ECOSYSTEM APPROACHES TO REDUCE POLLUTION IN CITIES

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SUMMARY

Further reductions in pollution will require new management approaches based on ecosystem concepts. In this paper we examine three case studies that analyze flows of nutrients in urban ecosystems to identify new approaches for pollution management. An analysis of the food chain for the Twin Cities (Minneapolis-St. Paul, Minnesota, USA) is used to examine generation of nutrient pollution in the pre-consumption, consumption, and post-consumption phases. Several scenarios of reduced protein consumption are analyzed for their impacts on agricultural fertilizer requirements. Reductions of up to 40% of fertilizer N are possible with small to moderate dietary adjustments. We then examine phosphorus inputs to urban residential landscapes to show how a newly enacted lawn P fertilizer restriction has reduced P inputs by 90%. The third case study examines fluxes of carbon, nitrogen and phosphorus fluxes through suburban households. This study shows that "high-consumption" households may have 3-4 times higher carbon and nitrogen fluxes than "low-consumption" households. Developing pollution reduction policies based on ecosystem concepts will require new research that integrates social and biophysical sciences.

KEY WORDS: Nonpoint source pollution, Urban ecosystems, Households, Lawns, Carbon, Nitrogen, Phosphorus.

INTRODUCTION

With the exception of bans on certain toxic chemicals (e.g., lead and many chlorinated hydrocarbons) modern pollution reduction practice has mostly dealt with treating pollutants at the "end of the pipe". This has worked to a large extent to reduce pollutants discharge from pipes, like city sewers and industrial discharges. However, thirty-four years since passage of the Clean Water Act and the expenditure of more than $500 billion to construct municipal wastewater treatment plants alone, we have achieved only limited success. The amount of oxygen-consuming material and bacterial contamination in rivers below cities has been reduced (EPA, 2000), but large portions of the surface and groundwater resource of the country remain polluted, particularly with nutrients (Mueller and Helsel, 2000; Nolan and Stoner, 2000).

By the early 1990s, it became apparent that nonpoint sources of pollution were responsible for much of the water pollution throughout the United States (Smith et al., 1987; Baker, 1992). At that time, cropland and pastures accounted for 43% of the N and 35% of the P entering U.S. surface waters (Carpenter et al., 1998). Nitrate contamination of groundwater underlying agricultural areas is now widespread (Neilsen and Lee, 1987; Mueller and Helsel, 2000). Urban stormwater is generally contaminated with nutrients, several metals, sediments and coliform bacteria (USEPA, 1983) and now is regulated under the National Pollution Discharge Elimination System (NPDES). Salinity has emerged as a major contaminant for reclaimed water in the southwestern U.S. (Thompson et al., 2006).

When the focus of water pollution shifted to control of nonpoint source pollutants, the "end-of-pipe" paradigm was brought along. Engineers designed stormwater detention basins, wetlands, buffer strips and other structural devices to remove pollution. Whereas structural treatment systems operate in a predictable manner when used to treat well-defined, steady flows such as municipal wastewater, operation of such systems is far less predictable when used to treat nonpoint source runoff, with variable, poorly-defined composition and flashy flows. Further, most pollutants are not truly removed (like BOD in a wastewater treatment plant); they simply are stored, eventually requiring removal. Structural controls are expensive to build; they require land that may not be available; they often are not maintained, or when maintained, maintenance costs often exceed expectations. With the notable exception of erosion reduction, there are few examples of long-term reductions in nonpoint source pollution in either agricultural or urban settings as the result of deliberate water quality management.

One reason for our failure to control nonpoint source pollution is that we have continued to rely upon the end-of-pipe paradigm. In this paper we propose that ecosystem-based approaches have great utility for reducing nonpoint source pollution and illustrate the potential with several case studies. The focus here is on ecosystem mass balance analysis, variously termed "ecosystem models", "box models" (or multicompartment box models), and "ecological flow networks" (Suh, 2005). These have been widely used for analysis of lakes, forests and other natural ecosystems. Over the past half dozen years we and others have used the concept of ecosystem network analysis to analyze urban pollution (Boyd et al., 1981; Decker et al., 2000; Baker et al., 2001a; Faerge et al., 2001), leading to a better understanding of nutrient flows through "human ecosystems" – the cities where most of us live and the farms that sustain us.

In this paper we examine nutrient flows through urban systems at three scales to examine potential pollution reduction efforts. First we examine nutrient flows through the human food chain of the Twin Cities (Minneapolis-St. Paul, Minnesota). We then examine P inputs to suburban landscapes to illustrate the potential impacts of a newly enacted lawn fertilizer P restriction. Finally, we examine nutrient movement through household ecosystems to show how consumption patterns have a major impact on fluxes of C and N.

CASE STUDIES OF NUTRIENT FLOWS

Case study 1. Flow of N an urban food chain

Our first case study is nitrogen flow through the Twin Cities food chain. Adopting the terminology of industrial ecology, we refer to the agricultural system as the "pre-use" system, eating as the "consumption" system and disposal via wastewater and landfill waste as the "post-use" system. The total population of humans and dogs is represented. We estimated human food consumption using data from the Continuing Survey of Foods (Borrud et al., 1996), mapping nutrition intake by age and sex into the Twin Cities population using U.S. Census data. Food intake for dogs was based on an equation to estimate metabolic energy from weight (IAMS Corporation) and the average composition of dog foods (Baker et al., 2001a; Baker et al., in review). We assumed 0.6 dogs per household based on an estimate of the U.S. dog population in the U.S. population (Patronek, 1995) and the average number of individuals per household in the 1990s.

In the United States, two-thirds of human protein consumption is derived from meat and one-third from crops. We assumed most dog food is derived from vegetable protein. In addition to food actually consumed, there is considerable food waste during industrial processing and usage. Here we use a value of 30% for overall food waste (Kantor et al., 1997), recognizing that this value has a high degree of uncertainty.

We then estimated N inputs and outputs needed to supply this amount of protein to the Twin Cities. Despite the complexities of the global food system that sends food to the Twin Cities, we can make reasonable estimates of major N transfers based on typical on "transfer efficiencies" from fertilizer-to-crop and from crop (feed)-to-livestock. Issues associated with estimating fertilizer-to-crop efficiencies are discussed by Cassman et al. (2002); here we use a rate of 50%, which applies to both manure and manufactured fertilizer. Feed-to-livestock transfer efficiencies vary from 15% to 50% (Ensminger, 1993; Van Horn et al., 1994; Lin. et al., 1996; Baker et al., 2001b; Stuewe, 2006). For the high efficiency systems that provide the bulk of animal products, a reasonable rate is 30%. This means that 30% of the N in livestock food becomes meat, milk or eggs; the remaining 70% becomes manure. We assumed that 30% of manure N is volatilized before the manure reaches the field (Stuewe, 2006). Loss of N via leaching and runoff associated with farming is estimated at 15% of input (Stuewe, 2006). Other transfers in the agricultural system include denitrification, volatilization of fertilizer and manure and change in soil storage (Baker et al., 2001b; Stuewe, 2006).

In the consumption phase, we assumed that excretion is approximately equal to consumption of N for both humans and dogs. Human excretion in the Twin Cities enters the wastewater system and is treated in eight municipal wastewater treatment plants. The sum of computed N excretion for the Twin Cities population is nearly identical to the sum of N inputs to the wastewater treatment plants (Baker et al., in prep.). The overall treatment efficiency for N (sum of inputs – sum of outputs/sum of inputs) was 79%. Based on Lauver and Baker (2000) we assumed that 10% of the N in a plant designed for denitrification become incorporated into sludge and the remainder of the N loss was denitrification.

Figure 1 shows that 56 Gg N/yr of fertilizer is needed by the hypothetical agricultural system to support food production for the Twin Cities. Only 16 Gg N/yr is exported from the agricultural system to the urban system in the form of animal and vegetable protein. About 8 Gg/yr is lost via leaching and runoff, and the rest is lot via volatilization and denitrification. These calculations assume no gain or loss of soil N.

At the consumption phase, dog food is 16% of the total food N supplied to the Twin Cities and human food is 84% of the total. These totals include food waste. Food N is the second largest source of N to the Twin Cities, with the largest being abiotic fixation of N_2 by combustion (Baker et al. 2005). For typical suburban households in the Twin Cities, human and dog food comprise ~40% of total N input.

In the post-consumption phase, about one-fourth of the food N enters landfills and three-fourths enters the sewage system. Eighty percent of wastewater N is "removed", becoming sludge (9%) or N_2 (91%).

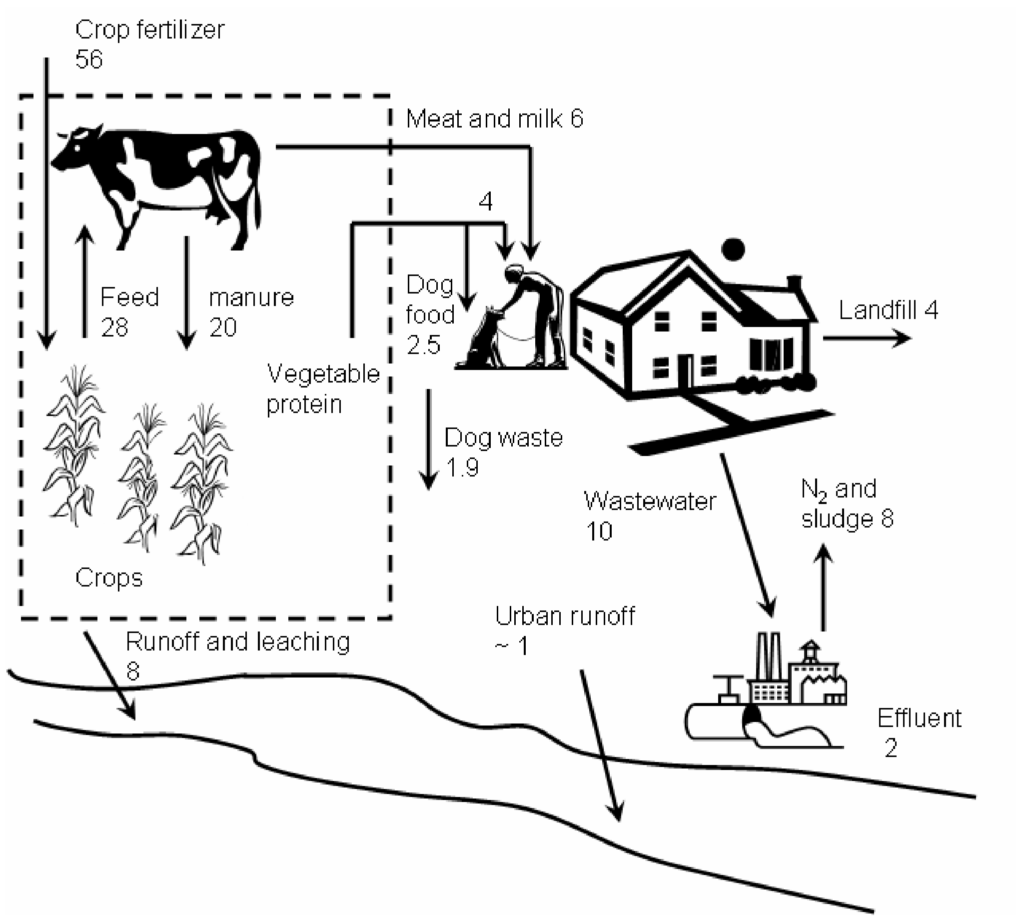

Figure 1. Flows of N from agriculture (pre-consumption), through urban environment (consumption phase) and to ultimate disposal (post-consumption phase).

Nitrogen flows through the urban food chain illustrate how modifications of urban consumption might alter the amount of fertilizer needed to supply this food. As a whole population, residents in the Twin Cities consume about 30% more protein than is required, based on the “recommended daily allowance” (RDA). This calculation is based on age- and sex-stratified RDA values extrapolated to the Twin Cities population, in comparison to actual nutrient intakes obtained from the Continuing Survey of Foods, also age- and sex stratified. Current consumption requires 56 Gg of fertilizer N (Figure 1). Reduced consumption scenarios (Figure 2) show that reduced consumption of food by the urban population would have a significant impact of fertilizer N requirements. Reducing human protein intake by 30%, maintaining a 2:1 ratio of meat:vegetable protein, would reduce the fertilizer requirement by 27%. If the meat:vegetable protein ratio also were shifted to 1:1, the fertilizer requirement would be 35% lower than the current requirement. Reducing the dog food input by 50% (by reducing the population of dogs, or the size of dogs or some combination of the two) alone would reduce the fertilizer N requirement by 5%. All of these steps (30% reduction in human protein; 50:50 meat:vegetable protein; 50% reduction in dog protein) would reduce the fertilizer N requirement by 39%.

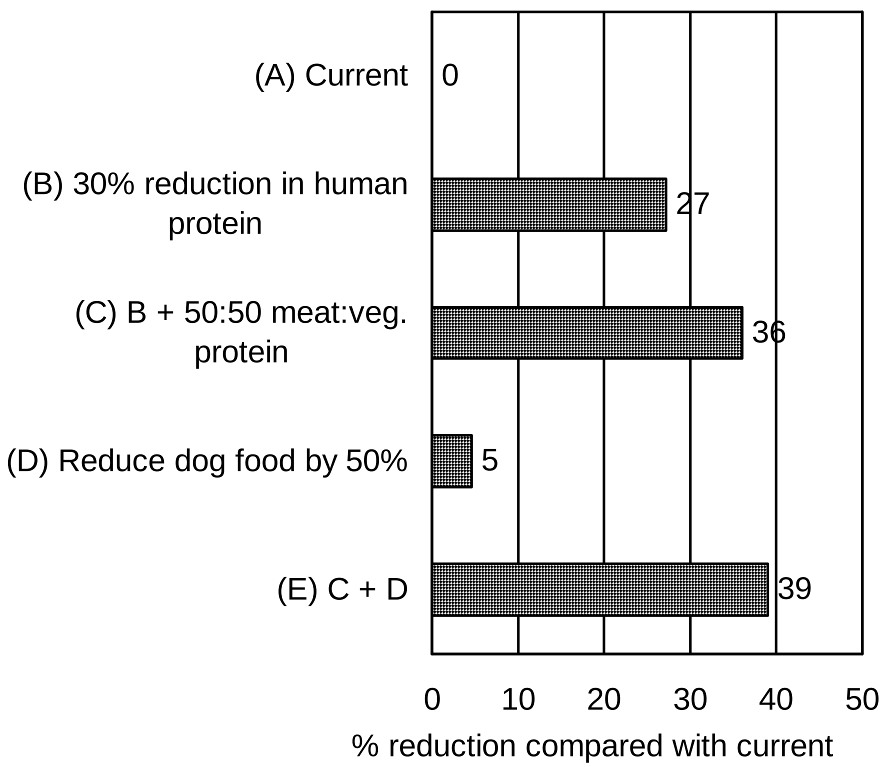

Figure 2. Reductions of fertilizer N requirements to produce food for an urban population.

In the post-consumption phase, about 5 Gg N/yr becomes sewage sludge or landfill waste. In the Twin Cities, sewage sludge is burned and the ash is landfilled. In the future, sludge may become regarded as a valuable resource, as fertilizer prices increase due to energy costs (for N fertilizer) or exhaustion of mineral sources (for P fertilizer).

The use of a food-chain analysis to identify opportunities for pollution reduction is a dramatic departure from the current pollution reduction paradigm, which relies almost exclusively on end-of-pipe treatment, either on the farm (through agricultural BMPS, which have rarely been applied successfully over large areas) or through sewage treatment. Reductions in N fertilizer requirement that could be accomplished with decreased consumption – to levels well within current health guidelines – are greater than could be accomplished with conventional agricultural BMPs. Such a dietary shift might also be healthful.

Case study II: Phosphorus in urban stormwater

In the United States, urban stormwater in cities with populations > 10,000 (and some smaller ones) is now regulated under the National Pollution Discharge Elimination System (NPDES). This has compelled great interest in reducing pollution in urban stormwater. The main response has been a flurry of construction of end-of-pipe BMPs, such as wet and dry detention ponds, wetlands, infiltration basins, rain gardens and a plethora of proprietary devices. Education, public participation, and pollution prevention are mandated components of stormwater management programs, but there is an overwhelming reliance on constructed BMPs.

Here again we can use ecosystem mass balances to develop novel approaches for pollution reduction. For urban stormwater, we can establish several boundaries to establish frameworks for analysis. The boundary for the Level 1 analysis is the watershed itself. Excluding movement of P into the human food system, which is generally separated from the landscape system, common inputs include fertilizer P, dog food, polyphosphate used in municipal water systems for corrosion control, road sand and atmospheric deposition (Figure 3).

The boundary for Level 2 analysis is the pervious landscape. The outer boundary is the watershed and the inner boundary is the edge of the street. Most P inputs to the watershed enter the pervious landscape, either directly (like fertilizer P) or indirectly (like P in municipal water, which enters landscapes through irrigation). Main P exports include runoff (soluble and particulate forms), tree leaves and grass clippings.

The boundary for Level 3 is impervious surfaces in the watershed. Most inputs come from pervious landscapes, with the exception of atmospheric deposition and road sand, which are probably minor sources. Outputs are stormwater and street sweepings. The upper boundary for Level 4 is the stormwater grate; the lower boundary is the discharge to a stream. Level 4 is the level used for designing structural stormwater BMPs.

We have analyzed the effect of Minnesota's lawn fertilizer P restriction on total P inputs to a hypothetical 5 km^2 residential watershed. This is a Level 1 analysis. The impervious surface was 20% of total area, and we assumed that 80% of pervious surfaces were fertilized at 1 lb P_2O_5 /1000 ft^2, following the University of Minnesota guidelines for medium fertility lawns. We assumed 0.6 dogs per household, with an average dog weighing 20 kg. Polyphosphates are commonly added to drinking water in Minnesota, most often at levels ~ 1 mg PO_4 L^{-1}

(Rezania, 2005). We assumed 0.5 m irrigation per year. An atmospheric deposition rate of 0.25 kg ha^{-1} yr^{-1} was used (Barr, 2004).

The Level 1 calculation shows that lawn fertilizer constituted 90% of total P input to the watershed prior to the fertilizer regulation, and the total P input was 5,702 kg yr^{-1}. Assuming 100% compliance with the regulation, total P input would have been reduced by 90%, to 569 kg yr^{-1}. In the post-regulation watershed, the main input of P to lawns was phosphate from irrigation water, which accounted for 74% of total input. This analysis shows that lawn fertilizer was overwhelmingly the major source of P to our hypothetical residential watershed.

Table 1. Effect of Minnesota's lawn fertilizer P restriction.

	Before fertilizer P restriction			After fertilizer P restriction (all other inputs unchanged)	
Source	Input rate	total, kg/yr	%	total, kg/yr	%
Fertilizer P	23 kg ha^{-1} yr^{-1} (1 lb P_2O_5/1000 ft^2)	5,133	90	0	0
Dogs	0.9 kg dog^{-1} yr^{-1}	21	0	21	4
Irrigation	1 mg PO_4 L^{-1} phosphate; 0.5 m/yr	422	7	422	74
Deposition	0.25 kg ha^{-1} yr^{-1}	125	2	125	22
Total	-	5,702	100	569	100

The impact of the P fertilizer restriction will not be an immediate 90% decline in stormwater P. Immediate losses of lawn fertilizer P to runoff range from 1% to 20%, depending largely on timing relative to irrigation or precipitation events (Baker 2006, in prep.). Following cessation of fertilizer P inputs, grass and trees will "mine" stored soil P. Most lawn soils in Minnesota are enriched with adsorbed P (as measured by "Bray" or "Olsen" P) (Barten and Jahnke, 1997). Soluble P is released when grass is mowed and decomposes; this soluble P moves into streets during precipitation or irrigation events. Grass clippings and tree leaves also are deposited directly to streets, where they decompose and release P. Over the period of many years, available soil P will become depleted, reducing the P content in plants, which in turn will reduce the quantities of P released during decomposition. At some point, soil P could become depleted to the point that plant growth is inhibited. The rate of depletion of soil P depends largely on the magnitude of other P inputs (e.g., dog feces) and the rate at which P is exported via grass clippings and tree leaves. We predict that high maintenance lawns, in which grass clippings and tree leaves are bagged and removed from the property (thereby exporting P), will exhibit faster decline of runoff P concentrations than low maintenance lawns, where tree leaves and grass clippings are recycled, thereby returning P to the soil pool.

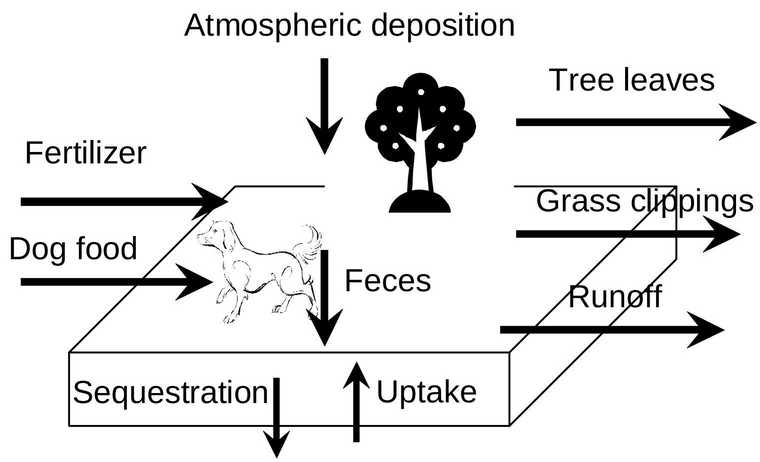

Figure 3. Schematic of nutrient flows through an residential lawn.

Case study III: nutrient balances for households.

The third scale of study for urban systems that may be useful is the individual household. This scale of analysis is important for several reasons: (1) households account for a major fraction of the C, N and P fluxes for cities; (2) understanding these fluxes could lead to novel methods for reducing pollution; (3) focusing on households enables us to elucidate the ultimate, rather than proximate causes of human biogeochemical perturbations; (4) the family household is a socially meaningful and practical unit of measurement; and (5) a model of household fluxes could become a valuable pedagogical tool to enable citizens to understand the impacts of their activities on their surrounding environment (Baker et al., in review).

We recently developed a “household flux calculator” (HFC) to compute fluxes of C, N and P through households and have used the HFC to develop scenarios for “low consumption”, “typical consumption” and “high consumption” households. All three scenarios were intended to represent four-person (two adults and two teenagers) living in a single-unit house located at an average commuting distance from work (8.2 miles in the Twin Cities). The “typical” household was developed using data from various government databases (OEA, 2000; EIA, 2001; USDA, 2001; Census, 2004), plus information from a pilot survey of 35 homes in the St. Paul suburb of Falcon Heights. High and low consumption scenarios were developed to be reasonable circumstances, not wild extremes. For example, the low and high consumption household energy scenarios were based on the 10th and 90th percentiles for household energy use in the Midwest climate region (EIA, 2001). Our high consumption household had two SUVs and one mid-size car and drove a total of 39,100 miles, whereas our low consumption household had two high mpg vehicles and drove a total of only 15,100 miles (see Baker et al, in review for other details).

The order of major C fluxes were transportation > household energy > air travel >> food. For N, the order of the top four fluxes was human food > transportation (NOx emissions) > fertilizer > dog food. The modeling analysis revealed that household consumption patterns had a very large effect on C and N fluxes (Figure 4). Modeled C flux was 3.5 times higher for the high consumption household than the low consumption household. For N and P the ratios were 2.7 and 1.2. The ratio of P fluxes was small because we assumed that no fertilizer P was used in any of the households.

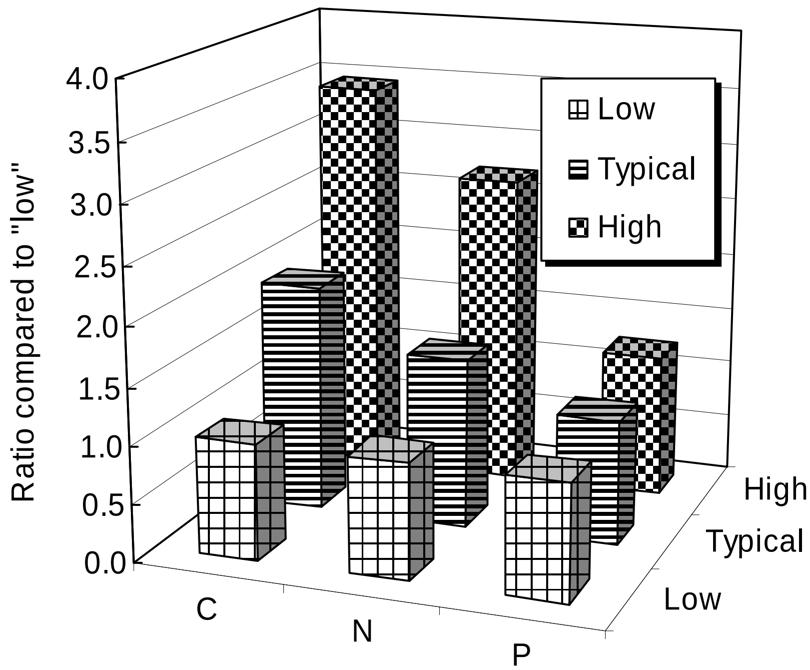

Figure 4. Ratio of C, N and P fluxes through low, typical and high consumption scenario households, compared with the low consumption scenario.

For our high consumption household, we estimated that reasonable modifications of behavior could result in a 20% reduction within one year (mainly through reduced driving and air travel), a 40% reduction in 3-5 years (replacing SUVs with more efficient cars, etc.), and a 70% reduction in 10 years (moving to a smaller home, etc.). We are currently analyzing C, N and P fluxes through each of the 35 homes we surveyed in Falcon Heights. Preliminary analysis suggests that the range of C and N fluxes is ~ 3-5-fold. These scenario calculations and preliminary analysis of data from real homes both suggest that changes in consumer behavior could have a profound effect on element fluxes.

POLICY IMPLICATIONS AND RESEARCH NEEDS

These three case studies of ecosystem nutrient flows point to new directions in pollution management, with an emphasis on reducing the production of pollution rather than treating pollution at the end of the pipe. There are several reasons for moving in this direction. First, the end-of-pipe treatment approach that works well for point source pollution does not work as well for non-point source pollution. Source reduction also offers significant economic benefits through avoided waste (e.g., less cost for fertilizer; gasoline) and by lowering the cost of remaining pollution. Finally, source reduction is fairer, moving responsibility from the community-at-large to the potential polluter.

Developing pollution management policies based on ecosystem approaches raises many new research questions. Some of these include:

(1) Disproportionality. There is good evidence that consumption and pollutant generation/export is highly skewed, with a small fraction of the population (individuals, household or farms) producing a disproportionate fraction of the consumption or pollution. Examples include household water consumption (Mayer et al., 1999), tailpipe exhaust (Calvert et al., 1993), household energy use (unpublished analysis of data from the Energy Information Agency) and farmer fertilizer use (Birr, 2005). Nowak et al. (2006) argued that edaphic characteristics of landscapes accentuate this skewness for pollution production for agricultural systems and that the intersection of farmers' behaviors and landscape characteristics accentuates disproportionality. We hypothesize that this "second order" disproportionality exists for lawn runoff also. If so, targeting source reduction involves identifying both the behaviors (e.g., excessive fertilization) and landscape features (e.g., high slopes) that lead to pollution hotspots. Policies based on careful targeting could be highly effective and possibly more widely accepted.

(3) Linkage between agricultural and urban systems. Tracking nutrients through the complex food system will require new approaches that meld techniques from industrial ecology (IE) and ecosystem ecology. IE techniques may make it possible to identify clear linkage between consumers in a given city and the specific geographic source of their food. For example, it should be possible to identify how much nutrient pollution in the Minnesota River, which drains a large agricultural watershed and empties into the Mississippi River, is caused by consumption within the Twin Cities. We hypothesize that this linkage would establish a sense of "ownership" of the specific agricultural systems which provide food to a city, encouraging reductions in consumption.

(4) Feedback loops. The potential for creating feedback loops for environmental management is growing rapidly. New technologies for ground-based and remote sensors are expanding rapidly, paralleling exponential growth in our ability to acquire, store, and process data generated by these sensors. There is an unprecedented opportunity for harnessing this new technology to create information feedback loops and develop adaptive management strategies to encourage source reduction.

(5) Adaptive management. As the concept of human ecosystems becomes established, we postulate that adaptive management concepts could play a major role in solving water quality

problems. We know of one case study of adaptive management that resulted in documented improvement of water quality (Baker et al., 2006). Research is needed to identify appropriate types of feedback, types of communication that are effective for various groups and approaches to develop informal collaborative efforts.

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