 36% of dietary N and 92% of P.

In developing nations, the three waste-management patterns had very similar proportions of dietary N and P becoming water pollution (64-65% of N and 52-54% of P, Fig. 2, Table 1). In contrast, they had very different proportions and sources of recycled N and P: there was no recycling in Arba Minch; 3% of both N and P were recycled in Jakarta, Indonesia via direct reuse of human waste; and 9% of N and 15% of P were recycled in rural Indonesia via fertilizing with latrine solids.

Waste management also shifted the stoichiometry of many pollution and recycling fluxes relative to the diet fluxes. Figure 3 shows the stoichiometry of total water pollution and total recycling for each waste-management pattern under three diet patterns. Overall P removal was higher than N removal (20-100%) in all waste-management patterns except the high-tech metropolitan area, resulting in an overall lower N:P ratio of fluxes to water pollution than the N:P ratio of human waste. However, there was substantial variability in the stoichiometry of different fluxes to water pollution (Table 2c). Groundwater loading had higher N:P ratios than surface-water loading (except in the high-tech metropolitan area with denitrifying septic technology), driven by higher soil attenuation rates for P than N. Surface-water loading (WWTP effluent, direct discharge, and/or solids dumping/discharge) in the three developing-nations waste-management patterns had lower N:P ratios than in the three US patterns. This was due not only to the direct discharge of untreated (and unmodified) human waste, but also to the dumping/discharge of septic and latrine solids, which have a lower N:P ratio than human waste.

The N:P ratio of nutrient recycling fluxes was even more variable (Table 2c). Septic and latrine solids in all waste-management patterns had an N:P ratio roughly half that of human waste since a greater proportion of P than N biochemically partitions into solids, and the N:P ratio of WWTP solids was even lower (except in Jakarta, which only uses primary treatment), due to denitrification in secondary or tertiary WWTP. Direct reuse of human waste in Jakarta did not change its stoichiometry, and WWTP effluent reuse in the US had a substantially higher N:P ratio than human waste, again because a greater proportion P than N was removed as solids in the WWTP.

DISCUSSION:

Diet choices and waste-management approaches both exert substantial influence on the amounts and ratio of N and P moving through urban areas. In the US, we found that people tended to consume excess N and P, which exacerbated nutrient pollution. Excess P could be explained by P additives and is likely related to increased consumption of processed foods and a related shift to low protein, high carbohydrate and fat diets in the US. The CO_2 diet further diluted protein in diets, increased processed food consumption, and therefore increased consumption of P additives. Existing waste management practices in the US released a substantial fraction of N and P from human excreta into receiving waters, and of the fraction removed, more was moved into non-recoverable inert pools than recycled to the food-production system. In developing nations, the little data that are available suggest that dietary N and P intake were not adequate to maximize human health. In these regions, lack of waste management infrastructure and cultural preferences for open defecation resulted in the majority of N and P becoming water pollution. Comprehensively, our data demonstrate there are considerable opportunities to improve both diet choices and waste-management systems in ways that would simultaneously decrease nutrient pollution and improve human health by lowering dietary P (which can have negative effects on kidney function) and decreasing obesity.

Multidimensionality of nutrients in human diets

In the typical US diet, people ate an excess of both N and P relative to the diet recommended by the RDI (Fig. 1C). In the US, there has been increased consumption of 'ultra-processed foods' or 'empty calories' that are high in carbohydrates and fats, but low in protein. Indeed, despite having a moderate excess of total dietary N, which is caused by excess protein consumption, people in the US over-consumed carbohydrates and lipids to a greater extent than protein. This trend has increased since the 1960's and is predicted to further increase under scenarios of CO_2 enrichment due to an increased soluble carbohydrate to protein ratio in food crops (Fig. 1A; Raubenheimer, et al. 2014, Simpson and Raubenheimer 2005).

There is growing evidence that a decrease in the ratio of dietary protein to carbohydrates and fats, in conjunction with a dominant protein appetite in humans, contributes to excess energy consumption and the obesity epidemic (Gosby, et al. 2011, Gosby, et al. 2014). Our study illustrates how P will also be over-consumed when humans eat diets largely consisting of ultra-processed foods because these foods have disproportionately high P to protein ratios due to ubiquitous use of P additives (Kalantar-Zadeh, et al. 2010, León, et al. 2013). When protein is diluted (i.e., below the intake target ratio for protein to carbohydrates and fats as shown for the US and the CO_2 diet in Fig. 1A), humans will over-consume total energy (and P in this case) in order to maintain a similar dietary protein content. In our diet patterns, the US vegetarian diet had a slightly decreased N content compared to the omnivore diet (Fig. 2C), suggesting that overall energy consumption would increase for vegetarians, in turn increasing total nutrient content of waste streams.

Over-consuming P not only increases nutrient pollution, but also negatively impacts human health. While most research in nutritional physiology and ecology has focused on nutrients as being limiting, over-consuming nutrients can also have negative consequences (Raubenheimer et al 2005). In humans, high dietary P may lead to kidney disease and is a considerable problem for patients already diagnosed with kidney disease (Gutiérrez, et al. 2010). Our diet pattern comparisons suggest that if US omnivores were to cut out dietary P from additives, they would have a dietary P content similar to that of the RDI recommended diet (Fig. 1C). Unfortunately, in the US today it can be difficult to avoid high P foods because the amount of dietary P from P additives is typically not given on nutrition labels and foods without P additives are more expensive. León et al. (2013) estimated an average increase of $2.00 USD per day per person to purchase commonly eaten foods in Ohio grocery stores that did not contain P additives.

It is important to note that our reliance on US diet data provides extreme examples of protein to non-protein food consumption and processed food intake. When compared to other major regions of the world, the US has the highest carbohydrate and fat intake (Fig. 1A). However, global diet patterns show considerable variation among regions in the proportion of energy intake from protein and non-protein (carbohydrate and lipid) sources (Fig. 1A). In developing regions, protein consumption tends to be less than the protein target for humans and below the RDI recommended amount of protein, as is the case for Arba Minch (Fig. 1C). The RDI diet also represents a recommended protein intake that is below 10% of energy intake and thus, provides a low-end intake recommendation for protein and N. Global diet patterns also show considerable variation in the contribution of various food staples (Fig. 1B) to total diet (Charrondiere, et al. 2004, FAOSTAT 2011). Because foods are multidimensional, eating different amounts of various food staples simultaneously influences N and P intake, as well as other nutrients. For example, a diet pattern that includes a high portion of nuts will have a similarly high N content relative to a diet high in eggs or fish, but about a 60% increase in P content (Fig. 1B).

Sanitation and nutrient management

Although human wastes can be a strong driver of ecosystem nutrient dynamics, waste-management practices are primarily designed to improve sanitation and reduce disease by reducing contact with human wastes (WHO/UNICEF 2008) and do not necessarily address nutrient pollution. In developing nations, the majority of N and P from human waste becomes pollution (64-65% of N and 52-54% of P, Fig. 3, Table 1), even though 61-84% of people have access to improved sanitation. In Arba Minch and Jakarta, the practice of illegally dumping septic and latrine solids does reduce contact with human waste, but contributes directly to nutrient pollution. In the process of constructing new sanitation infrastructure in developing regions, there is an opportunity to design systems that reduce nutrient pollution as well as disease. Human wastes can provide valuable fertilizer, as is seen in rural areas where latrine solids are widely applied to cropland, and increased N and P recycling can assist local food production as well as reduce global nutrient scarcity (Corcoran, et al. 2010, Cordell, et al. 2009).

In developed nations, some wastewater treatment standards are also designed to protect downstream ecosystems by limiting effluent biological oxygen demand or nutrient concentrations (e.g., USEPA 2008). These practices have substantially reduced nutrient pollution; under current US waste management practices, we estimate that 37% of N and 24% of P in wastewater contribute to water pollution (Fig. 3, Table 1). However, these practices do not prioritize nutrient recycling. In the US, 52% of N and 48% of P in wastewater are moved into non-recoverable inert pools, while only 11% of N and 28% of P are recycled (Fig. 3, Table 1).

Estimating human waste contribution to total wastewater

Consumer effects are only one part of nutrient dynamics; in addition to human waste, wastewater N and P also comes from food waste, detergents and household products, and commercial and industrial processes. To estimate the proportion from human waste, we combined the typical US omnivore diet pattern and the current US average waste-management pattern. Multiplying our per-capita values by the US population of 314 million (US Census 2012), we estimated that human waste contributes 331 Gg y^{-1} N and 26 Gg y^{-1} P to WWTP effluent nationally. This is 55% and 20% of total WWTP effluent fluxes (national wastewater loading = 600 Gg y^{-1} N and 130 Gg y^{-1} P; Van Drecht, et al. 2009). To our knowledge, ours is the first estimate of the proportion of N and P in wastewater loading from human waste, so we are unable to validate our results against existing data. We had not initially expected to find this large of a difference in the proportional contribution of human waste to total effluent N vs. P, but this estimate is consistent with other work that estimated P fluxes in household greywater (from detergents and other household products) were approximately equal to P fluxes from human waste, while N fluxes in greywater were < 15% of N fluxes from human waste (Fissore, et al. 2011).

Potential environmental effects of changing diet and waste management

Unlike most animals, humans have the ability to deliberately alter diet and waste-management practices in order to decrease nutrient pollution and protect aquatic ecosystems. Currently, however, diet changes are on a trajectory for increasing N and P consumption and hence increasing water pollution. In developing regions, improving diet focuses on increasing protein (and hence N) intake to adequate levels, which likely will also involve increasing total food consumption and thereby increasing P consumption. In the US, the future scenario of the CO_2 diet contains 110% of the P intake and 97% of the N intake in the current US omnivore diet. If the entire US population of 314 million people were to shift to this diet, we estimate that national P consumption and excretion would increase by 13 Gg y^{-1} P, and water P pollution would increase by 3.0 Gg y^{-1} P (under the “current US average” waste patterns). Conversely, we estimate that national N consumption and excretion would decrease by 44 Gg y^{-1} N, and water N pollution would decrease by 16 Gg y^{-1} N.

Fortunately, there are possibilities for improving diet in developed countries to reduce N and P fluxes. Shifting from the typical US omnivore diet to the RDI diet would decrease N and P water pollution to 73% and 62% of current fluxes, respectively (Table 2). If the entire US population were to shift to the RDI diet, we estimate that national nutrient consumption and excretion would decrease by 342 Gg y^{-1} N and 50 Gg y^{-1} P. This would reduce national water pollution by 127 Gg y^{-1} N and 12 Gg y^{-1} P under current (US average) waste management. Comparing to national N and P loading, which includes non-dietary nutrient fluxes to wastewater, these reductions are equal to 22% of N and 10% of P in current WWTP effluent (600 Gg y^{-1} N and 130 Gg y^{-1} P; Van Drecht, et al. 2009). Focusing on urban receiving waters and applying our hypothetical metropolitan area waste-management pattern to a population of 5 million people (similar to the Boston or Atlanta metropolitan areas), shifting to the RDI diet would reduce the N and P fluxes in effluent by 2340 Mg y^{-1} N and 255 Mg y^{-1} P.

In contrast to diet flexibility that is limited by minimum human nutrient requirements, there is greater potential to reduce water pollution by improving waste management. Shifting to the high-tech metropolitan area pattern would reduce N and P pollution to 4% and 8% of current fluxes (hypothetical metropolitan area pattern), respectively. Again considering a hypothetical population of 5 million people, this shift would reduce urban water pollution by 8260 Mg y^{-1} N and 610 Mg y^{-1} P. Simultaneously improving diet and waste management (RDI diet and high-tech metropolitan area waste management) would reduce N and P pollution to 3% and 5% of current fluxes (US omnivore diet and hypothetical metro area waste management), and reduce urban water pollution by 8355 Mg y^{-1} N and 635 Mg y^{-1} P. If the US diet were to change to the CO_2 diet, the increase in P excretion could still be more than offset by improvements in waste management for a net decrease in water pollution.

Just as many consumers increase primary productivity by moving nutrients across ecosystem boundaries, human waste management has the potential to return essential N and P to the food production system, thereby reducing the demand for mined or manufactured fertilizer. The effects of shifting diet would come primarily from upstream effects (nutrient losses in agriculture and food wastage), which we did not attempt to quantify in this study. Shifting from the hypothetical metropolitan area waste-management pattern to the high-tech waste-management pattern would triple N and P recycling (306% and 290%, respectively, of current recycling fluxes), bringing recycling up to 36% of dietary N and 92% of dietary P (Fig. 3, Table 1). For a metropolitan area population of 5 million people, recycling fluxes would increase by 4870 Mg y^{-1} N and 1265 Mg y^{-1} P.

It is important to recognize that dietary N and P fluxes are only one part of global nutrient use for food production. Even recycling 100% of dietary P would not be sufficient to eliminate the reliance on mined (and increasingly scarce) P fertilizer (Cordell, et al. 2009). However, recycling nutrients from human waste is an essential tool, complementary to increasing agricultural nutrient-use efficiency and decreasing food wastage, in closing N and P cycles and reducing global demand for mined and manufactured fertilizer.

Stoichiometric implications of diet and waste management patterns

Humans and other consumers drive nutrient dynamics through not only the amounts of N and P in their wastes but also the ratio of these elements. To understand stoichiometric effects on aquatic ecosystems, we compared the N:P ratio of water-pollution fluxes (Table 2C) to the Redfield ratio: the measured ratio of C:N:P in ocean waters and phytoplankton (Redfield 1958), which is commonly used as a metric of environmental homeostasis in aquatic ecosystems. Phytoplankton growth is generally considered P-limited when the N:P ratio (by mass) of available nutrients is > 7.2 and N-limited when the ratio is < 7.2. We estimated the N:P ratio of diet-derived WWTP effluent to be higher than the Redfield ratio under current US average waste management (Table 2), ranging from 11 under the CO_2 diet to 20 under the no-P-additives diet. In areas where human sewage is a primary driver of nutrient loading, we expect phytoplankton growth could shift from N to P limitation, however, the ecosystem-level results will be highly context dependent. In some cases, freshwater P limitation could favor communities dominated by green algae instead of N-fixing cyanobacteria, and could reduce the incidence of cyanotoxin production. In other cases, marine P limitation could stimulate the production of carbon-rich toxins by dinoflagellates (Van de Waal, et al. 2014).

In the developing-nations waste-management patterns, by contrast, nutrient loading to surface water overall had an N:P ratio < 7.2. Directly discharged human waste under the Arba Minch diet had a lower N:P (= 6.2) than US diets, below the Redfield ratio, and the dumping or discharge of septic and latrine solids (N:P = 3.8) lowered the overall N:P ratio even further. This could shift phytoplankton growth to N limitation, favoring N-fixing cyanobacteria and increasing the incidence of cyanotoxin production. This is especially troubling in areas where the same waterbodies are used for waste disposal and drinking water. Cyanotoxins can be fatal to humans and livestock and, unlike many pathogens, cannot be removed by particulate filtration or detoxified by boiling the water (Lindon and Heiskary 2009, Weirich and Miller 2014).

Stoichiometry must also be considered when recycling diet-derived N and P to the food-production system. To effectively reduce the demand for mined and manufactured fertilizer, recycled nutrients must be applied in amounts and ratios that crops can use; excess nutrients will be wasted and likely contribute to water pollution. However, current recycling practices often do not effectively manage N and P. In some cases, nutrients are ignored: WWTP effluent is used for irrigating cropland to alleviate water scarcity, but fertilizer application is not reduced to account for the N and P in effluent (Lauver and Baker 2000). In other cases, only N is considered: land-applied solids are regulated to measure and balance N inputs with crop N requirements, but P is not measured or regulated (USEPA 1994).

We compared the N:P ratios of dietary fluxes to nutrient recycling with the N:P ratios of crop nutrient requirements for a food crop, corn, and a forage crop, bromegrass hay (N:P = 5.2 and 9.6, respectively; USDA 2014). The N:P ratios of recycled materials span a wide range; for example, under the scenario of the RDI diet and high-tech waste management, we estimate that WWTP solids N:P = 2.5, septic solids N:P = 6.7, and WWTP effluent N:P = 22.5 (Table 2C). In the developing-nations patterns with the Arba Minch diet, the largest recycling flux is latrine solids used as fertilizer in rural Indonesia (N:P = 3.8); in Jakarta, WWTP solids N:P = 5.7, and direct reuse of human waste N:P = 6.2. If recycling practices continue to ignore nutrients or be managed based solely on N, applying material with a lower N:P than crop requirements has the potential to over-apply and waste much of the recycled P. Since non-dietary fluxes to wastewater have a substantially lower N:P ratio than human excreta, we expect that the actual N:P ratio of these recycled materials would be lower than our estimates, and the mismatch with crop requirements more severe. Even when applying a material with higher N:P than crop requirements, effectively recycling P will require measuring the amount of P in recycled materials and reducing P fertilization accordingly. It is essential that recycling efforts explicitly measure and balance both N and P application; otherwise, they will waste much of the nutrients they are attempting to conserve.

Key data gaps in human diet and waste management patterns

Our work highlights the dearth of data available for current diet and waste management practices in developing nations. A key limitation of our analysis of developing country diets is that data from FAOSTAT are for food availability, not actual consumption, and dietary P contents are currently only estimated coarsely using relationships to protein contents (Meinzinger, et al. 2009). Therefore, the comparisons we make between the FAOSTAT diet estimates and the US diet survey data should be used as course comparisons made with the best available data. Waste-management data are reported using categories such as "improved sanitation" that aggregate practices with very different effects on N and P cycling; furthermore, there is evidence that many existing sanitation facilities are unused or poorly maintained (Corcoran, et al. 2010). Even less information is available to construct practical scenarios for improving these conditions. More research is essential in developing nations, because existing developed-nations solutions are often misaligned with the socio-economic, political, cultural, and biogeophysical realities of these regions.

Even in the US, there are a number of uncertainties in the parameters used to identify consumption and fates of N and P under different waste management approaches. For example, our calculation of the US vegetarian diet by replacing meat food items with their vegetarian alternative without adjusting consumption quantities may underestimate the type and quantity of non-meat food items consumed by vegetarians in the US. In relation to nutrient fates, the proportion of N and P that leaches from septic and latrines to groundwater varies widely (based on soil type, water table, climate, and other biophysical factors in addition to septic or latrine design and condition) and is not yet well characterized. We assigned intermediate soil attenuation rates of 50% and 75% for N and P, respectively, for all waste-management patterns. If soil N attenuation were 20% or 80% (as in Withers, et al. 2014), total water N pollution would increase or decrease 17%, respectively, in the US average waste-management pattern. In contrast, our estimates showed greater sensitivity to changes in soil P attenuation; if soil attenuation were 10% or 100% (as in Withers, et al. 2014), total water P pollution would increase 50% or decrease 19%, respectively, for the US average.

Most research in consumer-nutrient dynamics, including this study, has been limited to understanding the influence of consumers on N and P cycling. However, other elements including, but not limited to C, S, K, and Ca, are important components of human diets, enter the environment via waste disposal systems, and may influence community structure and ecosystem function in fresh waters. Examining links among diet patterns, wastewater treatment, and the cycling of other elements will promote a greater understanding of consumer-driven nutrient dynamics and the influence of urbanization and freshwater resources.

Conclusions

As with many consumers, human diet and excretion can be a major driver of N and P dynamics. Yet, unlike other organisms, there are substantial data describing human diet and waste-management decisions that allowed us to model the flux of N and P through human populations. Dietary decisions also result in land use change to meet dietary demands. While beyond the scope of this paper, studies on the relative influence of human excretion and changes in agriculture will be important for understanding the effects of diet decisions on N and P dynamics at a regional and global scale. The majority of studies considering the role of diet in consumer-driven nutrient dynamics have assessed diet quality only in terms of %N and %P, thus stopping short of considering how specific food items and macronutrients influence the concentrations of N and P in waste products. This is problematic because it omits critical interactions at the individual level that can accumulate into impacts at the ecosystem-level. Here, we explicitly link human diet and waste management through social-ecological systems with insights from ES and GF. Importantly, this approach enables the inclusion of our understanding of how basic regulatory physiology drives foraging decisions of consumers and therefore connects individual behavior to ecosystem-scale biogeochemical processes.

ACKNOWLEDGEMENTS:

We would like to thank the organizers of Woodstoich 3, Bob Sterner and David Raubenheimer, for creating such a wonderful, collaborative working environment for this project. We would like to thank Jim Elser for creating and supporting Woodstioch for the past decade and for his devotion to all things Woodstock-related—Peace, love, and stoichiometry! We especially appreciated the support and advice of our mentor, Mike Kearney and of the senior postdoc organizers, Jim Hood and Erik Sperfeld. We would like to acknowledge the work of two anonymous reviewers who provided salient feedback in a very restricted time period. This unique opportunity for early-career scientists would not have been possible without their support. Financial support was provided by the Charles Perkins Center at the University of Sydney and the U.S. National Science Foundation (award DEB-1347502).

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Table Legends:

Table 1. Summary of proportions of dietary (excreted) N and P for six waste management approaches.

Table 2A. Summary of nitrogen flows for different diet and waste management approaches.

Table 2B. Summary of phosphorus flows for different diet and waste management combinations.

Table 2C. Summary of nitrogen to phosphorus ratios for different diet and waste management combinations.

Figure Captions:

Figure 1. Integrating global diet patterns and nitrogen (N) and phosphorus (P) consumption. A) Global patterns of macronutrient consumption (closed circles), comparing protein energy against non-protein (lipids plus carbohydrates) energy with the USA plotted over time (approximately every 10 years from 1961-2011; open circles; original data from FAOSTAT 2011, adapted from Simpson and Raubenheimer 2005). The estimated human intake target (Simpson and Raubenheimer 2005) is denoted by a target. B) N and P contents of major food groups from the USDA National Nutrient Database (USDA 2013). The KFC™ biscuit (USDA item # 21419) data was calculated using a pre-2007 recipe and the "vegetables" includes roots and tubers. C) N and P contents of different diet patterns (described in *Methods – Calculating human diet patterns)*. The Recommended Daily Intake (RDI) diet pattern is highlighted with a star. Rails (lines connecting dots with the origin) are drawn to illustrate stoichiometry, as the angle indicates the element ratios.

Figure 2. Waste fluxes under different waste-management patterns. A) US waste-management patterns. B) Developing-regions waste-management patterns. These Sankey diagrams illustrate pathways of nitrogen (N, left-hand column) and phosphorus (P, right-hand column) fluxes from human excreta undergoing transformations in the waste streams that determine the amount becoming nutrient pollution (dark red arrows), inert forms (yellow arrows), or nutrient recycling (light green arrows). Arrow width is proportional to flux magnitude (note that small fluxes are drawn with a minimum width of one pixel to remain

visible, and are not proportional). Waste-management patterns are described in *Methods – Calculating waste-management patterns.*

Figure 3. Influence of diet patterns on the total amounts and ratios of nutrient fluxes to water pollution and nutrient recycling. A) Water pollution from different diet patterns under the "hypothetical US metropolitan area" waste-management pattern. B) Nutrient recycling from different diet patterns under the "hypothetical US metropolitan area" waste-management pattern. C) Water pollution from different diet patterns under the Arba Minch waste-management pattern, which has no nutrient recycling (see *Methods – Calculating diet patterns* and *Methods – Calculating waste-management patterns* for descriptions). Axis scales are constant among all three panels. Rails (lines connecting dots with the origin) are drawn to illustrate stoichiometry, as the angle indicates the element ratios. The gray line in all panels illustrates the N:P ratio of dietary consumption in the Recommended Daily Intake (RDI) diet pattern, and the star just outside the boundary of panel (C) illustrates the total amount of N and P consumption.

Figure 4. Influence of waste-management patterns on the total amounts and ratios of nutrient fluxes to water pollution and nutrient recycling. Within each panel, different dots represent nitrogen (N) and phosphorus (P) fluxes to either water pollution (left-hand column) or nutrient recycling (right-hand column) under different waste-management patterns for the same diet pattern (see *Methods – Calculating diet patterns* and *Methods – Calculating waste-management patterns* for descriptions). We chose three diet patterns (rows) to illustrate the range of dietary N and P consumption (RDI = Recommended Daily Intake diet pattern). Rails (lines connecting dots with the origin) are drawn to illustrate stoichiometry, as the angle indicates the

element ratios. The gray line in water-pollution panels illustrates the Redfield ratio ((mass) N:P = 7.2); the gray lines in nutrient-recycling panels illustrate crop requirements for corn ((mass) N:P = 5.2) and bromegrass hay ((mass) N:P = 9.6).