PHOSPHORUS FLOWS IN CHINA

Physical Profiles and Environmental Regulation

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Physical Profiles and Environmental Regulation

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Preface

Water eutrophication has become one of the most severe environmental problems in China, in parallel with rapid economic and societal transitions of this developing country. The lack of overall knowledge on societal cycles of nutrients, however, has contributed to the longstanding absence of effective and efficient eutrophication control in China. In order to provide a scientific basis for future improvement of eutrophication policy, this thesis has developed a comprehensive methodology by combining a substance flow analysis approach with an environmental policy evaluation framework. By applying this combined methodology for phosphorus, this thesis has systematically analyzed the physical profiles and the environmental regulations of substance flows, at both national and local (Dianchi Basin) levels. Although this study has not revealed the entire secret of curbing eutrophication via ecologizing phosphorus flows, it does prove its value, in both methodological and practical terms, for the scientific field of societal metabolism and in particular for the Chinese ongoing practice of establishing a circular economy.

This doctoral research had been implemented in the framework of the ENRICH (Environmental Management in China) project, a collaborative education and research program between Wageningen University in the Netherlands and Tsinghua University in China. The project was sponsored by SAIL foundation (now part of NUFFIC), to which I am very grateful.

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Abbreviations

AQSIQ	State General Administration of the People's Republic of China for Quality Supervision and Inspection and Quarantine
CAAMS	Chinese Academy of Agricultural Mechanization Sciences
CAAS	The Chinese Academy of Agricultural Sciences, P.R. China
CAPM	The Chinese Academy of Preventive Medicine, P.R. China
CEDT	Center of Environmental Design and Technology, Ministry of Chemical Industry, P.R. China
CEEP	Centre européen d'Etudes des Polyphosphates (West European Phosphate Industry's Joint Research Association)
CEFIC	European Chemical Industry Council
CFBS	China's Food Balance Sheet, FAO
CG	China Green International Consulting Company, P. R. China
CGHF	Compilation Group of the "Handbook of Fertilizers", China Agricultural University
CNCIC	China National Chemical Information Center
CNEMC	China National Environmental Monitoring Center
CNLIA	China National Light Industry Associations
CSYB	China's Statistical Yearbook, NBS
DE	Domestic Extraction
DESE	Department of Environmental Science and Engineering, Tsinghua University, Beijing
DHF	Domestic Hidden Flow
DMI	Domestic Material Input
DPO	Domestic Processed Output
DRSD	Department of Rural and Social Development, MOST
EEA	European Environment Agency
EC	European Communities
EPBs	Environmental Protection Bureau(s)
ERI	Ecological Restructuring Indicators
ERM	Environmental Resources Management, United Kingdom
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
GTZ	Deutsche Gesellschaft fur Technische Zusammenarbeit
IFA	International Fertilizer Industry Association
INFH	Institute of Nutrition and Food Hygiene, CAPM
ISF	Institute of Soil and Fertilizers, CAAS
IWA	International Water Association
JSDLR	Jiangsu Department of Land and Resources
JAD	Jinning Agricultural Department
KABIC	Kunming Administration Bureau of Industry and Commerce
KAA	Kunming Agricultural Academy

KAB	Kunming Agricultural Bureau
KDAB	Kunming Dianchi Administration Bureau
KEPB	Kunming Environmental Protection Bureau
KIES	Kunming Institute of Environmental Science
KWHIDI	Kunming Water Conservancy and Hydropower Investigation and Design Institute
KSB	Kunming Statistics Bureau
LCA	Life Cycle Assessment
MFA	Material Flow Analysis
MLR	Ministry of Land and Resources, P.R. China
MLI	Ministry of Light Industry, P.R. China
MOA	Ministry of Agriculture, P.R. China
MOC	Ministry of Construction, P.R. China
MOH	Ministry of Health, P.R. China
MOST	Ministry of Science and Technology, P.R. China
NBS	National Bureau of Statistics of China
NECW	National Engineering and Technology Research Center for Urban Water
NEAS	National Environmental Accounting System
NPC	National People's Congress, China
OCFI	Office of China Feed Industry, MOA
OECD	Organization for Economic Cooperation and Development
PHOSFAD	Regional Static Phosphorus Flow Analysis Model of the Dianchi Watershed
PHOSFLOW	National Static Phosphorus Flow Analysis Model of China
PIOT	Physical Input-Output Table
SAIQ	State Administration for Entry-Exit Inspection and Quarantine
SBQTS	State Bureau of Quality and Technical Supervision, P.R. China
SCKPC	Standing Committee of Kunming People's Congress
SCLIYB	Society of China Light Industry Yearbook
SCNPC	Standing Committee of National People's Congress
SCOPE	Scientific Committee on Phosphates in Europe, Belgium
SCYPC	Standing Committee of Yunnan People's Congress
SEPA	State Environmental Protection Administration
SFA	Substance Flow Analysis
SFC	Southwest Forestry College
STPP	Sodium Tripolyphosphate
STSB	State Technical Supervision Bureau
TMR	Total Material Requirement
TPC	Total Phosphorus Concentration
TVEs	Town and Village Enterprises
UNEP	United Nations Environment Programme
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USGS	United States Geological Survey
WB	World Bank
WBCSD	World Business Council for Sustainable Development
WWTPs	Wastewater Treatment Plants
YIES	Yunnan Institute of Environmental Science

CHAPTER 1

Introduction

1.1 General Introduction

The root of the deep conflicts between social development and the ecological environment derives from anthropic interventions in material flows. It has been argued that human-induced material flows in modern economies are much more intensive than those in natural cycles. Moreover, the paths and transformations of the societal processing of materials are far more complicated. Since rapid economic development is not yet in harmony with sustainable natural ecosystems, the underlying conflict between them has become the most important factor influencing an increase in social welfare. To reconcile economy and ecology it is essential to systematically reflect on the societal mechanism of material flows, thus ecologizing the economy by shifting production and consumption paradigms to improve the overall efficiency of material throughput.

Phosphorus (P) is one of the most common elements on earth and is essential to all living organisms. As a non-renewable mineral, P reserves exist only at the earth's crust in the form of phosphate rock, which is currently being mined. Historically, as one of the main plant nutrients, P was recycled in agricultural communities: food was consumed close to its place of production and the resulting animal and human manures were applied to the same land. The growth of cities and the intensification of agriculture have depleted soil nutrients, which has increased the need for synthetic P fertilizers. This has broken open the originally closed loop of P nutrient and has caused an increase in P in human societies. As a consequence, the natural balance of production and consumption in the food chain is disturbed, usually leading to algae becoming the dominant form of life in water. This in turn shapes one of today's challenges, i.e. responding to the increasing demand for food and agriculture's growing need of nutrients without exhausting P mineral resources and threatening surface water.

A structural clash between resources and the environment has been ongoing in China since the late 1970s because of the economy-oriented development strategy, in which ecological rationality has to a large extent been ignored. It is clear that continued economic growth and an expanding population together with accelerated processes of industrialization and urbanization – which is a likely scenario for the near future – could significantly influence or even completely destroy the underlying ecological basis if China persists in its current development mode. This in turn could substantially undermine the overall capacity of China's long-term sustainable development. In this sense, the ecological restructuring of the economy will become the greatest challenge confronting China, the world's largest developing economy, in a crucial period of social and economic transitions.

Water eutrophication is one of the most severe environmental problems in China. The pressures created by an expanding population, urbanization, industrialization and agricultural intensification over the last two decades have resulted in a massive increase in P from various sources. More importantly, the long- term pursuit of economic profits has weakened the domestic productivities of P throughput, ranging from mining, manufacturing, and farming to breeding. Combined with the prevalent adaptation of conventional end-of-pipe control technologies, the P flows within China's economy could differ significantly from the natural paradigm. It has been argued that, compared to nitrogen, P will become the key limiting nutrient and will dominate the rate of algae growth and thus the state of eutrophication in China.

1.2 Phosphorus Flows in Natural and Societal Contexts

1.2.1 The biogeochemical cycle of P substances

P is the eleventh most common element on Earth, accounting for 0.1% of the elemental abundance in the lithosphere (Jiang 1999). P is indispensable for all living organisms, playing an essential role in the processes of photosynthesis and metabolism, in particular the transferring and transforming of biological energy (Karl 2000). In nature, P always occurs combined with oxygen and other elements, forming phosphates. The formation of PO_4^{-3} consists of over 300 kinds of P compounds in nature, and the elementary P occurs only under controlled conditions in laboratories (Han, Li and Huang 1999).

At a global level the biogeochemical P cycle can be represented by seven sinks with P linkages between them, as shown in Figure 1.1. The sink of the deposit terrane, the largest in the natural cycle, includes the stable P in the lithosphere 600 mm below the surface of the soil and unexploitable P minerals. In contrast, the second sink, i.e. the terrestrial sphere, refers to the P at the surface of the lithosphere, which is closely related to plant growth, stream transportation, atmospheric sedimentation, etc. The sink of terrestrial organisms consists of various kinds of P reserves in the biological mass, in which forestry and aquatic ecosystems are the major and minor contributors, respectively. The marine organisms exist mainly at euphotic layers and are responsible for a small part of P contained in the entire ocean. A seawater layer at 3 m under the ocean's surface constitutes the epipelagic sink, where the most active exchange of P between marine organisms and seawater takes place via biological decomposition. The sea under the epipelagic layer, the bathypelagic sink, is the largest pool of dissolved P. The final sink lies in the atmosphere, which connects to the terrestrial and epipelagic zones via volatilization and sedimentation.

The natural P flows, therefore, can be defined as biogeochemical processes involving both mechanical transference and physical, chemical and biological transformations of P among these sinks. As illustrated in Figure 1.1, the dominant P flows in the natural cycle begin with wind and water erosion (efflorescence) of phosphorite rock in small quantities, then continues through terrestrial (organisms and soils) and freshwater (shallow and deep) ecosystems, and terminates in the oceans, while the atmosphere has a relatively slight impact on the magnitude of the flows. A closed cycle of P is attributed mainly to ocean currents, which bring P up to continental boundaries and upload P at epicontinental zones (Han, Li and Huang 1999). This subsequently leads to a booming expansion of marine organisms along the coastlines. While the enormous biomass residue deposits on the continental shelf are subject to lithification, involving complicated physical and chemical transformations, mineral P sedimentation shapes up ultimately (Zhang 1997; Clark, Ingall and Benner 1998; Zhang, Liang and Geng 2001). In addition, marine P at the epipelagic layer can also return to the continents in the form of droppings from seabirds.

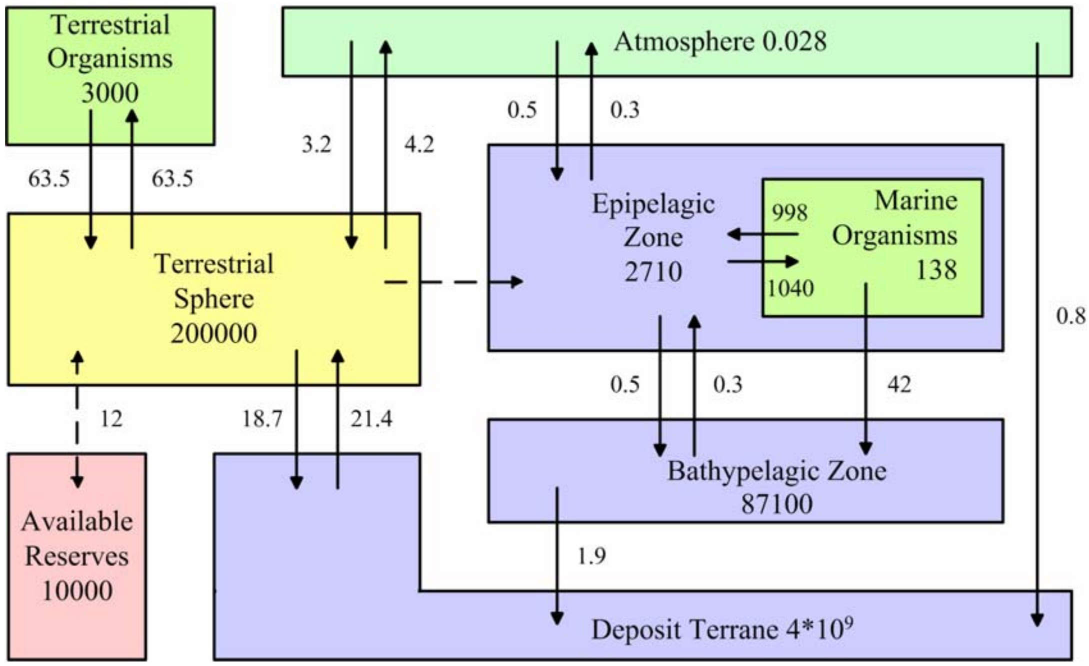

Figure 1.1 The biogeochemical P flows in nature (unit: Mt P)

Sources: based on Han, Li and Huang (1999).

Since P has to experience such a time-consuming series of geological transformations, one single loop of the natural cycle can take place over one million years (Dongye 1996; Tiwari and Rao 1999). More importantly, the natural P flows remain relatively constant in quantity and contribute to a stable closed loop. However, increasingly intensified human activities have significantly disturbed the natural P cycle, particularly the exploitable P reserves, cultivated areas, biomass communities and freshwaters in the terrestrial sphere, problems we will now turn to.

1.2.2 Societal metabolism of P bulk-materials: A historical view

Each step of the productivity of human society has remarkably affected the natural P cycle. Originally, human beings, characterized by small-scale habitation and the nomadic life style (hunting and plucking), were in harmony with nature. The emergence and development of farming made it possible for human beings to survive on relatively small tracts of land by artificially cultivating various plants and animals. This also shaped a close-looped P cycle within the agricultural society: the byproducts and residues of crops were used as feed, and human and animal manures were recycled into farmlands to maintain soil productivity. As the population grew, however, it became increasingly difficult to support both humans and livestock with these insufficient organic fertilizers. People began to seek new methods to improve agricultural production. As early as 1831, liquid P fertilizer was made up by dissolving bones and coprolites with dilute sulfuric acid and was then applied to farmlands in the Bohemia region of Europe. Subsequently, wild guanos teeming along the Peruvian coasts, were transported across the world. Around 1850, a P deposite was discovered in South Carolina in the United States and has been industrially mined since 1867. Large- scale mines were continuously found one after the other in Europe and North America. As a consequence, P ores accounted for 90% of the global consumption of P in 1889. Once the modern phosphate fertilizer industry was founded with the establishment of the first ammonium phosphate factory in US in 1917 (UNEP 1998), farmlands could be supplemented by an increasingly intensive application of synthetical P in a short period to provide a higher yield of crops. However, this dramatically accelerated the slow biogeochemical rate of the P flow from the lithosphere into the soil.

Intensified farming, made possible by the mechanization processes that took place between 1920 and 1950, has been detrimental to the environment (Matson et al. 1997). First, small pieces of cultivated areas were integrated by removing windbreak forests in order to make it easer for agro-machines to work the land. As a result, P loss from soil by wind erosion was enhanced. Second, geometrical farming on large, even tracts of land instead of traditional ploughing along the contour lines of slopes accelerated P leakage by water erosion and agricultural runoff. Moreover, heavy machines put pressure on the farmland and weakened the soil's capacity to conserve water, thus facilitating serious erosions. Third, by substituting agro-machines for farm animals, intensified farming lost one major source of manure. Accordingly, modern agricultural production has destroyed the previous ecologically closed circle by applying chemical P fertilizers to compensate for its decrease in farmlands; meanwhile, modern agriculture is responsible for the losses of P through various paths, resulting in an increasing dependency on chemical P fertilizers.

Stimulated by the development of the feed industry, animals, liberated from industrialized farming, are now bred centrally (Erkman 1999). As a result, the concentrated generation of P in the form of animal manure now commonly exceeds the capability of farmlands where the animals are raised (Brown 2002). Therefore, a large amount of P cannot return to the farmlands where it originated.

The construction of concentrated residences has resulted in a nutrient metabolic rift (Foster 1999). Because more and more of the population lives in cities, they have a looser nutrient relationship with farmlands since the distance between them has significantly stagnated the agricultural reuse of urban human excreta (Driver 1998). On the other hand, the excess of human waste in cities resulted in serious sanitation crises, as evidenced by the cholera epidemics of 1831–1832 and 1848–1849 in Europe. Urban sewage systems, combining household sanitation, sewer networks and pollution purifiers at the end of the pipe were gradually developed to collect, transfer and eliminate human waste (Stauffer 1998). Again, because this vast urban infrastructure has been adopted worldwide, P recovery and recycling has decreased (Steen 1998).

1.2.3 P use and induced resource and environment issues

Today, the mineral P is intensively exploited for multipurpose economic applications. Agriculture is by far the largest user of P, accounting for 80% to 85% of the total consumption worldwide (Jiang 1999). Second, food-grade and feed-grade phosphates are widely used as additives. Third, P, in form of sodium tripolyphosphate (STPP), is added to detergents to increase hygiene and cleaning performances compared to soap-based products. Fourth, P is used in such diverse applications as metal surface treatment, corrosion inhibition, flame retardant, water treatment and ceramic production (Chen and Tan 1989). Despite such widespread use, these applications represent only about 3% of the total consumption (CEEP 1997).

The societal metabolism of P, triggered and consolidated by agricultural and industrial revolutions as well as by population growth and urbanization, significantly differs from its counterpart in nature and thoroughly reshapes the P cycle as a whole, as shown in Table 1.1.

Table 1.1 Estimation of anthropic impacts on the global P cycle (unit: Mt P/year)

P flows	Natural	Pre-industrial (1800)	Recent (2000)
Natural fluxes intensified by human activities			
Erosion	> 10	> 15	> 30
Wind	< 2	< 3	> 3
Water	> 8	> 12	> 27
River transport	> 7	> 9	> 22
Particulate P	> 6	> 8	> 20
Dissolved P	> 1	< 2	> 2
Biomass combustion	< 0.1	< 0.2	< 0.3
Anthropogenic fluxes			
Crop uptake	—	1	12
Livestock wastes	—	> 1	> 15
Human wastes	—	0.5	3
Organic recycling	—	< 0.5	> 6
Inorganic fertilizers	—	—	15

Sources: Smil (2000).

The disturbed P flows in modern economies have resulted in serious resource and environmental issues. The increasing demand for P ores, dominated by agricultural production, has threatened the mineral reserve. It was argued that global consumption of synthetical P fertilizers rose from 1.73 million tons (P_2O_5) in 1920 to 328.8 million tons (P_2O_5) in 1998, approximately a 19-fold (IFA 1999). Moreover, it was projected that the trend is unlikely to decline in next 30 years, but will instead probably increase from 0.7% to 1.3% annually (FAO 2000). This strongly suggests that P, as a finite non-renewable resource, can be completely exhausted at the accelerated consumption rate. It has been shown that the global average P content in raw ores dropped to 29.5% in 1996 from 32.7% in 1980, and the global reserve can sustain the current mining intensity for only another eighty years (Isherwood 2000).

The industrialized extraction of P itself can also result in ecological deterioration. A huge amount of rock and soil that are moved to reveal raw ores and waste mineral generated during mineral separation and concentration, are likely to harm the environment if they cannot be disposed of adequately. Moreover, the successful restoration of aquifers, vegetation, habitats, landscape and surface water is difficult and time consuming. During the subsequent process of mineral manufacturing, a great deal of solid waste is generated in the form of P-gypsum, which is one of the largest industrial wastes. Because of its high metal concentration, humidity and techno-economic barriers, however, P-gypsum is usually subject to ocean dumping and land-filling (Rutherford, Dudas and Samek 1994). These solutions, instead of recycling, have contributed to increased P consumption and ecological risk.

Some of the leakage of P from various sources, such as chemical industries, farm lands, livestock feedlots and residential communities, eventually ends up in surface bodies of water. The excessive P enrichment in lakes, reservoirs, slow-flowing rivers and coastal seas can result in eutrophication, one of the most crucial P-related environmental issues (Mainstone and Parr 2002). The term “eutrophication” has its origins in the Greek word “eutrophos”, meaning well-nourished. But all too often, the term has come to be associated with adverse water quality. Since P is essential for the growth of phytoplankton (algae), the overloaded nutrient can stimulate a booming growth of algae communities. Insufficient oxygen caused by surplus algae metabolism can subsequently lead to the death of most aquatic organisms and can even destroy the whole aquatic ecosystem. The ecological deterioration of surface water, therefore, imposes great impacts on human health as well as local economic and social development.

In the 1960s many developed countries began to alleviate the P surplus in surface water by constructing municipal sewage infrastructures and implementing P discharge restrictions on production sectors (ILEC and UNEP 2003). However, eutrophication is still widely spreading. It has been argued that non-pointed sources, compared to industrial factories, have a major responsibility for the undesired spread of P nutrient (Haygarth 1997; Zeng 1999). Based on a pollutant tracking approach, for instance, a long-term project for monitoring chemical fertilizer use and loss at Cheney Basin in Kansas, the United States, showed that 65% of the total P load came from agricultural runoff, much more than from natural and industrial sources (Pope, Milligan and Mau 2002). Likewise, the study of European experiences revealed that animal husbandry, human excreta, farming practices, and use of detergents dominated the structures of P emissions, although they varied considerably among these countries, as shown in Table 1.2 (Morse, Lester and Perry 1993).

Table 1.2 P sources and inputs to surface waters in European countries in 1992

Countries	Total (kt/year)	P load percentage by various sources (%)					
		Natural background	Industrial enterprises	Agricultural runoff	Animal husbandry	Human excreta	Detergents
Austria	13	12	6	16	36	20	10
Belgium	13	5	8	7	43	26	11
Denmark	15	6	5	11	55	12	11
Finland	9	38	3	15	17	18	9
France	106	11	6	19	31	18	15
Germany	97	7	6	12	44	28	3
Greece	17	15	5	34	18	21	7
Ireland	15	9	2	24	49	9	7
Italy	56	11	8	18	26	35	2
Netherlands	24	3	5	9	57	23	3
Portugal	15	12	7	16	27	24	14
Spain	72	14	7	26	18	19	16
Sweden	14	33	7	14	15	21	10
U.K.	82	6	8	14	29	24	19
Average	548	10	7	17	32	23	11

Sources: Morse, Lester and Perry (1993).

It can be concluded that the P throughput in modern economies has significantly affected the ecosystems. The continuing intensification of farming and breeding as well as urbanization could result in an exhaustion of mineral P reserves in the near future. At the same time, most of the processed P is considered as waste that needs to be controlled rather than as a reusable resource. This part of P, entering back into natural ecosystems via various paths, contributes to a series of ecological issues. Among these, the excessive concentration of P in surface waters is definitely one of the substantial challenges to sustainable development.

1.3 Eutrophication Control in China

1.3.1 The state of eutrophication as indicated by P

China is a country with numerous lakes, 2,300 of which are larger than 1 km^2. The total area and storage capacity of natural lakes accounts for 70,988 km^2 and 708 billion m^3, respectively. There are 86,852 artificial lakes, i.e. reservoirs, with 413 billion m^3 of storage capacity. All these lakes annually contribute 638 billion m^3 of freshwater to China (Jin 1995).

As the largest developing country in rapid socio-economic transition, China has experienced severe human-induced eutrophication since the 1980s (Jin 1995; Dokulil, Chen and Cai 2000; Yang and Ren 2000; Jin 2001). Besides academic research by expert individuals (refer to Introduction sections in Chapter 3 and 4), this problem is also discussed in the important report of the World Bank, i.e. "China: Air, Land and Water" (WB 2001), and in the release of official monitoring results by the State Environmental Protection Administration (SEPA 2003). In the WB's report, lake eutrophication was addressed as one of the most crucial environmental issues. It was argued that the eutrophication of Taihu Lake, Chaohu Lake and

Dianchi Lake, the national key watersheds of environmental protection promoted at the Fourth National Environmental Protection Conference in 1996, had shown very little improvement. In contrast, the results of the total phosphate (TP) concentration of all three lakes clearly revealed an aggravated trend, as shown in Table 1.3. In terms of reservoirs, SEPA's monitoring results in the Three Gorges (the section between Jiangjin-Sichuan and Yichang-Hubei) suggested that the TP had become the second- ranked pollutant in 2002 (SEPA 2003), which could in turn result in a serious eutrophicated state of the largest artificial lake in China.

Table 1.3 Trend of average TP concentration of the "Three Lakes" in China (unit: mg P/L)

	1992	1994	1996	1998
Taihu Lake	0.08	0.13	0.11	0.07
Chaohu Lake	0.25	0.34	0.25	0.36
Dianchi Lake	0.23	0.30	0.23	0.41

Sources: WB (2001).

1.3.2 Eutrophication control approaches: a view of P flows

China implemented eutrophication control programs in the early 1990s. In the case of Dianchi Lake, a vast expenditure, accounting for 3.09 billion RMB, was spent on a number of engineering and management programs between 1990 and 1998, aiming at alleviating the increasing eutrophication. These expenses were mainly distributed into eight projects, including transferring water resources from adjacent watersheds, constructing municipal sewage systems, truncating domestic wastewater, non-pointed sources pollution control, restoring stream ecosystems, afforestation, excavating lake sediments, and controlling interior source pollution (Xu 1996; Wang 1998; Huang 1999; Zhang 2000). Meanwhile, two regulatory measures were carried out, i.e. "Implementation of the Deadline for Industrial Pollution Discharge up to the National Standards in Dianchi Basin" (*Lingdian Xingdong*) and the limitation and prohibition of the production and consumption of laundry powders containing P. However, monitoring data series suggest that these control measures have barely produced the expected improvement of water quality in Dianchi Lake (refer to Figure 4.2 and 4.3; Meng 1999; Liu 2001; Meng 2002). The same is true in the other lakes (Jin 2001).

Because of the difficulty and complexity involved in restoring freshwater aquatic ecosystems, the expected remediation of lake eutrophication cannot be achieved overnight. But, this by all means does not necessitate the simplification and fragmentation of P control strategy in China. It is apparent that the existing measures have basically relied on large- scale engineering instruments, aiming at cutting down P loads from various sources, diluting P in lake water, or removing P in sediments. Although these kinds of end-of-pipe measures are necessary and helpful, they are not sufficient or efficient enough for an adequate P control strategy. Moreover, limiting or prohibiting detergents containing P is also questionable (Shu et al. 1999; Huang et al. 2001; Qian 2000; Song et al. 2003). Regarding the continuing extension of eutrophication in most domestic lakes, it is questionable whether China has proposed an effective control strategy.

In order to answer this question, it is necessary to return to the underlying physical basis of eutrophication, i.e. the societal metabolism of P within the Chinese economy. The rapid social and economic development has led to significant changes in the physical structure of domestic P throughput. In the year 2000, China's population reached 1.27 billion, 22% of the total population in the world, an increase of 28.4% from 1980. The urbanization rate was 36.2% in 2000, almost double to that in 1980 (NBS 2001). The huge population growth and the improvement in living standards together with economic growth have significantly magnified the flux intensities of domestic P flows and reshaped the underlying physical structure of P throughput. As indicated by agricultural production, the national grain output rose to 462.2 million tons in 2000, increasing by 58.8% from 1980. Likewise, the yields of meat, milk and eggs increased by 3.2, 3.2 and 4.2 times, respectively, between 1985 and 2000 (see Figure 3.4). The intensified farming and breeding sectors in turn have stimulated the productions of P minerals and fertilizers, which reached 1.8 and 2.9 times those in 1980. Combined with fertilizer import, the application of various chemical P fertilizers accounted for 4348 thousand tons, increasing 3.5 times compared to levels in 1980 (NBS 2003).

The substantial shifts in the structure and function of P throughput bring forth a challenge for eutrophication control in China. However, current policies and approaches have to a large extent ignored the underlying physical characteristics of P flows. It is fair to say that the lack of a holistic knowledge of the societal regime of P leads to the long-term absence of an integrated policy and efficient instruments. Therefore, it is not surprising that China has placed the reliability of effectively controlling water eutrophication upon the end-of-pipe technological regimes.

Undoubtedly, there is an urgent need for China to develop innovative approaches to handle eutrophication. The desired policy framework should systematically take the underlying characteristics of P flows, interrelated with mining and industrial production, farming and breeding practices and urban and rural livelihoods, into consideration. It is believed that understanding the physical profiles of material throughput can definitely facilitate the innovation of a policy framework, by shifting from end-of-pipe abatement to overall environmental improvements in economic structures, social behaviors, technological application and institutional arrangements.

1.4 Research Questions

The core objective of this study is to analyze and understand the characteristics of P flows through the Chinese economy, to assess the contribution of the various P flows on water eutrophication and to evaluate current environmental policies in China aimed at curbing eutrophication by P.

There are two research questions:

1) What are the main structural characteristics of P flows in the Chinese economy; and where, to what extent and which P flows impose negative influences on water eutrophication?
2) What has been the contribution of current environmental policies in China to reduce these negative environmental impacts of P flows on water eutrophication?

In addition, it is necessary to explain why this study is dedicated to P flows analysis for eutrophication control. The emergence of water eutrophication is dependent on many factors, including climate, hydrology, nutrients, aquatic communities and other geographical conditions. Among these, nitrogen (N) and P are regarded as the key causal factors. This study chooses P substance as the central research target because: 1) P is considered the dominant limiting nutrient in most cases, especially in shallow lakes (Jin 1995; Dokulil, Chen and Cai 2000); 2) some natural N flows, in particular the processes of atmospheric sedimentation, biological nitrification and denitrification, account for a considerable proportion of total N flux. Moreover, compared to the natural process of denitrification, applied P cannot be naturally recycled. Regarding the central concern with the metabolism of substance, this may justify that N is less appropriate research target in comparison with P; and 3) N is unlikely to be exhausted, while P is regarded as a non-renewable mineral resource. Thus, the inquiry into societal P flows can result in a distinct co-advantage, besides the benefit for eutrophication control, for understanding the issue of P resource conservation.

1.5 Structure of Thesis

This thesis contains seven chapters. In order to seek answers to the research questions, a systematic approach, combining a static substance flow analysis approach and an environmental policy evaluation framework is developed in chapter 2, based on an analysis of various existing physical-based analytical methods and policy evaluation models. Based on several criteria for research strategy, Dianchi Basin has been selected as a regional case study.

Chapter 3 and 4 are dedicated to the formulations of two static P flows models at the national and local levels, i.e. PHOSFLOW and PHOSFAD, respectively. By tracking P flows from its natural origins, through the economies to environmental destinations, the key metabolic structures and material efficiencies of P throughput in conjunction with resource extraction, phosphate industries, agricultural production, livestock husbandry, domestic consumption and waste treatment are identified for both the national and the local economies, respectively. Based on the experiences in modeling P flows, chapter 5 contains a reflection on the applicability of the SFA approach for the current data condition in China. Applying an elaborated strategy of data processing and new collected data, the two static P flows models are discussed and modified for a better reflection on the 'real' P flows. The key structural features are subsequently compared to produce an understanding of the physical differences in P throughput between the nation and the local economy in Dianchi Basin. Chapter 5 also draws up a feasible agenda for evaluating P-related policies by discriminating the environmental impacts of P flows and thus contributes to a linkage with the next chapter.

Chapter 6, the environmental policy evaluation part, focuses on three clusters of environmental policies associated with intensive livestock regulation, urban centralized P abatement, and the ban on detergents containing P. By operationalizing the EEA's evaluation framework, the relevance, cost-effectiveness, effectiveness and social welfare of related policies are subsequently examined at both the national and local levels. This chapter, in response to the second research question, is conducted relying on statistical data and literature as well as the empirical study at Dianchi Basin, aiming to facilitate the understanding of how successful the central and local governments are in reducing the negative impacts of P production and consumption on bodies of water.

Chapter 7 closes this thesis with some substantial conclusions on both the physical characteristics and social arrangements of P flows in the national and local economies. The final chapter also includes recommendations on the ecological reconstructing of the model-based physical structure and on the desired improvement of the policy-oriented regulation of P throughput. A reflection on the methodology applied in this thesis is devoted to its contribution to a scientific field of material metabolism study and future research directions in China.

CHAPTER 2

Research on Societal Metabolism of P: Methodology

2.1 Introduction

The research questions necessitate a comprehensive methodology, which should provide a physical insight to societal P cycle and an examination of the functions of current P-related environmental regulations.

The starting point to develop such a methodology is to seek for an appropriate method to analyze the physical features of societal P metabolism.[1] Such a method should be capable to analyze and identify the directions and intensities of P flows which influence the water environment. While a number of existing economic analyses and environmental studies can hardly generate a sound insight into material flows within modern economies, a cluster of promising physical-based analytical methods has been recently developed, in particular within the school-of-thought of industrial ecology. Because the methodological characteristics of these approaches vary considerably, there is a need to screen them to identify the most appropriate approach for this study. We will start investigating the main strengths and weaknesses of various physical-based approaches which have been applied widely. By focusing on three material flow analysis approaches, finally a substance flow analysis (SFA) approach will be selected and operationalized for analyzing P flows in this study (section 2.2).

As none of these physical-based approaches, including a SFA method, can serve as a tool to assess and evaluate P-related environmental regulations, we will redirect our focus on policy evaluation methods. A policy evaluation method provides a framework to analyze normative goals on P flows in relation to their underlying socioeconomic factors, such as organizational fit, institutional capacity, technological trajectories, costs and benefits, knowledge, preferences and behaviors. Via a systematic investigation on P-related environmental policy design and implementation, a methodology for inquiring the second research question in this study can be constructed. In section 2.3, we will select a proper evaluation method, based on a discussion of the core criteria and main models of environmental policy evaluation.

[1] The word "metabolism" was originally used to describe the microcosmic life process, involving nutrient ingestion, assimilation and waste excretion. Along with the studies on material throughput within modern economies, the concept has been entailed to socio-economic metaphor. Analogous to the biological process, the materials' societal metabolism (also called socioeconomic or society's metabolism) consists of resources exploitation, processing and manufacturing, consumption, waste disposal and recycling within modern economies (Fischer-Kowalski 1994).

The research methodology employed in this study hence consists of two methods which are further integrated by a case study research strategy, as discussed in section 2.4. The methods of data collection, that operationalize this methodology, are reported in the final section.

2.2 Analyzing Societal Metabolism of P

Neither conventional capital-based monetary accounts nor economic statistics, however, provide an appropriate and sufficient elaboration of the overall material throughput within modern economies (Cropper and Oates 1992; Adriaanse et al. 1997; Bouman et al. 2000; Bartelmus and Vesper 2000). There are three reasons for this. First, although these traditional approaches examine economic activities and related environmental impacts independently, they can hardly depict the structural characteristics and evolutionary trajectories of the underlying physical basis because they are initially oriented to capital growth and the price principle. Second, while social development, economic growth and environmental consequences are connected in these traditional approaches, the results cannot systematically recognize the substantial material correlations between them and thus fail to provide profound control strategies for ecologizing the economy. Third, these traditional approaches cannot give a comprehensive investigation of the human-induced material disturbances. In particular, some material flows which have large effects on the ecology but do not enter into the economy, have been generally ignored (e.g. removed rocks and mineral tailings generated via resource exploitation, crop residues and biomass waste of lumbering). Therefore, these traditional studies have rarely fostered comprehensive insights into the physical profiles of modern economies.

As early as the mid nineteenth century, Marx proposed an interrelated ecological connection between human society and nature, by criticizing the metabolic rift of nutrient cycle caused by rapid urbanization (Foster 1999). However, nearly no analytical method has been subsequently developed based on Marx's ideas since then. Till the 1960s, some economists, involving Kneese, Ayres and Leontief, recognized the underlying importance of materials' societal metabolism within modern economies. They strongly suggested that there was a need to reflect on material flow systems as soon as possible. Meanwhile, they presented preliminary approaches for mass balance analysis based on economic theory and input-output methods, respectively, in order to explain the issue of externality of production and consumption (Ayres and Kneese 1969; Leontief 1970). These studies have greatly promoted the application of material-based approaches for identifying the environmental impacts caused by industrial structure. The concept of Industrial Metabolism was firstly introduced by Ayres in 1988 (Anderberg 1998).[2] He subsequently argued that the industrial metabolism could refer to the aggregation of overall physical processes by which raw materials are transformed into products and wastes via labor forces in modern economies (Ayres 1994). With his work the research paradigm of material metabolism was formally built up. Afterwards, the

[2] There are few distinctions between the concepts of "industrial metabolism" and "material metabolism", as the former emphasizes the underlying structure and operation of industrial economies. In addition, these concepts have been widely adopted by different academic schools, i.e. economists and sociologists, respectively, while both of them are used in environmental sciences and ecology.

establishment and development of Industrial Ecology Theory (IET) in the 1990s have provided substantial theoretical support for developing approaches and applying outputs of the study on materials. This has further promoted the development of modern material-based approaches and thus facilitated profound cognizance of the essential principle of economic development and its induced fatal impacts on natural ecosystems.

Over ten material-based approaches have been employed in various kinds of societal metabolism studies, ranging from transformations or expansion of environment- economic analyses, through simple mass bookkeeping, to complex material flow modeling. The next section will briefly examine some material-based methods which have been applied widely, and then focus on material flow analysis approaches.

2.2.1 Physical-based analytical methods

The methods for inquiring societal metabolism of materials are physical-based approaches[3], aiming at systematically analyzing the distribution and intensity of material flows which have significant influences on natural ecosystems within certain spatial and temporal boundaries. These physical-based approaches emerged since the 1990s and have been directed towards a methodologically strict and quantitative elaboration of the relationship between material throughput and environmental consequences.

The main objective of societal metabolism studies is to provide a new epistemic basis for reinventing ecological modernization policies, by analyzing the overall lifespan of material throughput within modern economies and illuminating implications for sustainable economic growth and environmental improvement. It is fair to say that these physical-based approaches do not aim to deny or substitute traditional analytical methods based on capital flows, but serve as methodological complements for exploring the relationship between the ecology and the economy. In fact, to examine economic systems from a perspective of societal metabolism has become one of the innovative directions of traditional economic theories and methods. Input-output analysis and life cycle assessment approaches, for instance, have been modified to invite the physical processes of material flows in conjunction with industrial production and services (Duchin 1992; Konijn and Dalen 1997; Nijkamp and van den Bergh 1997; Hofstetter 1998; Joshi 2000; Lenzen 2001; Matthews and Small 2001; Sliva 2001; Hinterberger, Giljum and Hammer 2003). Consequently, these two analytical methods play an important role in the research field of materials' societal metabolism.

The basic concepts, data processing and analytical approaches vary considerably in the domain of material metabolism study, because of the complexity and diversity of material throughput, data insufficiency, and the differences in national statistics and academic background of researchers working in this field. Compared to the conventional

[3] The research realm of energy metabolism has been independently developed in parallel with the studies on material flows, because social production and consumption of various kinds of energy have shown distinct features in both economic and environmental contexts, although materials are related to energy flows. For sophisticated discussions on energy flows see Cook (1971); Rappaport (1971); Haberl (2001; 2002); EEA (2002).

environmental-economic methods, however, all these physical-based approaches are characterized by four underlying traits as described below:

1. The first law of thermodynamics, i.e. the principle of Mass Conservation, serves as the fundamental rationality for applying the analytical technique of mass balance (Kleijn 2000);
2. The physical indicators, involving the quantities of weight, volume and area, instead of currency or price, are employed as the basic units (Mathews et al. 2000);
3. The analytical framework aims to construct the causal relationship between economic activities and natural ecosystems, by analyzing the transfer and transformation of crucial material flows within and across the economy (Udo de Haes, van der Voet and Kleijn 1997);
4. The alternative solutions rely on identification and assessment of the ecological rationalities and environmental impacts of the direction, turnover and intensity of material flows (Daniels and Moore 2001).

A number of physical-based methods have been widely applied (cf. Table 2.1). They mainly include: 1) Material Flow Analysis (MFA) approach, a cluster of various kinds of analytical methods, concentrates on important elements, bulk materials, products or total material throughput (see detailed discussion in the next section); 2) Physical Input-Output Table (PIOT), based on standard or expanded Leontief matrix, describes the material relationship among production and consumption sectors at different phases of material processing (Leontief 1970; Silva 2001; Hubacek and Giljum 2003); 3) Life Cycle Assessment (LCA) technique is basically operationalized to evaluate material use and environmental impacts associated with the entire lifespan, i.e. "from cradle to grave", of certain specific products or techniques (Krozer and Vis 1998; Graedel 2000; Heijungs and Kleijn 2000; Udo de Haes 2000). 4) Ecological Footprint Analysis (EFA) attempts to measure the desired land acreage for sustaining production, consumption, products or services (Wackernagel and Rees 1997; Wachernagel et al. 1999; Senbel, McDaniels and Dowlatabadi 2003). 5) Sustainable Process Index (SPI) is used to address the ecological pressure caused by material consumption and waste generation of certain manufacturing process or technical craft (Krotscheck and Narodoslawsky 1996).[4]

The research contents, analytical scales, model features and policy implication of these five main physical-based approaches are distinguished in Table 2.1. Apparently, each of these approaches has strengths and weaknesses. MFA methods compete against other candidates with distinct traits, involving wide compatibility of various kinds of material flows, a flexible formulation structure, adequate data requirements and relevance for policy analysis. This study, therefore, will apply an MFA approach to analyze P flows within China's economy.

[4] Besides these approaches, some other physical-based models are recently developed. Hinterberger and his colleagues, for instance, developed the indicator of Material Intensity per Unit Service (MIPS) to replace conventional natural capital analysis (Hinterberger, Luks and Schmidt-Bleek 1997; Hinterberger and Schmidt-Bleek 1999). Eder and Narodoslawsky (1999) adopted the concept of Dissipation Area (DA), i.e. the land acreage per unit load of pollutant emission, to represent the environmental impact of industrial activities.

Since MFA approaches have been applied in different ways, we now turn to focus on them in details in the following section.

Table 2.1 Methodological Comparison of five physical-based approaches

Items	MFA	PIOT	LCA	EFA	SPI
Research objectives	Material metabolism	Physical relationship among industries	environmental impacts of entire lifespan	Ecological carrying capacity	Environmental pressure; ecological potential
Research objects	Resources; production; waste; recycle; accumulation; hidden flows [a]	Resources; production; waste; recycle; [b] accumulation; hidden flows	Resources; production; waste; recycle; hidden flows	Resources; waste	Resources; production; waste; recycle; hidden flows
Research targets	National and local economy	Goods; industry; national economy	Production; service	Consumption; National and local economy	Production; consumption
Spatial scale	National; local	National; local [c]	National	National; local	Local; national
Temporal scale	Year	Year	Lifespan of production or service	Year	Lifespan of manufacturing or consumption
Measure units	Weight; volume	Weight	Weight; acreage; volume	Acreage	Acreage
Model structure	Various kinds, less standardization	Modularization, flexible expansibility	Rigid formulation	Indicator aggregation [d]	Indicator aggregation [d]
Data requirement	Adequate	Large	Large	Small	Adequate
Policy simulation	Adequate [e]	Strong	Adequate	Weak	Weak
Appropriateness of indicators [f]	Adequate	Weak	Weak	Strong	Strong

Sources: based on Bouman et al. (2000); Daniels and Moore (2001); Daniels (2002).

Notes: [a] Hidden flows refer to the human-induced material movements which do not enter into the economy but stress on the environment (see discussion in the next section); [b] Only the recycling loops that occur among industries are taken into consideration; [c] Most applications stick to the national level; [d] Calculation results are presented in the form of an aggregated indicator; [e] Different MFA models exhibit distinct performances of policy simulation (see Table 2.2); [f] The appropriateness represents to what extent the material indicators derived from these studies can be used as criteria for sustainable development.

2.2.2 Material flow analysis approaches

MFA approaches investigate systemically the physical structure and dynamic mechanisms of the societal processing of materials, ranging from resources extraction, manufacturing and consumption, to waste disposal and recycling. Aimed at inquiring into alternative solutions for reconciling economic development and environmental improvement, these approaches are quantitatively operationalized to investigate the causality among resources consumption, waste emission and underlying economic activities. The Principle of Mass Conservation can be further formulated as “Input = Output + Net Additional Stock (NAS)”, where “NAS = Accumulation – Release”.

Based on original research purposes, MFA approaches can be divided into two categories (Bringezu 2000). The top-down analytical methods start from specific ecological issues or environmental problems, e.g. global climate change caused by a disturbed carbon cycle, river pollution by heavy metals, or water eutrophication because of nutrient surplus. The desired strategies and implementation schemes for resources conservation, ecological restoration and pollution control can be proposed, by exploring inefficient use of materials or substances. In contrast, the bottom-up approaches, the second cluster of MFA tools, are based on the motivation for examining material productivities of individual enterprises, industrial sectors, regional and national economies. They have been applied to measure the degree of sustainable development by accounting inputs and outputs of materials or substances. The overall improvement of material throughput relies on recommended priority of dematerialization or key measures for ecological restructuring.[5]

MFA approaches have been widely applied for various materials, products and substances, along with different social, economic and geographic units, as shown in Table 2.2. The element-based MFA, i.e. Substance Flow Analysis (SFA), usually targets on those metals and nonmetals that impose vital influences on the economy and the ecology, by focusing on the intensities and paths of allocation and storage in the realms of production and consumption as well as on termination in the environment. These SFA studies in general attempt to address the spatial distribution or temporal dynamics of selected elements within environmental sensitive regions or at the national level. With the causality between substance metabolism and related environmental problems, this research tradition can be applied directly for quantitative evaluation of ecological risks. The main difficulty of SFA modeling lies in data processing, because most substances are processed in the form of bulk materials and thus few substance flows have been recorded independently. Therefore, a successful application of SFA approaches depends on whether the element-based turnovers can be derived from formal economic and environmental data involved in various statistical yearbooks, monitoring tables, industrial databases and academic reports. Because of the diversity of statistical systems in

[5] Dematerialization refers to an absolute or relative reduction in use of natural material per unit of value or product output, aiming to achieve the same level of productions and services by minimized natural resources (Ayres 1994; Schütz and Welfens 2000; Bartelmus 2002). Different from conventional end-of-pipe approaches, this strategy advocates systematic measures, for instance, involving ecological design, extending producer’s responsibility, product-oriented environmental policy and returnable deposits, for materials management on the entire production-consumption chain, from the extraction of raw materials, through design and production, to the waste disposal (Ayres 1992; Frosch 1992; Elliott 1997; Powers and Chertow 1997; Andrews 1999; ERM 1999; Ruth 2001; Graedel and Allenby 2004). Compared to dematerialization, the concept of ecological reconstructing or ecological restructuring is not confined with a direct improvement of material use efficiency, but emphasizes on structural adjustment and functional integration at the macro level (Jänicke 1997; Pastowski 1997; Picton and Daniels 1999; Schandi, Huttle and Payer 1999; Femia, Hinterberger and Luks 1999; Jänicke, Binder and Mönch 1997; Jänicke 2000). It prefers a strategic transformation of production and consumption to reorganize and rearrange material flows within the economy, relying on ecological principles in regard to the structure and operation of natural ecosystems (Simonis 1994).

different countries, the analytical procedures and results of these SFA approaches have not been normalized yet into a standard framework.

Table 2.2 Categories and research paradigms of material flows analysis

	Physical dimension		
	Substances	*Materials or products*	*Bulk-materials*
Social dimension: *community/ city*	* Watanabe & Ito (1997) [a]	** Wolman (1965); Brunner et al. (1994); Decker et al. (2000); Kincaid & Overcash (2001)	*
Economic dimension: *enterprise/ sector*	** Dijk et al. (1996); Fujie & Goto (2002)	*** Chang et al. (2000); Verfaillie & Bidwell (2000); Korhonen et al. (2001)	*
Geographic dimension: *region/ nation*	*** Jolly (1992); Loebenstein (1994); Llewellyn (1994); Van der Voet (1996); Moolenaar et al. (1998); Sznopek & Goonan (2000); Kramer (2002); Ober (2002); Smith (2002); Hansen & Lassen (2003)	*** Jänicke (1997); Kelly (1998); Matos & Wagner (1998); Picton & Daniels (1999); Hekkert et al. (2000); Patel et al. (2000); Joosten et al. (2000); Fenton (2001)	*** Adriaanse et al. (1997); Spangenberg et al. (1999); Matthews et al. (2000); Bringezu & Schutz (2001); Eurostat (2002); Amann et al. (2002)
Main objects	*Heavy metals:* Cd, Pb, Cu, Al, Hg, Mn, Mg *Nonmetals:* C, Cl *Nutrients:* N, P	*Compounds:* PVC, CFC [b] *Raw materials:* construction materials; steel, minerals *Products:* battery, car, plastic	TMR, DMI, TDO, DPO
Policy relevance	Adequate	Strong	Weak

Notes: [a] * rarely applied; ** less applied; *** widely applied; [b] PVC – polyvinyl chloride, CFC – chlorofluorocarbon.

A second cluster of MFA approaches concentrates on the impacts of production and consumption of raw materials and products on related environments. These MFA models are basically developed to enable and conduct quantitative assessments of material productivity as well as material reuse and recycling. As the results tightly relate to productive processes and management activities of enterprises or industrial sectors, this kind of MFA models has been applied to identify dematerialization solutions and to compare material use efficiency among different enterprises or industrial sectors. Therefore, these models have increasingly become a key component of environmental management, adopted by a number of medium and large sized corporations.[6] Another important application of these MFA approaches occurs in analyzing the network relationships of raw materials, byproducts and waste utilization among

[6] The material flow studies on the whole life cycle of products and techniques create new opportunities for reconciling economic and environmental performances of enterprises and corporations. The improvement of eco-efficiency, relying on a series of managemental, operational and technological measures derived from the material flow studies, has been considered as one of the main industrial practices conducted by ecological thoughts. Henceforth, many multinational companies, including AT&T, Lucent, GM, Motorola, Kodak, Shell, Xeron, Dow and ABB, have promoted the research and practice on eco-efficiency and achieved considerable progresses. (WBCSD 1998; WBCSD 2000; Verfaillie and Bidwell 2000).

some enterprises located in a specific region, for instance, an ecological industrial park.[7] In addition, compared to substance flow analysis, these product-based MFA methods are less equipped to elaborate the causal interrelations between material uses and possible environmental issues. With respect to methodological formulation, the analytical framework, initialized by Kostick (1996) and subsequently adopted by the Interagency Working Group on Industrial Ecology, Material and Energy Flow at the Presidential Committee of Environmental Quality (Wagner 2002), has been widely applied. Other paradigmatic models include STREAMS (Joosten et al. 1999; Joosten, Hekkert and Worrell 2000) and MEFIS/NAMEA (Schoer et al. 2000) developed by Dutch and German researchers, respectively.

The third application in the form of bulk material flow analysis belongs to the discussion of the overall efficiency of total material use within local or national economies. These bulk-MFA approaches are inclined to apply aggregated indicators to quantify material productivities, rather than to exposit the detailed physical structure of material distribution and utilization within the economy. Therefore, the bulk-MFA provides an appropriately analytical basis for strategic assessment of national or local sustainable development. In particular it generates standardized and simple indicators for comparing the material use levels among different regions or countries (Bringezu 2001). The application of bulk-material flow analysis has led to a reflection on national environmental policies and improvements. While these advantages are inspiring and promising, the methodological legitimacy of the bulk-MFA has been discussed. First, since the environmental impacts caused by production and consumption of each individual material differ significantly, the methodology of simple adding-up all material fluxes could be criticized. Moreover, the deduced implications for improving material use efficiency could hardly lead to feasible and practicable directives for policy design (Kleijn 2001). Second, because the contribution of material use to economic growth varies as well depending on different materials, the use of a combination of total material turnover with economic growth as the indicator of physical efficiency can not provide a reasonable criterion for comparative analysis among different economies (Bringezu 1999).

The bulk-MFA analytical framework developed by Matthews and his colleagues (2000) is the one that has been broadly accepted for international comparative analysis, especially by European countries (Eurostat 2001; 2002). As illustrated in Figure 2.1, both input and output sides of the economy and the materials retained in the economy in the form of infrastructure and long-lived durable goods are taken into consideration. Based on this analytical scheme, a number of critical indicators representing material use efficiencies are subsequently introduced. On the input side, the direct material input (DMI), encompassing both domestic extraction and import, represents the turnover of the material flows entering into the domestic

7 The concept of ecological industrial park (EIP), coming forth in the early 1990s, suggests an efficient material use network among enterprises within a specific locality, i.e. industrial park, aiming to optimize the utilization of various material resources, involving raw materials, co-products, by-products and waste (Côté and Cohen-Rosenthal 1998; van Koppen 2004). The innovative trans-formation of industrial production has provided valuable practical experiences for ecologically reconstructing material flows at a regional level (Erkman 1999; Posch 2002). While the gradually spontaneous industrial symbiosis at Kalundborg in Denmark has promoted worldwide imitation, many local EIPs have emerged since the 1990s (Fleig 2000; Côté and Peck 2002; Posch 2002; Lowe and Geng 2003; Heerses, Vermeulen and Walle 2004; Roberts 2004).

economy. The total material requirement (TMR) includes both the direct input of commodities to the economy and the "hidden" flows of materials, which are associated with making commodities available for economic use but do not themselves enter the economy.[8] Different from the foreign hidden flows (FHF) that occur during production of commodities imported by the economy, the domestic hidden flows (DHF) represent a simultaneous input and output for the purpose of mass balance accounting. Besides DHF, the domestic processed output (DPO) indicates the total weight of materials, occurring at the processing, manufacturing, use and disposal stages of the entire economic production-consumption chain. While exported materials are excluded because their waste occur in other economies, the total domestic output (TDO), combining DPO with DHF, then represents the total quantity of material outputs to the domestic environment caused directly or indirectly by human economic activity. With respect to materials used in buildings and infrastructures or incorporated into durable goods, the difference between the quantities of new materials added to the economy's stock and of old materials released from the stock is accounted for in the net additions to stock (NAS).

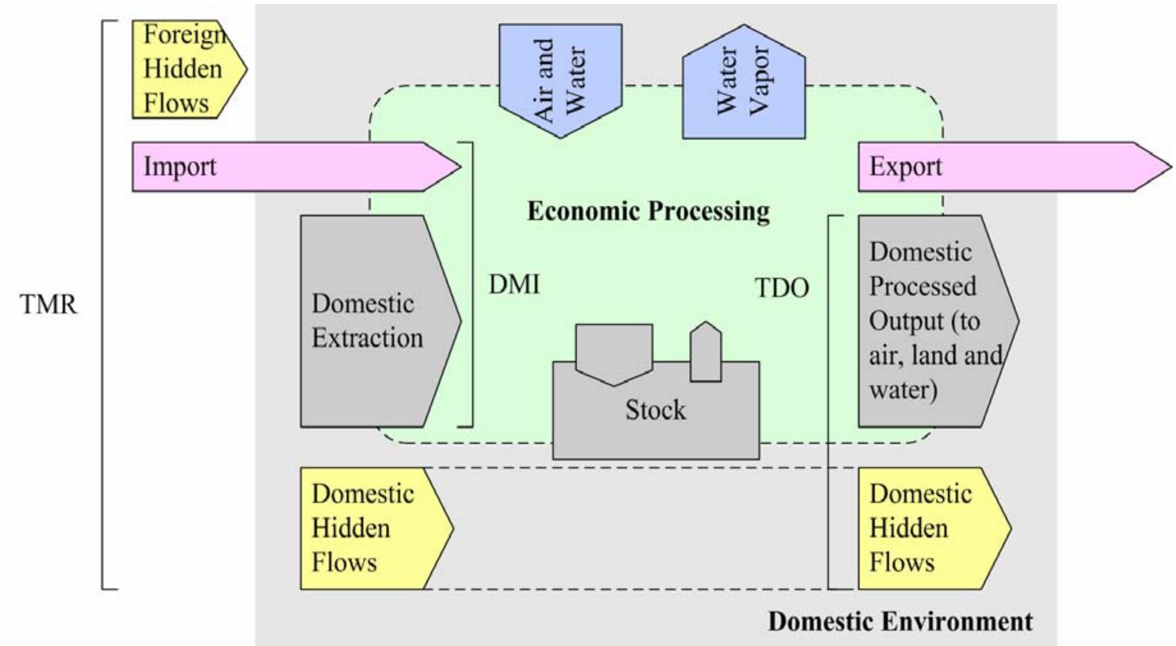

Figure 2.1 The schematic representation of bulk-MFA analytical framework

Sources: cited from Matthews et al. (2000).

MFA provides a holistic understanding on the underlying physical mechanism of the economy and thus enables decision-makers to distinguish the substantial causes that are related to environmental issues. Reconstructing and reorganizing material flows from an ecological

8 Hidden flows occur at the extraction and harvesting stage of material cycle. They comprise two components: ancillary flows and excavated flows. The former cluster of flows refers to plant and forest biomass that is removed from the land along with logs and grain, but is later separated from the desired material before further processing. The latter flows include the overburden that must be removed to permit access to an ore body, and soil erosion that results from agriculture. See details in Matthews et al. (2000). The hidden flows associated with mining activity and crop farming has been traditionally absent in most economic statistics. Therefore, the negative environmental impacts have been ignored to a large extent.

perspective to improve physical efficiencies of the economy, has become one of the essential objectives of environmental policies since the 1990s (Ayres 1994; Ayres and Ayres 1996). Moreover, a request for efficient monitoring of materials turnover and productivity within a national environmental accounting system (NEAS), as one of the most outstanding proposals of these material flows studies, has received increasing attention worldwide since the mid 1990s (Wernick and Ausubel 1995; Spangenberg et al. 1998; Berkhout 1999; Bartelmus and Vesper 2000; Eurostat 2000; Isacsson et al. 2000; OECD 2000; Schoer et al. 2000).

2.2.3 Modeling P flows by a SFA approach

Systematically organizing the quantitative information on the societal P metabolism, i.e. modeling P flows, is the essential gateway to formulate and identify the P-related physical structure of the economy. In principal, all these three material flow analysis approaches can be applied for this study. Comparatively, a SFA model is the most appropriate, with respect to the environmental problem of water eutrophication, for two reasons. First, as P is mainly mobilized as an indispensable nutrient throughout the whole economy and probably is leaking from all involved social sectors of production, consumption and waste treatment, a comprehensive analysis is desired. A process or product based approach cannot provide an insight into the entire lifespan of P. Although the production of P fertilizers dominates the overall turnover of societal P throughput, tracking P flows associated with fertilizer production and consumption will ignore other important information on, for instance, livestock production and detergents use. Second, although P is actually processed in various forms of bulk materials, an aggregated model of P flows can hardly generate a reliable result to identify the essential physical characteristics of P metabolism. By contrast, an element-based account purifies P from the disturbance of other substances and materials, and thus provides a uniform and measurable basis for P.

This study applies a static SFA approach for the quantitative formulation of P flows for two reasons. First, while dynamic SFA models can analyze the distribution and storage of materials within the economy and thus predict the future economic and environmental impacts, they have been basically operationalized to handle the materials or substances embedded in durable products. Since a major part of P is used as the essential nutrient compared to the small quantity of P in forms of various industrial products, the P stock within the economy is marginal. Second, the applicability of dynamic SFA models is largely complicated by their high data requirement compared to that of static analytical approaches. This is an essential criterion for selecting appropriate methods to implement P flow analysis in this study, because both economic and environmental data in China are relatively insufficient. Both the methodological and practical reason justify the application of a static SFA approach in this study, certainly as a start.

In order to improve the explanatory capacity of SFA models, this study will take a few further steps. First, since a static SFA model fails to examine the ecological reform tendencies or structural changes with respect to societal P throughput, an ecological restructuring analysis, based on indicators, is introduced (see Chapter 3). Second, based on primary studies on P flows, we will further develop and elaborate several substantial principles for data processing (see Chapter 5.2). They will conduct a modification of the initial SFA models to produce a more reliable output. Third, as SFA models are often interpreted in different ways, this study attempts to provide a normative formulation. By using some aggregated terminologies proposed by Matthews et al. (2000), the output of SFA models are further elaborated. This facilitates a further comparative analysis between different SFA models (see Chapter 3 and 4; Chapter 5.5).

The simple modeling principle, i.e. balancing mass flows, makes the application of SFA approaches theoretically easy, regardless of difficulties in data collection and processing. The definition of system boundary becomes an essential step in putting SFA approaches into practice. The general features of the natural cycle and the societal metabolism of P, as discussed in Chapter 1, facilitate the operationalizing process for modeling P flows in China. Based on the environmental and economic characteristics of P substance, some important implications can be concluded. First, the terrestrial deposited P, i.e. the geologically prospected mineral reserve, is the unique available source of human society to obtain P. Since the formation of P minerals takes over one million year, the total P reserve under current technological level is finite and nonrenewable. Second, while human activity has limited influence on the P cycle between continents and oceans, the loops and intensities of P flows within terrestrial and soil spheres have been significantly reshaped. Third, the atmosphere is a relative small sink of P. Moreover, the material connections with other sinks, including both sedimentation and volatilization of P, are less intensive, because the gaseous P in the form of PH_3 is unstable in air. Fourth, the leakage of P from various sources ultimately accumulates in cultivated soil, natural soil and water. The latter enrichment is responsible for eutrophication. Fifth, P in soil mainly concentrates in the superficial layer, because its longitudinal leaching rate is obviously slow compared to the transversal movement. Thus, P is less likely to entrophicate the groundwater. Therefore, only available mineral P reserve on the input side, and cultivated soil, natural soil and freshwater on the output side, within the national territory, are taken into consideration. Besides, the commodity import and export across the national boundary are supplemented to the input and output sides, respectively. The oceanic sphere and atmosphere as well as groundwater are excluded from the P flow models. The natural processes, encompassing wind erosion, water erosion, sediments release, etc., are also eliminated from the discussion on societal P flows.

The large national territory and complex hydrological condition undermine a direct causal linkage between the P load and eutrophication that occurs in some waterbodies. However, the P flow analysis for a national economy remains legitimate. Different from conventional analysis of pollutant sources and end-of-pipe control strategies, this study focuses on the macro-structural failure of the entire P throughput and the necessary substantial improvement of the social production and consumption in order to achieve the environmental objective. Thus, the top-down method can make sense for eutrophication control. In addition, the change in hydrological condition of surface water caused by the construction of reservoirs is likely to lead to an increase of eutrophication, if the surplus P comes from surrounding industries, farming and breeding practices and human sources. The construction of the Three Gorges project, for instance, has significantly reshaped the fundamental hydrologic condition of the corresponding section of Yangtze River. Accordingly, eutrophication becomes one of the key potentially ecological risks that parallel the slow-down of water velocity.

2.3 Implementation of Environmental Policy Evaluation

Putting the findings of material flow studies into practice is not an easy societal response to ecologically irrational material processing. All these physical-based analytical methods cannot be applied for answering the second research question. The underlying social-economic factors, ranging from organizational fit, institutional capacity, technological trajectory, cost and benefit, to knowledge, preference and communication, shape the societal mechanism of current material throughput and structure the social response to the proposed transformation of future material use. Accordingly, these factors are of central importance in understanding the societal causes of the physical traits of material use and thus in reinventing regulatory and incentive arrangements towards ecologizing material flows within the economy (van Koppen and Mol 2002).

The socio-economic dimension, however, has been largely ignored in material flow studies. First, while most of these studies have concentrated on modeling techniques and data processing, they have paid less attention to the environmental policies on material production and consumption. Second, as a consequence of the first reason, many policy recommendations, oriented at redirecting ecologically unfavorable material flows, prove infeasible in practice. Third, although some dynamic models have been developed for predicting the potential economic and environmental benefits or risks of different policy alternatives or scenarios, social and economic factors can hardly be taken into account by these proposed schemes. The disjunction between material flows models and policy analysis leads to a substantial gap in the research regime of material metabolism, and thus considerably weakens the practical significance of research outputs. Therefore, there is an urgent need to pay more attention to the social, economic and regulatory underpinnings of the manifested material flows (Dieu 2003; Ayres 2004).

2.3.1 Methods of environmental policy evaluation

The practices of policy evaluation emerged in the seventeenth century and have been systematically implemented since the 1930s, represented by three evaluation programs focusing on the improvement of rural sanitation in Middle East, the New Deal of the US President Roosevelt, and the industrial labor productivity, respectively (Patton 1980). After the Second World War, policy evaluation has been widely applied on various social topics, involving military equipment, technological innovation, higher education, public health, transportation, urban planning, poverty reduction and community construction (Rossi, Freeman and Lipsey 2002). Today, policy evaluation, as a subject in the field of social sciences, entails the systematic identification of policy effects and impacts relying on learning from and application of various related theories, models and analytical methods (Costanza 1993; Patton and Sawicki 2002).

While the key sense of the term 'evaluation' refers to the process of determining the merit, worth or value of something, or the product of that process, a scientific definition of policy evaluation is ambiguous. A key text lists 33 different models of evaluation and concludes that it is not practical to adopt a single definition (Chess 2000; Mickwitz 2003). The general idea behind evaluation is that systematic analysis of policy processes, outputs and outcomes helps to make better decisions in the future (Patton 1980). More concrete purposes can be addressed as: promoting program management, facilitating transparency and accountability, reducing risk and uncertainty, improving process, and fostering learning (Bellamy et al. 2001; Rossi, Freeman and Lipsey 2002). More importantly, since policy evaluation itself is always conducted in a conversational and negotiating way, information exchange and experience sharing become inherent features of this approach.

Although evaluation came later to the environmental field than to many other policy branches, a rapid development is under way in many countries (Cropper 1992; Jänicke and Weidner 1995; Dijk, Leneman and van der Veen 1996; Button and Nijkamp 1997; Daskalopoulos, Badr and Probert 1997; CEQ 1997; Rousseau and Proost 2002; van der Veeren and Lorenz 2002), and also at the European Union (EU) level (EEA 2001). More importantly, policy makers and administrators recently have expressed increasing interests in demand for environmental policy evaluations (Mickwitz 2000). The task of evaluating policies, for instance, is clearly formulated in the 6th Environmental Action Program for the EU (1600/2002/EC), which was finally adopted in June 2002 (European Parliament and the Council of the European Union 2002).

Environmental policy evaluation methods can be divided into three main categories (Patton and Sawicki 2002; Rossi, Freeman and Lipsey 2002). The oldest evaluation model is the goal-achievement model oriented to test whether the results are in line with the goals. It is apparent that this simple-formed model has obvious weaknesses, consisting of the disregards of side effects, unanticipated impacts, costs, and appropriateness of the initiative goals. The method of goal-free evaluation was developed subsequently in response to the criticism. While the side effects, universally occurring due to the complexity and uncertainty of environmental policies, are taken into account, the goal-free model leaves the cost criteria unanswered. While the previous models depart from either the objectives of the policy or the observed effects, the stakeholder model or client-oriented evaluation starts from those affected by the policy, and sometimes from those involved in carrying out the policy. The base for the third evaluation model is then the desires, expectations and concerns as expressed by clients or stakeholders or by their needs as determined by the evaluator. As argued by Mickwitz (2000), although the three forms of evaluation are by no means the only ones, they reveal enough of the issues involved in the choice of evaluation model. Besides, some other scholars presented a different typology, dividing policy evaluation analysis into three main groups: effectiveness models, economic models and professional models (Patton and Sawicki 2002; Rossi, Freeman and Lipsey 2002).

An environmental policy model in general takes a number of fundamental social, economic and environmental dimensions associated with the policy design and implementation into account. These elements are often considered as inputs, outputs, outcomes and impacts in evaluative terms. Again, likewise as the definition of evaluation itself, descriptions of these elements considerably differ between models. As defined by European Environment Agency (EEA 2001), for instance, the term of inputs refers to various administrative, human and financial resources dedicated to the design and implementation of a measure. While the outputs represent the tangible results of a measure imposed on target groups, the term outcomes is used to describe the responses of the target groups to these outputs, i.e. the changes in behavior. The ultimate expected and unexpected effects, caused by these changes, on the environment and human health as well as the economic society as a whole are discussed in terms of the impacts. Since these elements articulate the essential components of the policy process, they are desired to be fully addressed in a professional policy evaluation study.

The basis of policy evaluation models lies in the development of appropriate performance criteria to monitor and assess policy design, implementation and ultimate impacts (Chess 2000; Bellamy et al. 2001; Patton and Sawicki 2002). In principal, most important criteria relate to the general profile of policy (relevance, effectiveness, impact, persistence, flexibility, predictability, technical validity, etc.), the examinations of the economic efficiency (cost-benefit, cost-effectiveness, etc.), and the judgments on the functioning of democracy (legitimacy, transparency, equity, social justice, etc.) (Bellamy et al. 2001; van der Veeren and Lorenz 2002; Mickwitz 2003). Restricted by data collection, implementation cost and research efficiency, however, any evaluation practice can only concentrate on several selected criteria. More often, screening feasible evaluation models and schemes to produce scientific and reliable results which can support better policy design and implementation is likely to be an iterative process.

An evaluation scheme can be conducted in two ways, i.e. retrospective evaluation (*ex post*) and prospective evaluation (*ex ante*), based on the availability of data on the actual outputs and outcomes caused by the policy implemented over a short period or to be carried out. While *ex post* evaluation focuses on the effectiveness of existing measures in meeting their environmental objectives, *ex ante* evaluation is performed aiming at the possible impacts, in particular the environmental impacts of new polices including the alternative of no action and the proposals for legislation and publication of the results (EEA 2001). Ex-ante evaluation is given increasing attention and has become particularly important in the field of environmental policy, since one type of *ex ante* evaluation – environmental impact assessment – is nowadays mandatory in many countries.

It has been argued that implementation of environmental policy evaluation is rather difficult due to a number of key characteristics of environmental problems, e.g. the externality, complexity, uncertainty, long-term and diverging effects as well as impacts, and conflicting goals and belief systems of involved stakeholders (Weale 1992; Chess 2000). Because of the complexity and uncertainty, for instance, negative environmental effects either inside or outside the target area are in general difficult to anticipate (Jänicke and Weidner 1995). Moreover, since the time between action and ultimate effects of an environmental policy is liable to be long due to the nature of environmental processes, not all effects can be evaluated at any point in time. As a consequence, many concepts of environmental policy evaluation remain insufficient defined and the practices to a large degree are still not well standardized (Nijkamp and van den Bergh 1997; van der Veeren and Lorenz 2002; Mickwitz 2003).

2.3.2 Evaluating P-related environmental policies in China

In order to examine whether the current Chinese environmental policies are successfully working on P flows, a comprehensive and efficient evaluation method is needed for three reasons. First, as P flows are connected to many social activities (simply, say production, consumption and waste treatment), P-related environmental regulations are implemented in different forms and ways. In other words, all the elements underlying policy design and implementation differ from one policy to another. Consequently, different policies cannot be easily evaluated by a simple standard or an inflexible model. Second, while many evaluation models focus on policy implementation, they fail to provide valuable information on policy design. Due to the difficulty of eutrophication control, environmental policy objectives are often questionable. This suggests that the appropriateness and/or the feasibility of policy objectives are important criteria that should be taken into consideration. Third, the amount of evaluation criteria should be limited, balancing sufficiency and feasibility of an evaluation scheme. Among others, an economic efficiency should be considered, since the huge investment on domestic water eutrophication control has been argued in China.

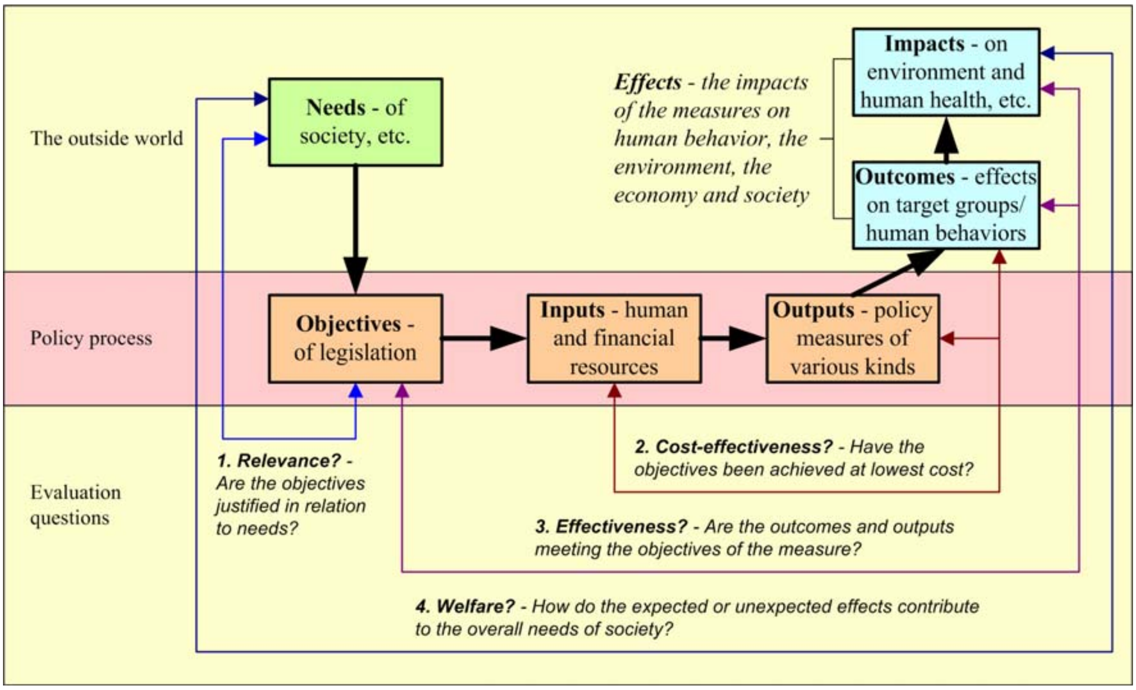

Figure 2.2 A framework for environmental policy evaluation

Sources: cited from EEA (2001).

This study applies the evaluation framework developed by EEA in 2001. This method originated from a consensus that all members of European Union should report their "state of action on the environment", with respect to their responses to EU legislation, in a formalized way. It presents a standardized framework to facilitate the collection and organization of information on environmental regulation and thus provides opportunities for comparing environmental performances and achievements among European countries. In short, this framework enables to examine environmental policies on four aspects, in conjunction with the process of policy design and implementation, as demonstrated in Figure 2.2. First, the evaluation approach compares policy objectives with the 'real' needs society has to meet for problem solving. Second, direct and indirect costs, input to policy design and implementation, are related to policy outputs and outcomes to test to what extent the policy is economically efficient. Third, expected and unexpected environmental effects caused by policy implementation are compared with initial objectives to assess if the policy works in an effective way. Forth, the change of overall social welfare is evaluated to generate an insight into the consequences of policy concerning its overall impacts on economy and society.

This scientific proposal of EEA is favorable for this study, as it meets the methodological requirements as discussed above. First, this model introduces the element of social needs, which enables evaluators to compare the environmental or regulatory goals with the needs of stakeholder. By doing so, it provides a comprehensive evaluation scheme that integrates the three basic evaluation models into one, and thus offers a flexible scheme to deal with different kinds of environmental policies. Second, this framework creates a systematic method in relation with the whole process of both policy design and implementation. The criterion of policy relevance enables to reflect on the initial motivation of eutrophication control strategies. Third, this model focuses on four criteria in regard to objective setting, environmental effectiveness, economic efficiency and overall social impact. These criteria, except for social democracy, sufficiently compass important general, economic and social issues involved in most policy evaluation studies. In addition, implementation of this evaluation framework can also contribute to a test of the current state of environmental statistics, monitoring and survey in China, and thus provide valuable recommendations on future steps to improve the infrastructure and capacity of environmental regulation and management in this developing country. In Chapter 6, we will operationalize this model for evaluating three selected environmental policies related to P flows.

2.4 Research Strategy

In addition to an overall research on P flows and eutrophication in China, the research strategy of case study is also applied in this study for various reasons. First of all, since water eutrophication is characterized by specific local socio-economic profiles, the investigation focusing on the national level is insufficient. There is a need to connect the methodology to local P regime. By doing so, it can also provide an effective test of the applicability of our methodology for generating a reliable insight, at the local level, on the physical and social profiles of P flows. Second, local case study research can produce a tangible causality between the societal P metabolism within the local economy and the local environment, whereas this is more difficult at the national level. This in turn benefits the comprehensive understanding on the physical features of P flows, regarding the difference in spatial scale of material throughput across the country. Third, the differences in socio-economic conditions, industrial and agricultural structures, technological application and social behaviors among the various local economies, necessitate the inquiry of the local societal dynamics of intervention and regulation strategies on physical P flows. With respect to ecologizing P flows in the context of eutrophication control, this can further provide valuable implications for policy design and implementation, also for those decision makers at the national level.

As our methodology, combing the development of SFA model and the evaluation of environmental policy, is liable to be a time-consuming framework, we can only concentrate on one case, in addition to a study at the national level. The selection of such a case should consider the criteria of feasibility and appropriateness, while the result should have more generalizable meaning. The water quality of the three lakes, as discussed elsewhere in this thesis, has been regarded as the most important indicator for China's water environment. Of the three lakes, Dianchi Lake can be an adequate research target, because existing studies on lake eutrophication have shown distinct decreasing interests in the sequence of Taihu Lake, Dianchi Lake and Chaohu Lake. This suggests that the research basis for Dianchi Lake could be relatively appropriate, although it cannot compete with that for Taihu Lake. Also, participation in the entire implementation progress of the project "Non-Point Sources Pollution Control in the Dianchi Lake" sponsored by the Ministry of Science and Technology (MOST) and the Yunnan Provincial Government considerably facilitates the determination of the case. Therefore, the Dianchi Basin, one of the national key environmental protection watersheds, will be taken as a case study area, both for characterizing the social P metabolism and for evaluating the policies to reduce eutrophication. The whole geographical watershed of Dianchi Lake, thereby, is defined as the system boundary for this study.

2.5 Methods of Data Collection

The study on societal P metabolism at the national level relies on secondary data. The fundamental data of P flows for constructing the SFA model are based on national databases in the form of various statistical yearbooks for economy, industry, agriculture (farming and breeding), urban construction, rural development etc. (see Table 3.3; Appendix IV) In addition, China's Food Balance Sheet provided by the Food and Agriculture Organization of the United Nations (FAO) is applied to modify the nutrient network among crops, livestock and population involved in the national P flow model. Since P flows are only partially recorded in these official databases, related literature become the second important data source, particularly for selecting and verifying model parameters to determine the directions and quantities of associated P flows. Third, the evaluation of P-related policies requires quantitative information on costs, outcomes and impacts in EEA's terms. While the relevant data in general remain insufficient, some can be derived from the reports and releases of official census, general investigations and environmental monitoring. All these data are confined to the mainland of China, excluding the Specially Administrative Region of Hong Kong and Macao and Taiwan region.

The case study enables participated observation which facilitates the understandings on the local P flows and related regulations. Primary data were twice collected for the case study.[9] The first field investigation between 30 July and 10 August in 2000 consisted of a general survey. It focused on local socio-economic development, volume and emission of domestic wastewater and garbage, utilization and disposal of crop residues, situation of key pollution sources, features of village ditches and irrigation canals, and fertilizer application. The data were collected at one town and three villages of Chenggong County, the most intensive agricultural production area, located at the east lakefront of Dianchi Lake. It was conducted together with 4 staff at Kunming Institute of Environmental Science (KIES) and 6 graduates of the Department of Environmental Science and Engineering (DESE) at Tsinghua University, Beijing (see Appendix 1). The results were presented in a report "General survey on non-point source pollution of demonstration plot in Dianchi Watershed: Final report" (KIES and DESE 2000).

The second field survey, during 2 July to 17 July in 2002, focused on the relationship among local governance, farmland management, individual behavior of rural farmers and environmental consciousness. Data were collected by face-to-face interviews and questionnaires. First, a number of semi-structured and open-ended interviews were applied. The interviewees involved a number of local officials at Kunming governmental agencies, administrators at district/county level departments and representatives of villagers. The list of interviewees is presented in Appendix 2.

Table 2.3 The distribution of questionnaire samples of the field survey in 2002

	Chenggong County	Jinning County	Guandu District	Xishan District	Total
Number of samples	70	180	190	60	500
Number of villages	8	11	10	5	34

Second, 500 questionnaires were distributed to farmers and villagers living in five towns and eight counties around Dianchi Lake. The geographic allocation of the interviewed persons was determined, as shown in Table 2.3, based on the amounts of rural population and cultivated areas at village level among four sub-districts of Kunming Municipality. The face-to-face interviewing of rural households was conducted by 12 recruited volunteers at Southwest Forestry College (SFC) at Kunming (see Appendix 1). Before starting the field survey, all of them attended one-week training. The main training contents consisted of an introduction to social survey methods, especially conducting questionnaires. The main statistical results of questionnaires were presented in Wang (2003).

[9] Both investigations are financed by the project "Non-Point Sources Pollution Control in the Dianchi Lake", committed by DESE at Tsinghua University, Beijing. Many faculties and students at DESE, staffs at KIES, as well as the volunteers from SFC had contributed to these investigations.

The main secondary data sources for inquiry of the local P societal metabolism within Dianchi Basin consisted of Kunming Statistical Yearbook and research reports and administrative documents provided by a number of local agencies. They include Yunnan Institute of Environmental Science (YIES), KIES, Kunming Agricultural Bureau (KAB), and Kunming Dianchi Administration Bureau (KDAB). Again, literatures of empirical studies are also reviewed in this study (see Appendix III).

CHAPTER 3

Material Flow and Ecological Restructuring in China: the Case of Phosphorus[10]

Abstract

Environmental problems are closely related to society's processing of materials through the entire economy. Because neither traditional environmental nor economic analytical methods can provide sufficient insight into the physical dimension of economies, this article presents an integrated methodology, combining a substance flow analysis (SFA) approach and an ecological restructuring analysis. This approach is applied to phosphorus (P) in China, one of the most rapidly growing industrializing economies, in order to better understand of the economy's material use and its change over time. A static national SFA model is developed with statistical data from 1996. By tracking the national economy's P flows from origins to destinations, the critical P flows with respect to environmental impacts are identified. Based on the regime of national P flows, this article analyzes the degree of ecological restructuring by dynamically describing the structural changes of related critical P flows over the last two decades with a set of ecological restructuring indicators (ERIs). Finally, some potential and desired changes are discussed, with the goal of ecologizing the national P flow regime; that is, reducing the ecological impact of the national P flow regime. The methodology of this article illustrates its applicability and value for presenting an overall insight into the physical dimensions of national economies.

Keywords

Biogeochemistry, ecological restructuring, eutrophication, fertilizer, nutrients, substance flow analysis

3.1 Introduction

Many environmental problems we face today are a result of society's material use (Kleijn et al. 2000; Bouman et al. 2000). An investigation of the physical distribution, cycle and flux of crucial material elements within economies is an essential step in seeking more sustainable forms of development. Neither traditional monetary accounts nor environmental statistics, however, provide an adequate basis for tracking resource flows into and out of the economy. They record only a part of resource inputs, lose sight of some materials in the course of

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[10] This chapter contains an article published as Liu, Y., A. P. J. Mol, and J. N. Chen. 2004. Material Flow and Ecological Restructuring in China: the Case of Phosphorus. *J. Industrial Ecology* 8 (3): 103-120.

processing, and sometimes may entirely ignore major hidden material flows that do not enter the economy at all, such as soil erosion from cultivated fields. On the output side, monetary accounts and environmental statistics record few material flows that are not subject to regulation or classified as wastes requiring treatment (Matthews et al. 2000). In contrast, material flow analysis/substance flow analysis (MFA/SFA) focuses on the origination, development (utilization), and destination of materials with respect to their productive efficiency and the resulting environmental stress. Whereas economic flows are measured in dollars, tons of materials provides an obvious metric for determining the physical flows inherent in production and consumption activities – the flows of natural resources, goods, pollutants, and wastes brought forward by an industrial economy.

Phosphorus (P) is both an important nutrient and a major raw material for industrial products. It makes up 0.1% of the lithosphere and ranks 10th among all elements in the earth (Jiang 1999; Smil 2000). P is significant for, among other things, the growth of crops, but is also a critical factor for surface water eutrophication. In contrast to the natural environment where phosphorus is supplied through the weathering and dissolution of rocks and minerals with very low solubility (Steen 1998), human activity has significantly shifted the natural P metabolism, particularly after the modern fertilizer industry and phosphate chemical industries were introduced and broadly adopted after World War II (Stauffer 1998; Han et al. 1999; Foster 1999).

In China, one of the most rapidly growing industrializing economies of the world during the last two decades, P is increasingly becoming a significant concern in both national and regional environmental quality control, because China has experienced severe human-induced eutrophication since the 1980s (Jin 1995; Dokulil et al. 2000; Liu 2004). In terms of China's Environmental Quality Standard for Surface Water (GB 3838-2002), investigation of 25 major freshwater lakes showed that the average total phosphorus (TP) concentration in only 12 of those lakes was higher than the quality for Class V.[11] Only two lakes are not subject to eutrophication, with TP concentrations lower than 0.02mg/L. Six lakes encountered severe eutrophication with TP concentrations over 0.5mg/L (see Table 3.1).

This article investigates the national phosphorus flows in China, focusing on two objectives. The first objective is to identify the major economic activities at the national level that lead to P pollution of soil and water in China. The second objective is to assess the extent to which a delinking occurs between, on the one hand, natural P resource use and environmental P pollution, and economic development on the other hand. We start with an introduction to the methodology for MFA/SFA, combined with ecological restructuring analysis. A national P flow analysis for China is subsequently conducted using a static SFA method based upon 1996 data, with a focus on the critical flows that relate to environmental impacts. In the

[11] China's Environment Quality Standard for Surface Water (GB 3838-2002) was issued on 28 April 2002 and implemented from 1 June 2002. In this system of national standards, surface-water quality is classified based on utilization functions. Surface waters types categorized in Class I represent headwaters or natural ecological conservation areas. Class II and III refer to the first-level and second-level protection area of water resources, or to protected areas of valuable, rare, and common fish species, respectively. Class IV surface waters is suitable for general industrial utilization and indirect physical contact. Class V refers to the poorest water quality that can only be used for agriculture. If a water body has more than two functions, its water quality should fulfill the standard of the highest class related to the two functions.

following section, the development of P use and pollution is analyzed with respect to the critical flows over the last two decades, searching for evidence of ecological restructuring, dematerialization, or delinking in the Chinese economy. We conclude with a discussion of desired and potential measures for (further) ecologizing the economy in the context of national P flows. “Ecologizing” refers to redesigning the economy according to criteria of ecological rationality (Dryzek 1987; Mol 1995).

Table 3.1 The State Environmental Standard (GB 3838-2002) and the total phosphate (TP) concentration of 25 major lakes in China

State Environmental Standard (GB 3838-2002)	*Class*					*Beyond Class V*		*Total*
	I	*II*	*III*	*IV*	*V*			
TP Concentration for lakes and reservoirs (mg P/L)	≥0.01	≥0.025	≥0.05	≥0.10	≥0.20	>0.20	>0.50	
Number of lakes	9				4	6	6	25
Percentage of lakes (%)	36				16	24	24	100

Sources: Based on Si et al. (2000).

3.2 Material Flow Analysis and Ecological Restructuring

As traditional monetary accounting cannot generate a complete picture of the economy, physical approaches have been increasingly accepted by politicians and experts as a useful complement. Usually either MFA/SFA or life cycle assessment (LCA) methods are applied (Voet 1996; Moolenaar et al. 1998; Guinée et al. 1999; Kleijn et al. 2000; Matthews et al. 2000; Bringezu and Schütz 2001; Liu et al. 2004). It has become increasingly evident that the MFA/SFA approach has unique advantages in calculating physical flows (Bouman et al. 2000; Daniels and Moore 2001; Daniels 2002; Liu 2004).

Modern MFA/SFA approaches, originating from physical analysis in the context of industrial metabolism (Frosch and Gallopoulos 1989; Ayres and Simonis 1994), are an analytical tool for describing material regimes of the economy based on the Law of Mass Conservation – a rather simple rule used to illustrate the complex socio-economic systems of material metabolism – that is, inflow equals outflow plus accumulation. Analogous to long-standing practices for investigating monetary flows, MFA/SFA describes the origin of the concerned flows, the way they move through economic processes, and how and where they finally accumulate (Guinée et al. 1999; Kleijn et al. 2000). MFA/SFA can help to identify hidden flows and emissions, and detect accumulation of stocks in the economy and the environment. It can also identify which activities or products are primarily responsible for these flows and accumulations, and subsequently design interventions to create environmentally more effective and economically more efficient flows (Matthews et al. 2000).

The application of MFA/SFA can be highly variable, from “straightforward” material bookkeeping at a given moment in time to a dynamic analysis calculating future flows and stocks. In general, static models are often used to assess the origins of pollution problems and, in a manner comparable to input-output tables, to estimate the impacts of certain changes in material management. Dynamic models are mainly used to estimate the emissions and waste generation in the future (Bouman et al. 2000). Although regulations and economic instruments rarely take sufficient account of upstream or downstream effects when they seek to prevent or mitigate certain impacts, the MFA/SFA tool enables policy makers to trace the origins of pollution problems and to evaluate the appropriateness of these instruments for the entire societal management of materials and substances (Kleijn et al. 2000).

Ecological restructuring – the second methodological tool employed in this article – examines at a national level the degree to which economic growth delinks or decouples from material use (Simonis 1994; Jänicke 1995; Picton and Daniels 1999; Matthews et al. 2000; Mol 2003). It uses indicators to analyze and illustrate how physical intensities relate to economic dimensions, and thus enables the examination of long-term tendencies of material use compared to economic growth. Ecological restructuring is then a technical or organizational innovation of economic structures, resulting in improved material utilization, more efficient water and energy use, or less waste generation (Jänicke 1995). Ecological restructuring studies, however, generally focus on the overall tendency of national materials throughput and rarely on one specific substance. In doing the latter, critical (bulk-) material or substance flows should be distinguished from the complex industrial structure of production and consumption.

As an analytical tool, a static MFA/SFA can elaborate the processes of material handling within the whole physical economy by tracking the material from its origin, through different processing steps, to its final destination. But it fails to connect the physical flows to economic or societal aspects, and consequently it hardly contributes to the examination of ecological reform tendencies or structural changes with respect to material use and waste emission. This calls for an agenda to integrate MFA/SFA approaches with macroenvironmental-economic studies in the form of ecological restructuring indicators (ERIs) analysis. Only then are we able to fully understand the physical dimensions of the economy and the ecologizing of national material flows. This methodology not only opens up the black box of the physical regime by tracking material into and out of economic activities. It also explores the (increasing or decreasing) resource and environmental efficiencies of economic activities. Such a methodology provides a valuable insight into past policy making on both environmental effectiveness and economic efficiency.

3.3 Framing National Phosphorus Flow

In this section a static physical flow model is developed for P in China, making operational the methodology of SFA and subsequently applying it to China's statistical data for 1996.

3.3.1 The static SFA model

Because of the study objective and data limitation, we developed a static SFA model regarding China's national P flows for 1996, and the relevant data were mainly from the

China Statistical Yearbook.[12] Because the static model is based upon the conception that the physical economy operates in a steady state, it is assumed that there was equilibrium between inflow and outflow, that is, the fundamental formula of mass conservation can be further simplified as IN = OUT, regardless of stock and accumulation within the economy. As China becomes increasingly linked with the world economy, import/export of P material was also taken into account in the model. The P exchanges at the interfaces between atmosphere, geosphere, and hydrosphere were not taken into account.

Using an SFA modeling approach, the physical regime of P is decomposed from a "one-node" system in which the whole economy is regarded as a black box, to a physical framework in which each industrial or non-industrial process constitutes a separate node linked to a number of other nodes via inflows and outflows of P. Each node is quantitatively balanced, and all links between nodes are formulated by mathematical equations, thus generating a formal description of the system in which flows are dependent on one another.

The conceptual framework of the static national P SFA model is presented in Figure 3.1. It starts from P resources, through mining, chemical/fertilizer industries, agricultural crops, animal husbandry, food industry, fodder industry, food consumption, and environmental industries, to end up in the environment. The node labeled by P_i (i = 1, …, 13) represents process i handling P-related products. The P resource reserves and P commodities imported are indicated by R and I_j (j = 1, 2), respectively, and are treated as sources of P entering the national economy. The environmental sinks, represented as S_k (k = 1, 2, 3), include agricultural soil, natural soil and surface water. Although China's exports of both agricultural and chemical commodities form a second destination of P, only the former one, indicated by E, is calculated. The arrows, denoted by X_n (n = 1, …, 34), together with RP, IP_j, and EP, which refer to the related P flows occurring on the interface between the domestic and foreign economies, represent the P flows from one economic process to another. All these nodes and arrows constitute the skeleton of the SFA model.

The mathematical equations and variables in the model can be classified into three groups. *Exogenously fixed variables* are those that have fixed and known values and are independent of any other flow. For instance, the absolute quantity of the flow RP from node R to node P_1, representing the national P resource extraction, can be obtained from statistical sources. Secondly, there are the *dependent equations* that describe how a flow is dependent on other flows, such as $X_6 = K_4 \times X_1$ for calculating P pollutant load from fertilizer production. Thirdly, the model consists of *balancing equations* to balance the flows into and out of a given node in the context of the Law of Mass Conservation. For instance, the equation $X_2 = X_7 + X_8$ for node P_3 illustrates the economic efficiency and environmental impact of washing powder production. The mathematical equations are given in Table 3.2 and the relevant parameters are listed in Table 3.3.

[12] Insufficient data for cultivated areas have limited the analysis to the year 1996. The most reliable data of national cultivated area are given in China Statistical Yearbook in 2000 according to the latest agricultural census in 1996, which was conducted by the Ministry of Land and Resources, the National Bureau of Statistics, and the National Agricultural Census Office of China. For rich discussions of China's cultivated land, see works by Smil (1995), Prosterman and colleagues (1996), and Heilig (1999).

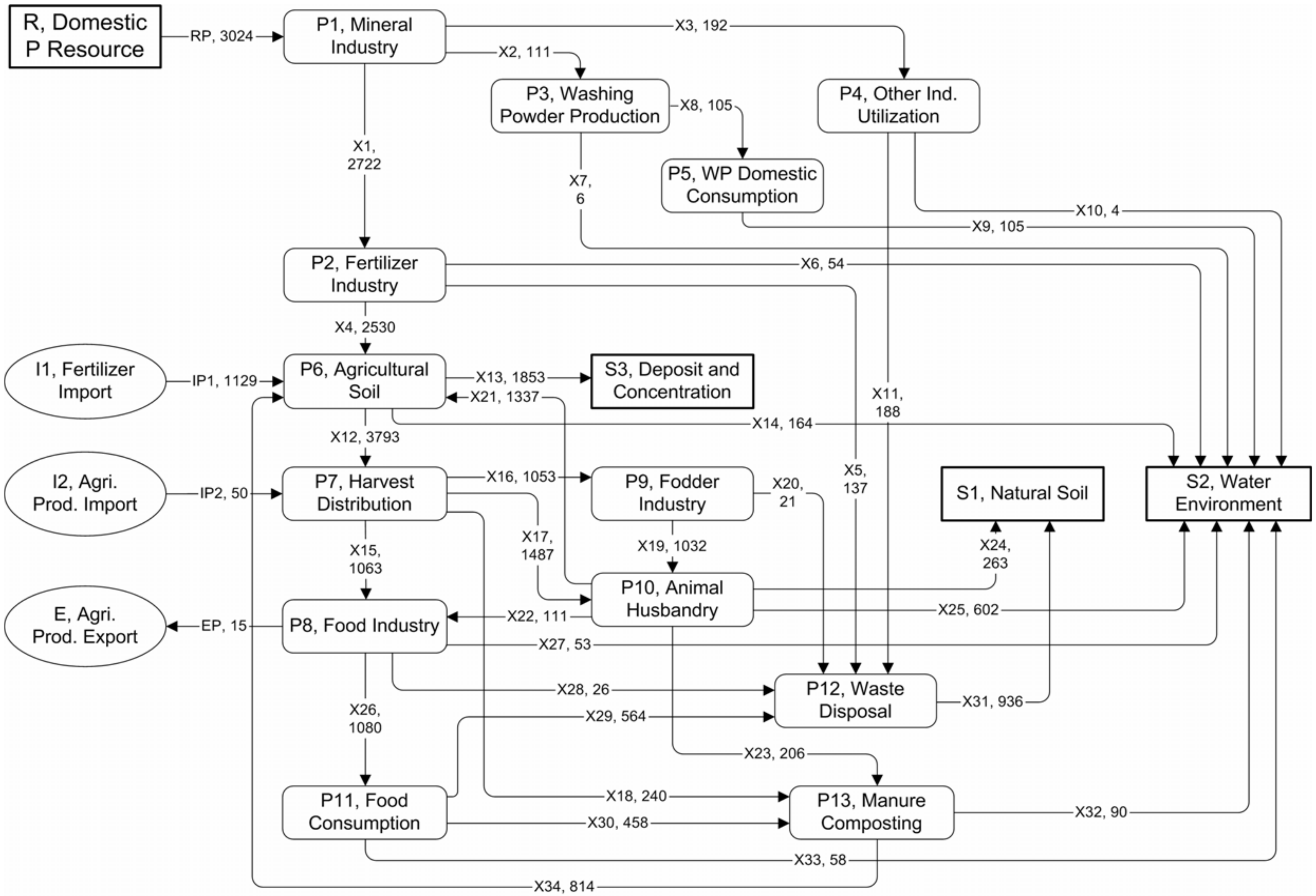


Figure 3.1 National static SFA model of China: the phosphorus flows in 1996 in thousands of metric tons (kt). Variables are explained in Table 3.2.

Table 3.2 Data and equations of the national static SFA model

Flow n.	*P flow description*	*From node*	*To node*	*Value in 1996*	*Equation*	*Coefficient description*	*Coefficient value*
RP	Ore exploitation yield	Source: domestic P resource	P_1: mining industry	3024	$RP = K_1 \times K_2 \times 22910$	K_1: conversion factor from P_2O_5 to P K_2: fraction of P_2O_5 content in ore	$K_1 = 0.44$ $K_2 = 0.3$
X_1	Ore applied for fertilizer production	P_1: mineral industry	P_2: fertilizer industry	2722	$X_1 = K_3 \times RP$	K_3: utilized proportion of P ore for fertilizer industry	$K_3 = 0.9$
X_2	Ore applied for WP production	P_1: mineral industry	P_3: WP production	111	$X_2 = X_7 + X_8$ (balance P_3)		
X_3	Ore applied for other industries	P_1: mineral industry	P_4: other industries utilization	192	$X_3 = RP - X_1 - X_2$ (balance P_1)		
X_4	Applied domestic fertilizer	P_2: fertilizer industry	P_6: agricultural soil	2530	$X_4 = K_1 \times 5751$		
X_5	Waste from fertilizer industry	P_2: fertilizer industry	Sink (S_1): natural soil	137	$X_5 = X_1 - X_4 - X_6$ (balance P_2)		
X_6	Load from fertilizer production	P_2: fertilizer industry	Sink (S_2): water environment.	54	$X_6 = K_4 \times X_1$	K_4: discharge coefficient of fertilizer industry	$K_4 = 0.02$
X_7	Load from WP production	P_3: WP production	Sink (S_2): water environment	6	$X_7 = K_5 \times X_2$	K_5: discharge coefficient of WP production.	$K_5 = 0.05$
X_8	Domestically consumed WP	P_3: WP production	P_5: WP domestic consumption	105	$X_8 = K_6 \times 1900$	K_6: average P content in WP	$K_6 = 0.055$
X_9	Load from WP production	P_5: WP production	Sink (S_2): water environment	105	$X_9 = X_8$ (balance P_5)		
X_{10}	Load from other P chemical industries	P_4: other industrial utilization	Sink (S_2): water environment	4	$X_{10} = K_7 \times X_3$	K_7: discharge coefficient of other P industries	$K_7 = 0.02$
X_{11}	Waste from other P chemical industries	P_4: other industrial utilization	Sink (S_1): natural soil	188	$X_{11} = X_3 - X_{10}$ (balance P_4)		
X_{12}	Nutrient uptake from agricultural soil	P_6: agricultural soil	P_7: harvest distribution	3793	$X_{12} = K_1 \times 5894 + 1200$		
X_{13}	Nutrient deposit in agricultural soil	P_6: agricultural soil	Sink (S_3): deposit and concentration	1853	$X_{13} = X_4 + IP_1 + X_{33} + X_{21} - X_{12} - X_{14}$ (balance P_6)		

(Continued)

Flow n.	*P flow description*	*From node*	*To node*	*Value in 1996*	*Equation*	*Coefficient description*	*Coefficient value*
X_{14}	Leaching P	P_6: agricultural soil	Sink (S_2): water environment	164	$X_{14} = K_8 \times 130039.3$	K_8: P nutrient loss coefficient	$K_8 = 12.6$
X_{15}	Harvested crops for food	P_7: harvest distribution	P_8: food industry	1063	$X_{15} = X_{12} + IP_2 - X_{16} - X_{17} - X_{18}$ (balance P_7)		
X_{16}	Harvested crops for fodder	P_7: harvest distribution	P_9: fodder industry	1053	$X_{16} = X_{19} + X_{20}$ (balance P_9)		
X_{17}	Animal-consumed crops/grass	P_7: harvest distribution	P_{10}: animal husbandry	1487	$X_{17} = X_{21} + X_{22} + X_{23} + X_{24} - X_{19}$ (balance P_{10})		
X_{18}	Composted crop stalks	P_7: harvest distribution	P_{13}: manure composting	240	$X_{18} = K_9 \times 1200$	K_9: utilization coefficient of crop stalks	$K_9 = 0.2$
X_{19}	Animal-consumed fodder	P_9: fodder industry	P_{10}: animal husbandry	1032	$X_{19} = 1032$		
X_{20}	Waste from fodder production	P_9: fodder industry	P_{12}: waste treatment	21	$X_{20} = K_{10} \times X_{16}$	K_{10}: nutrient loss coefficient of fodder industry	$K_{10} = 0.02$
X_{21}	Un-composted animal wastes applied for agriculture	P_{10}: animal husbandry	P_6: agricultural soil	1337	$X_{21} = K_{11} \times 1486$	K_{11}: utilization ratio of animal wastes generated by family farms	$K_{11} = 0.9$
X_{22}	Animal products for food	P_{10}: animal husbandry	P_8: food industry	111	$X_{22} = 111$		
X_{23}	Animal manure composting	P_{10}: animal husbandry	P_{13}: manure composting	206	$X_{23} = K_{12} \times 1032$	K_{12}: utilization ratio of scaled breeding animal wastes	$K_{12} = 0.2$
X_{24}	Nutrient loss from animal husbandry into natural soil	P_{10}: animal husbandry	Sink (S_1): natural soil	263	$X_{24} = 2408 - X_{21} - X_{23} - X_{25}$		
X_{25}	Nutrient loss from animal husbandry into water	P_{10}: animal husbandry	Sink (S_2): water environment	602	$X_{25} = K_{13} \times 2408$	K_{13}: nutrient loss coefficient of animal husbandry	$K_{13} = 0.25$
X_{26}	Domestically consumed food	P_8: food industry	P_{11}: food consumption	1080	$X_{26} = X_{29} + X_{30} + X_{33}$ (balance P_{11})		

(Continued)

Flow n.	*P flow description*	*From node*	*To node*	*Value in 1996*	*Equation*	*Coefficient description*	*Coefficient value*
X_{27}	Pollutant load from food industry	P_8: food industry	Sink (S_2): water environment	53	$X_{27} = K_{14} \times X_{15}$	K_{14}: discharge coefficient of food industry	$K_{14} = 0.05$
X_{28}	Waste from food production	P_8: food industry	P_{12}: waste treatment	26	$X_{28} = X_{15} + X_{22} - EP - X_{26} - X_{27}$ (balance P_8)		
X_{29}	Domestic waste from food consumption	P_{11}: food consumption	P_{12}: waste treatment	564	$X_{29} = 147 + 152 + 265$		
X_{30}	Human excreta for manure composting	P_{11}: food consumption	P_{13}: manure composting	458	$X_{30} = 458$		
X_{31}	Waste accumulation in natural soil after treatment or disposal	P_{12}: waste treatment	Sink (S_1): natural soil	612	$X_{31} = X_5 + X_{11} + X_{20} + X_{28} + X_{29}$ (balance P_{12})		
X_{32}	Nutrient loss from manure composting	P_{13}: manure composting	Sink (S_2): water environment	90	$X_{32} = K_{15} \times (X_{18} + X_{23} + X_{30})$	K_{15}: nutrient loss coefficient by composting	$K_{15} = 0.1$
X_{33}	Load from food consumption	P_{11}: food consumption	Sink (S_2): water environment	58	$X_{33} = 44 + 14$		
X_{34}	Recycled nutrient by manure composting	P_{13}: manure composting	P_6: agricultural soil	814	$X_{34} = X_{18} + X_{23} + X_{30} - X_{32}$ (balance P_{13})		
IP_1	Imported P fertilizer	Source (I_1): fertilizer import	P_6: agricultural soil	1129	$FP_1 = K_1 \times 2565$		
IP_2	Import agricultural products for fodder	Source (I_2): agricultural products imported	P_7: harvest distribution	50	$IP_2 = 50$		
EP	Exported agricultural products	P_8: food industry	Sink (E): agricultural products exported	15	$EP = 15$		

Note: WP = washing powder.

Table 3.3 Parameters in the national static SFA model

Parameter	*Description*	*Assumed value*	*Reference*
K_3	Utilized proportion of P ore for fertilizer industry	0.9	Jiang 1999; Cui 2000
K_4	Discharge coefficient of fertilizer industry	0.02*	SEPA 1996
K_5	Discharge coefficient of washing powder production	0.05*	SEPA 1996
K_6	Average P content in washing powder	0.055	Shu et al. 1999; Zhang 1999
K_7	Discharge coefficient of other P chemical industries	0.02*	SEPA 1996
K_8	P nutrient leaching coefficient	12.6	Liu et al. 1995; Cai et al. 2001; Zhang and Qian 1998; Gao and Zhang 1999; Wang et al. 1999; Yan et al. 1999; Chen et al. 2000
K_9	Utilization coefficient of crop stalks	0.2	Huang 1997; Qin 1997; Lu 1997; Niu 2000; Zhou 2001
K_{10}	Nutrient loss coefficient of fodder industry	0.02*	SEPA 1996
K_{11}	Utilization ratio of animal wastes generated by family farms	0.9	Chen 2002; Jia et al. 2002; Shen 2001
K_{12}	Utilization ratio of scaled breeding animal wastes	0.2	Jia et al. 2002; SEPA 2002
K_{13}	Nutrient loss coefficient of animal husbandry	0.25	SEPA 2002
K_{14}	Discharge coefficient of food industry	0.05*	SEPA 1996
K_{15}	Nutrient loss coefficient by composting	0.1**	

* These parameters are assumed as the highest values when data intervals or choices from among multiple data sources exist.

** The value is estimated based on interviews with several officials in the State Environmental Protection Administration (SEPA) and in the Environmental Protection Bureau (EPB) of Kunming municipality, Yunnan Province.

3.3.2 Analysis and identification of the structure of flows

From a perspective of physical accounting, a set of aggregated flows can be calculated in relation to the economy and environmental impacts (see Matthews et al. 2000). The direct material input (DMI), consisting of domestic extraction and imports, indicates the inflow intensity stressing on national economy. In the case of P in China, the DMI was 4203 kilotons (kt)[13] in 1996, a sum of the inflows of RP, IP_1, and IP_2. The domestic P resource extraction (RP) and P chemical fertilizers import (IP_1) contributed 72% and 27% to DMI, respectively; the agricultural commodities import (IP_2) only encompassed a small proportion. The fact that imported P fertilizers formed 31% of the total chemical P fertilizer application implies that P imports contribute significantly to the domestic environmental stress. On the output side, the domestic processed outputs from productive and consumptive activities to domestic environments added up to 2172 kt, of which 1200 kt went to natural soils and 972 kt entered the water environment. The domestic hidden flows, which flows to the domestic environment in the course of providing commodities for economic uses while P does not enter the economy, can also contribute to environmental deterioration. Because of data limitations, only agricultural soil erosion was taken into account as a hidden flow, calculated as 164 kt (X_{14}) in 1996. It is apparent that this amount did not play a major role in the total domestic output. In addition, exports contributed only 15 kt P (EP), which are transferred from the national economy to overseas. In comparison with imports, commodity exports can barely fill the gap in the national phosphorus trade balances. Consequently, there is a net import of P and arguably a net import of environmental problems in China.

Regarding the substance destination and related environmental impacts, the P flows entering agricultural and natural soils and surface waters are given most attention in this study. The total P removed from agricultural soil by harvested crops was 3793 kt (X_{12}) and the total organic and inorganic P nutrient input to soils added up to 5810 kt, a sum of IP_1, X_4, X_{21}, and X_{34}, in 1996. The difference of 2017 kt was balanced by erosion (X_{14}) caused by agricultural runoff from and additional deposit (X_{13}) in the agricultural soils. Although industrial inorganic wastes from the fertilizer industry and other P chemical industries consist of 325 kt ($X_5 + X_{11}$), their contribution to natural soil is relatively small. They can hardly be reused by industries, however, because treatment of and extraction from industrial P wastes, in particular from industrial P gypsum, are usually costly and have low efficiencies (e.g., see the work of Power and Papendick 1992; Sun and Li 1999). Another flow into the natural soil sink is organic wastes generated from fodder industry (X_{20}), food industry (X_{28}), and food consumption (X_{29}) (which refers to excreta and kitchen garbage). The last item is significant because the large amount of organic wastes, particularly in rural areas, have been subject to barely any treatment and are often dumped on uncultivated soils in towns and villages, thus contributing to accumulation of P in natural soil.

The water environment, particularly lakes and reservoirs, has been threatened by annually incremental P discharge from industrial and non-point sources. In comparison with industrial

[13] One kiloton = 106 kilograms ≈ 1,102 short tons. Unless otherwise noted, all tons in this article are metric tons.

P loads, agricultural runoff, animal husbandry, and human livelihoods are the three major contributors to eutrophication, for their P loads account for 90% of the total, as shown in Table 3.4. On a national level, animal husbandry, characterized by large-scale breeding and dispersed (family farms) breeding activities, is the most significant source for water eutrophication in our model. Apart from direct and indirect reuse of P nutrients via manure, a large part of the total output of wastes of livestock and poultry flows into the water environment without any treatment, accounting for 602 kt (X_{25}) and 53% of P flow in the water environment, remarkably higher than the average of European countries (see Table 3.4). Because urban infrastructure has been underdeveloped in China for years, 77% of urban excreta, responsible for 147 kt P of total (X_{29}), was collected and transported out of cities for land-filling and dumping (and eventually transfer into natural soils). The remaining P is discharged into urban sewage systems, which accounts for only a small percentage compared with that for developed countries.

Table 3.4 Phosphorus Sources to Surface Water in China, in thousands of metric tons, 1996

	Industry	*Fertilizer*	*Composting*	*Livestock*	*Detergents*	*Human source*	*Total*
P load (kt)	117	164	90	602	105	58	1136
Percentage (%)	10.3	14.4	7.9	53.0	9.3	5.1	100
Average of EU (14)[1] (%)	7.8	18.9	n.a.[2]	35.6	12.2	25.5	100

Notes: [1] European Union (EU) reference data are cited from Morse et al. (1993), from which Luxemburg was excluded.

[2] n.a.: No available data.

As it has been argued that the use of P-containing detergents has a significant impact on water quality (e.g., see Lee and Jones 1986; Moss 2000), domestic consumption of washing powder is examined in this article. The result shows it contributed 105 kt P to the total water pollution, and is thus responsible for only 9% of the total water pollution by P. This is approximately equal to European countries' experience, if all P-containing compounds were ultimately assumed to go to the water environment. That may imply that on a national level the domestic use of detergents should not be among the top priorities for water quality control, especially because of the high cost and limited effectiveness of P control measures (e.g., Lee and Jones 1986; Litke 1999; Moss 2000). This might be different in certain local areas where massive detergent use contributes considerably to freshwater eutrophication.

Recycling/reuse can be identified in two flows of the static SFA model. The flow from animal production to agricultural soil (X_{21}) refers to the direct reuse of animal wastes by individual farmers in family-level enterprises. Approximately 62% of the total reused P volume follows this flow, mostly without composting. Crop residues treated as by-products of agricultural production, animal wastes from large-scale breeding, and human excreta from rural population together form the second major flow of P that is partly reused (X_{33}). Although the quantities of crop residues generated annually are considered to be massive and the quantities of P-related nutrients account for 1200 kt in 1996, only a small part (indicated by utilization coefficient K_9) is reused on farmland. Organic wastes produced by large-scale breeding livestock and poultry are likewise partially recycled (indicated by K_{12}), for most feedlots have not constructed reuse relations with other agricultural sectors. In contrast, all human excreta from rural population are assumed to be reused entirely by individual farmers after preliminary composting processes.

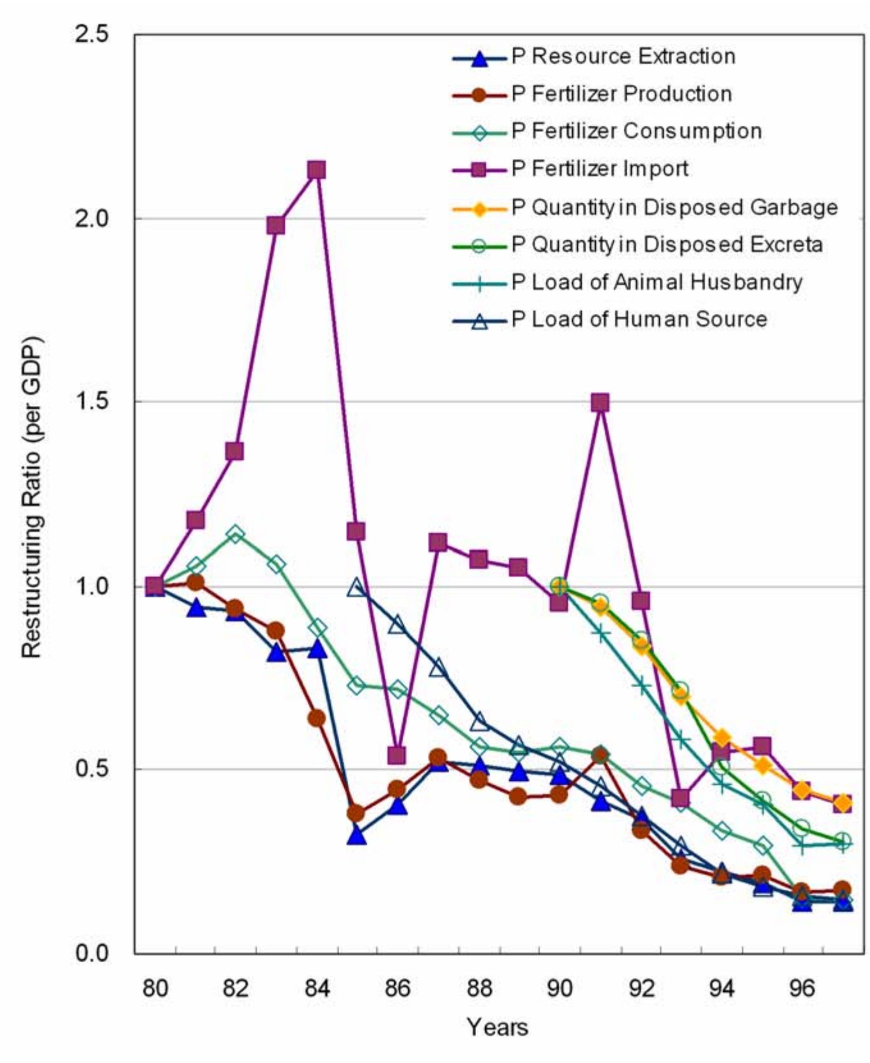
3.4 Ecological Restructuring Analysis

ERIs have been widely used to describe and analyze the development and trends of natural resource use in economies (e.g., Ayres and Simonis 1994; Simonis 1994; Jänicke 1995; Picton and Daniels 1999). Based on the static SFA model described above, a set of ERIs is used and connected to those critical P flows that dominate domestic P throughputs and environmental impacts with Chinese national socio-economic development.

Environmentally-oriented structural change should reconcile economic development and environmental deterioration via (de-)linking[14] of economic growth from material throughputs. As presented in Figure 3.2 and 3.3, although an apparent delinking between economic growth and extraction intensity took place in China, the physical production of P ores increased annually, from 1.4 kg P per capita in 1980 to 2.7 kg P per capita in 1997. With current exploitation intensity and the fact that the economic reserves of P are estimated to be 6950 million tons (Jiang 1999; Cui 2000)[15], P reserves could sustain China's development for approximately 170 years if the current techno-economic condition is maintained. P fertilizer industries use a major proportion of the extracted P reserves, ranging from 61% to 92% and an average of 73% over the years from 1980 to 1997. External P inputs into the domestic economy via fertilizer imports fluctuated annually because of domestic market and global price, but showed a gradual increase in general. The quantity of imports increased form 14% of total chemical P use in 1980 to 38% in 1997, suggesting that China has been dependent on international P production to a significant degree. Subsequently, imports contributed to the significant and enhanced stress on the domestic environment.

[14] A few relationships between material throughput and economic growth are expected to be direct ratio, say, exhibiting linking rather than delinking. For instance, the quantities of treated flows and reuse flows, indicating the capacities to what extent an industrialized economy can clean up or recycle wastes, should increase along with or even beyond economic growth levels.

[15] China has adopted and still uses the "socialist" method for P resource classification, copied from Soviet Russia in the 1950s. This method categorizes mining resources into four classes: producible reserves, stock reserves, industrial reserves, and prospective reserves (commonly denoted by alphabetic characters A, B, C, and D, respectively, in technical literatures). Economic reserves refer to the sum of producible reserves (A), stock reserves (B), and industrial reserves (C), that is, A + B + C. This method hardly takes into consideration economic efficiency and technological feasibility for the mining industry.



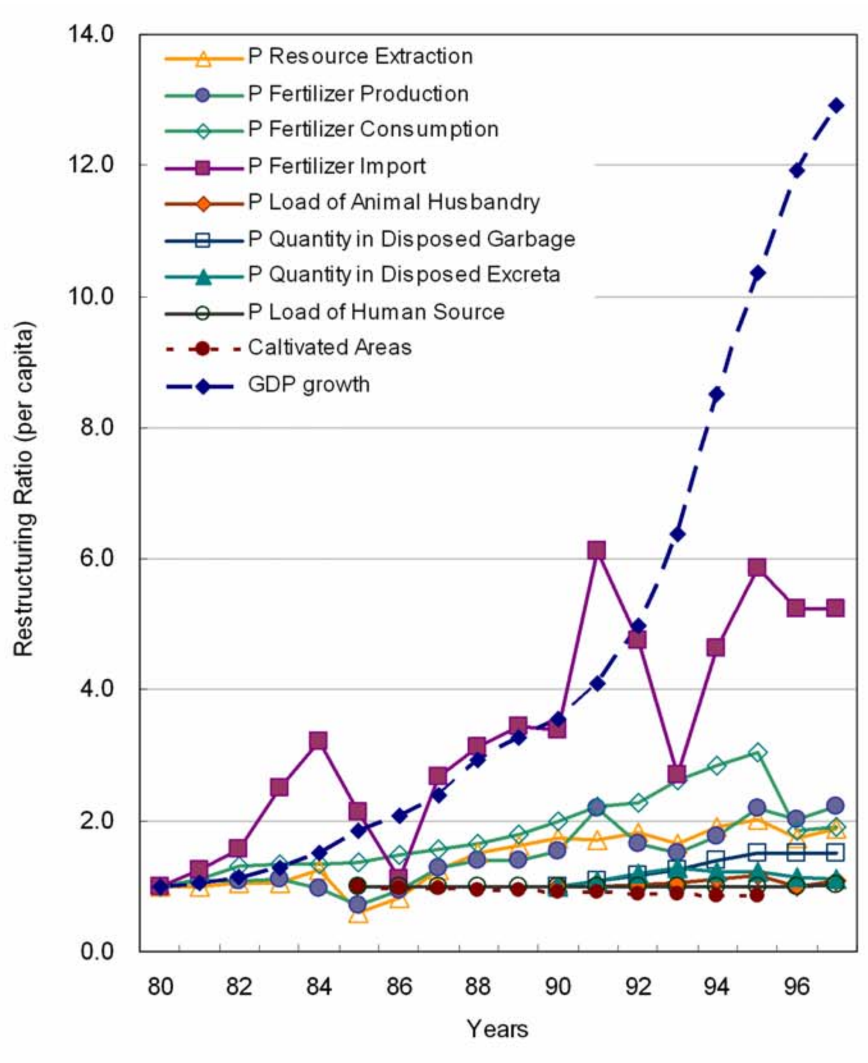


Figure 3.2 Ecological restructuring: national P flows per unit GDP, 1980-1997

Figure 3.3 Ecological restructuring: national P flows per capita, 1980-1997

Notes: Except for the P loads of human and animal sources, the other variables show a radical decline of absolute quantities between 1995 and 1996. This is caused by inconsistencies in the data series of China's statistical yearbooks

P consumption on farmland per capita showed an incremental though considerable increase during the last two decades, but decreased dramatically normalized by GDP (see Figures 3.2 and 3.3). With the slight decline of cultivated area in the same period, the rising chemical P application intensity, from 13 kg P/ha[16] in 1980 to 50 kg P/ha in 1995, undoubtedly contributed to agricultural production. Grain output in China has substantially increased and reached 494 million tons in 1997, about one and a half times more than the output in 1980. But it is also notable that the P use efficiency was correspondingly reduced to 99 t grain/t P, only 40% of the level in 1980. The high intensity and relative low utilization efficiency implies an increasing impact on water environment via fertilizer runoff.

As identified in the static SFA model, animal husbandry was the major water pollution source as well as potential source for organic P nutrient reuse. Because the export of agricultural products, including meets and eggs, has only a slight influence on the P metabolic structure in our model, the P load of livestock and poultry was dominated by domestic consumption. Although the consumed quantity of P in livestock products reached 75 g/yr per capita in 1997, 59% more than that in 1990, the total P load per capita only increased 8% in parallel, much slower than economic growth. This slow increase occurred because the rapid diet change, mostly driven by increasing domestic demand for meet, eggs and milk (see Figure 3.4), was not caused by an increase in the number of livestock and poultry, but by structural change in the pattern of animal husbandry. It is evident that, in comparison with breeding by family-level enterprises, intensive-scale breeding, responsible for commercialized livestock products, rapidly developed in the 1990s in China (SEPA 2002). The increasing supply of livestock products, consequently, had considerably influence on domestic grain production (e.g., see Cheng et al. 1997; Heilig 1999), which in turn contributed to the increase of P fertilizer use and its negative environmental results. Furthermore, the low efficiency of manure reuse from large-scale stables and feedlots, as indicated by K_{12} in the SFA model, strongly suggested more water environmental stress along with the rapid shifts in breeding pattern.

The total output of P load generated from anthropogenic sources increased gradually in conjunction with population growth and ongoing urbanization. The latter, rising from 19% in 1980 to 30% in 1997, was more significant for the local environment as urban residents consumed more livestock products. But the related waste P flows barely returned to farmlands because of much higher collection and transportation costs. Urban P load from human sources reached 366 kt in 1997, double the amount in 1980. Municipal sanitation systems were responsible for 304 kt P removed from urban areas in 1997, which accounted for 83% of the total and which was 14% higher than in 1980. With respect to the fact that most solid wastes have been subject to simple storage and dumping for a long time, the increase of urban waste P flows and linkage of the increase in P flows to economic growth, as shown in Figure 3.2 and 3.3, is undesirable because it implies that rapid economic growth did not promote and contribute to the further development of urban waste treatment and nutrient recycling. This in turn leads to an urban environmental crisis in developing China. At the same time, the P

[16] One hectare (ha) = 10,000 m2 ≈ 2.47 acres. One kilogram/hectare ≈ 89 pounds/acre.

volume of disposed urban excreta has declined since 1993, as extensive construction of urban infrastructure in China has been promoted by governments since then. Although the development of centralized sewage systems changes the P flow structure with respect to human sources, it can hardly remedy eutrophication and contribute to nutrient recycling (Liu 2004)[17]. Although technical gaps, higher costs and ecological risks impede the reuse of nutrients from domestic wastewater treatment sludge (e.g., Power and Papendick 1992; IWAG 2001), the manure from dispersed rural human sources, in comparison with collective treatment, has a rather large potential for reuse, which is considered to be crucial for building closed-loop systems.

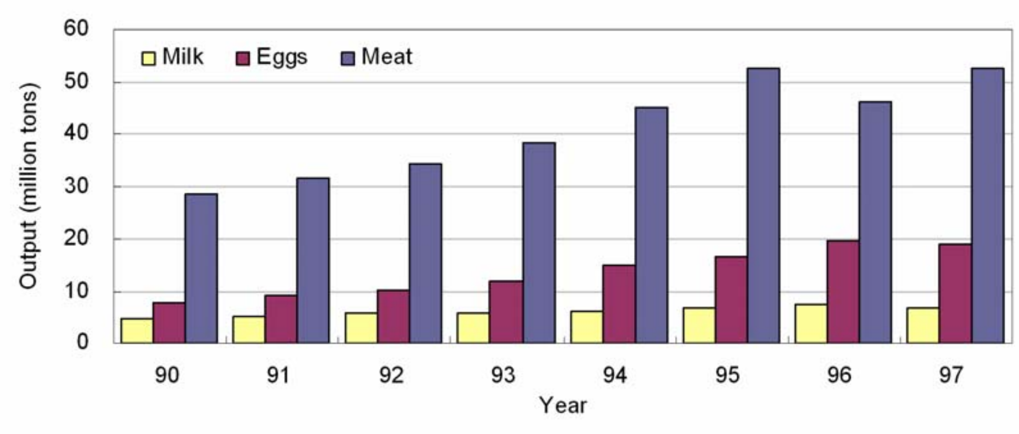


Figure 3.4 Output of livestock products in China, in millions of tons 1990-1997

Sources: China Statistical Yearbook 1997-1998

In general, the (de-)linking between socio-economic growth and relevant P flows has been paralleled by an annual increase in absolute quantities, particularly of P fertilizer importation, production and consumption. Natural soil and water environments continue to deteriorate because of extensive pollution by gradual increasing P loads from diverse sources.

3.5 Discussion and Conclusions

This article applied an SFA approach to reveal the national P regime of China and subsequently to combine the critical P flows with the developments in the economy. The results show that SFA can easily handle very large systems with the basic Law of Mass Conservation, as the structure of physical flows through economic processes can be elucidated, and the main sources of resource depletion and environmental deterioration can be traced and linked to their economic origins. Although the static SFA model was primarily developed for depicting the P-related overall metabolic structure in China by quantitatively balancing and estimating unknown but critical flows, it can also be used for decision making and policy

[17] Some alternatives do exist in terms of P recovery. In a few European countries, a limited number of wastewater treatment plants have started to recover P from sludge since the late 1990s (e.g., www.nhm.ac.uk/mineralogy/-phos/). Cost-effective techniques, however, are not yet widely available.

design by mathematical simulation by evaluating parameters (e.g., discharge coefficients and utilization ratios in Table 3.3).

The combination of critical P flows with economic issues offers possibilities for the identification and examination of potential solutions with respect to both environmental effectiveness and economic efficiency. The general conclusion is that over the last two decades China has become more efficient in the use of P materials. Despite strong economic growth, per capita resource inputs and waste outputs of P increased relatively little since 1980, and when measured against units of economic output, resource inputs and waste outputs fell dramatically. In all absolute terms – which are especially relevant for environmental quality – P flows have grown, and, more importantly, would be more of significance along with predictable development of animal husbandry and urbanization (e.g., see World Bank 2001; UNEP 2002).

With a better understanding of how industrializing China consumes and processes P material and with a clearer sense of how the structure of the P "economy" changes over time, a "rational" design of (policy) measures is within reach, which moves the national P flow system in more economically efficient and environmentally effective directions. Such measures should not only take industrial pollution into account, particularly P resource mining and P chemical industries, but also contribute to agricultural innovation by a considerable reduction of chemical fertilizer consumption. Maximizing the reuse of manure in farming and minimizing P fertilizer imports will lead to environmental improvements. Secondly, recycling of P nutrients from animal husbandry and human sources should be given close attention. The ecologizing of the agricultural chain by connecting animal husbandry with crop production should be introduced widely within the Chinese economy by effective economic incentives and enforcement of environmental regulation. In addition, decentralization of the collective system of urban wastewater treatment probably provides a new agenda for developing China, as it facilitates, for example, the recycling of nutrients from human sources by separating excreta and urine at the source (Otterpohl et al. 1999; Larsen et al. 2001), and consequently reducing the enormous investment in new central municipal infrastructure. Last but not least, all these measures should consider the physical system as an integrated and interrelated one, because the material flows can never be isolated from economy and society. Independent flows (say, the exogenously fixed variables in our model) only exist in mathematical terms. Whereas some instruments are invented and regarded as effective for ecologizing some critical flows, decision makers as well as scientists ought to look at their potential impacts on other flows within the entire economy.

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References

Ayres, R.U., and U.E. Simonis. 1994. *Industrial Metabolism: Restructuring for Sustainable Development*. Tokyo/New York/Paris: United Nations University Press.

Bouman, M., R. Heijung, E. van der Voet, Van den Bergh, C. J. M. Jeroen and G. Huppes. 2000. Material flows and economic models: an analytical comparison of SFA, LCA and partial equilibrium models. *Ecological Economics* 32: 195-216.

Bringezu, S., and H. Schütz, 2001. *Total material requirement of the European Union*. Technical Report No. 26. Copenhagen: European Environmental Agency.

Cai, C. F., S. W. Ding, Z. H. Shi, L. Huang, and G. Y. Zhang. 2001. Prediction on soil nutrients losses at typical small watershed of Three Gorges Area with GIS. *Journal of Soil and Water Conservation* 15(1): 9-12. (in Chinese).

Chen, F. 2002. *Agricultural Ecology*. Beijing: China Agriculture University Press. (in Chinese).

Chen, X., Z. Q. Wang, W. D. Yang, and X. J. Ye. 2000. Effects of sloping land use patterns on phosphorus loss in a micro-watershed of red soil area, Southern China. *Acta Ecological Sinica* 20(3): 374-377. (in Chinese).

Cheng, G. Q., Y. H. Zhou, J. M. Wang and Z. L. Shi. 1997. Estimate on China’s fodder supply and demand. *Agricultural Economic Issue* 5: 25-29. (in Chinese).

Cui, Y. T. 2000. *Fertilizers and Ecological Environmental Protection*. Beijing: Chemical Industry Press. (in Chinese).

Daniels, P.L. 2002. Approaches for quantifying the metabolism of physical economies: A comparative survey, part II: Review of individual approaches. *Journal of Industrial Ecology* 6(1): 65-88.

Daniels, P. L., and S. Moore. 2001. Approaches for quantifying the metabolism of physical economies – part I: methodological overview. *Journal of Industrial Ecology* 5(4): 69-93.

Dokulil, M., W. CHEN, and Q. CAI. 2000. Anthropogenic impacts to large lakes in China: the Tai Hu example. *Aquatic Ecosystem Health and Management* 3: 81-94.

Dryzek, J. S. 1987. *Rational ecology. Environment and political economy*. Oxford, UK: Basil Blackwell.

Foster, J. B. 1999. Marx's theory of metabolic rift: Classical foundations for environmental sociology. *American Journal of Sociology* 105 (2): 366-405.

Frosch, R.A., and N.E. Gallopoulos. 1989. Strategies for manufacturing. *Scientific American* 261(3): 144-152.

Gao, C., and T. L. Zhang. 1999. Contribution of agricultural phosphorus losses to eutrophication of waters and its controlling strategies. *Journal of Lake Sciences* 11(4): 369-375. (in Chinese).

Guinee, J. B., J. C. J. M. van den Bergh, J. Boelens, *et al.* 1999. Evaluation of risks of metal flows and accumulation in economy and environment. *Ecological Economics* 30 47-65.

Han X. G., L. H. Li, and J. H. Huang. 1999. *An Introduction to Biogeochemistry*. Beijing: Higher Education Press; German: Springer-Verlag. (in Chinese).

Heilig, G. K. 1999. *Can China Feed Itself? A System for Evaluation of Policy Options*. Laxenburg, Austria: IIASA. CD-ROM, Version 1.1.

Huang, B. F. 1997. Estimates on the supply and demand of fodder in China in 2000. *China Grains Economy* 4: 10-13. (in Chinese).

IWAG. 2001. Sewage Sludge Disposal: Sustainable and/or Reliable Solutions. ÖWAV / EWA Workshop, 10-11 Sep, TU-Wien, Vienna.

Jänicke, M. 1995. Green industrial policy and the future of “dirty industries”. Unpublished paper, Free University Berlin.

Jia, X. H., Y. F. Huang, and J. T. Xu. 2002. Manure Process and Utilization. Beijing: Chemical Industry Press. (in Chinese).

Jiang, S. X. 1999. *Phosphoric Acid, Phosphorous Fertilizer and Compound Fertilizer*. Beijing: Chemical Industry Press. (in Chinese).

Jin X. C. 1995. *Lakes in China – Research of Their Environment, Vols. I and II*. Beijing: China Ocean Press,. (in Chinese).

Kleijn, R. 2000. In=Out: the trivial central paradigm of MFA? *Journal of Industrial Ecology* 3(2&3): 8-10.

Kleijn, R., R. Huele, and E. van der Voet. 2000. Dynamic substance flow analysis: the delaying mechanism of stocks, with the case of PVC in Sweden. *Ecological Economics* 32: 241-254.

Larsen, T. A., I. Peters, A. Alder, R. Eggen, M. Maurer, and J. Muncke. 2001. Re-engineering the toilet for sustainable wastewater management. *Environmental Science & Technology* 35(9): 192A–197A.

Lee, G.F., and R.A. Jones. 1986. Detergent phosphate bans and eutrophication. *Environmental Science and Technology* 20(4): 330-331.

Litke, D.W. 1999. *Review of phosphorus control measures in the United States and their effects on water quality*. Water-Resources Investigations Report 99-4007. Denver, CO: U.S. Geological Survey.

Liu, B. Z., G. L. Li, F. Q. Wu, and X. G. Zhao. 1995. The regular patterns of the loss of soil nutrients on Southern Loess Plateau. *Journal of Soil and Water Conservation* 9(2): 77-86. (in Chinese).

Liu, Y. 2004. Study on phosphorus societal metabolism and eutrophication control policy in China. Ph.D. thesis, Tsinghua University, Beijing, People's Republic of China.

Liu, Y., J. Chen, and A. P. J. Mol. 2004. Evaluation of phosphorus flows in the Dianchi Watershed, southwest of China. *Population and Environment* 25(6): 637-656.

Lu, W. G. 1997. Estimates on grain consumption for feed in China. *China Grain Economics* (3): 38-40. (in Chinese).

Matthews, E., C. Amann, S. Bringezu, *et al.* 2000. *The Weight of Nations: Material Outflows from Industrial Economies*. Washington, D.C.: World Resources Institute.

Mol, A. P. J. 1995. *The refinement of production. Ecological modernization theory and the chemical industry*. Utrecht, The Netherlands: International Books.

Mol, A.P.J. 2003. Political modernisation and environmental governance: between delinking and linking, *EUROPAEA, Journal of Europeanists* 8 (1-2):.169-185.

Moolenaar, S. W., E.J.M. Temminghoff, and F.A.M. De Haan. 1998. Modeling dynamic copper balances for a contaminated sandy soil following land use change from agriculture to forestry. *Environmental Pollution* 103:117-125.

Morse, G.K., J.N. Lester, and R. Perry. 1993. The Economic and Environment Impact of Phosphorus Removal from Wastewater in the European Community. Centre Europeen D'Etudes des Polyphosphates E.V. London: Selper Publications.

Moss, B. 2000. Phosphorus, detergents and eutrophication control - A summary of experience in Europe and North America. *Detergent and Cosmetics* 23 (2, Supplement, July): 117-129.

Niu, B. J. 2000. The impact on and strategy of China's fodder industry after entering WTO. *Journal of Huazhong Agricultural University* (*Social Sciences Edition*) 3: 15-17. (in Chinese).

Otterpohl, R., A. Albold, and M. Oldenburg. 1999. Source control in urban sanitation and waste management: Ten systems with reuse of resources. *Water Science and Technology* 39(5): 153–160.

Picton, T., and P. L. Daniels. 1999. Ecological restructuring for sustainable development: evidence from the Australian economy. *Ecological Economics* 29(3): 405-425.

Power, J. F., and R. I. Papendick. 1992. Organic sources of nutrients, in *Fertilizer Technology and Use* (3rd edition), Engelstad O. P. edited. Translated by Zhang Guo-Xin *et al.* Beijing: Chemical Industry Press.

Prosterman, R., T. Hanstad, and L. Ping. 1996. Can China Feed Itself? *Scientific American* November: 70-76.

Qin, F. 1997. Discussion on market demand of feed in China. *Journal of Agrotechnical Economics* (4): 41-45. (in Chinese).

SEPA. 1996. *Handbook for Industrial Pollutants Production and Discharge and Emission Coefficients*. Beijing: China Environmental Science Press.

SEPA. 2002. *Investigation on the Pollution of National Livestock and Poultry Breeding in Scale and Its Control Strategy*. Beijing: China Environmental Science Press. (in Chinese).

Shen, Q. R., ed. 2001. *General Introduction on Soil and Fertilizer*. Beijing: High Education Press. (in Chinese).

Shu, J. H., W. Y. Huang, and X. Y. Gao. 1999. The evaluation of the effect of powdered detergent containing phosphate on Tai Hu Water Area Eutrophication. *Detergent and Cosmetics* 22(August Supplement): 129-145. (in Chinese).

Si, Y. B., S. Q. Wang, and H. M. Chen. 2000. Nitrogen and phosphorus losses from agricultural soil and surface water eutrophication. *Journal of Soil* (4): 188-193. (in Chinese).

Simonis, U.E. 1994. Industrial restructuring in industrial countries. In *Industrial Metabolism: Restructuring for Sustainable Development*, edited by R.U. Ayres and U.E. Simonis. Tokyo: United Nations University Press.

Smil, V. 1995. Who Will Feed China? *The China Quarterly* 143: 801-813.

Smil, V. 2000. Phosphorus in the environment: Natural flows and human interferences. *Annual Review of Energy and Environment* 25: 53-88.

Stauffer, J. 1998. *The Water Crisis: Constructing Solutions to Freshwater Pollution*. London: Earthscan.

Steen, I. 1998. Phosphorus availability in the 21st century: Management of a non-renewable resource. *Phosphorus and Potassium* 217(Sep/Oct): 25-31.

Sun, F. G., and Z. J. Li. 1999. Recent developments and prospects for fertilizer in China. In proceedings of 67th IFA Annual Conference, Manila, the Philippines, 17-20 May.

UNEP. 2002. *China Human Development Report 2002: Making Green Development a Choice*. Oxford UK: Oxford University Press.

van der Voet, E. 1996. Substances from Cradle to Grave: Development of a methodology for the analysis of substance flows through the economy and the environment of a region. Ph.D. Thesis, Centre of Environmental Science, Leiden University, Leiden, the Netherlands.

van der Voet, E., J. B. Guinee, and H. A. Udo de Haes. 1999. Metals in the Netherlands: a problem solved? *CML-SSP Working Paper* 99.003, the Centre of Environmental Science, Leiden University, the Netherlands.

Wang, X. X., T. L. Zhang, and B. Zhang. 1999. Nutrient cycling and balance of sloping upland ecosystems on red soil. *Acta Ecological Sinica* 19(3): 335-341. (in Chinese).

World Bank. 2001. *China: Air, Land, and Water*. Washington, DC: World Bank.

Yan, W. J., C. Q. Yin, P. Sun, X. Y. Han, and S. X. Xia. 1999. Phosphorus and nitrogen transfers and runoff losses from rice field wetlands of Chaohu Lake. *Chinese Journal of Applied Ecology* 10(3): 312-316. (in Chinese).

Zhang, X. D. 1999. The development and prospect of China powder detergent market. *Detergent and Cosmetics* 5: 17-20.

Zhang, Z. X., and Y. Qian, eds. 1998. *Urban Sustainable Development and Water Pollution Control Strategy*. Beijing: China Construction Industry Press. (in Chinese).

Zhou, X. Y. 2001. Demand of feed grains and adjustment of productive distribution in China. *Chinese Journal of Animal Science* 37 (2): 45-46. (in Chinese).

CHAPTER 4

Evaluation of Phosphorus Flows in the Dianchi Watershed, Southwest of China[18]

Abstract

Environmental problems are closely related to human societies' processing of materials. Along with a rapid economic growth, lake eutrophication has received broad attentions in the last two decades in China. As phosphorus is the dominant nutrient in lake environment, this study attempts to examine how human societies extract, utilize and release phosphorus which subsequently leads to eutrophication. Applying substance flow analysis (SFA) approach for the case of Dianchi Lake in southern China, this study establishes a statistical model by balancing the physical quantities of phosphorus flows in 2000. Resource extraction, phosphate industries, agricultural production, livestock breeding, and human living are given focused attentions in the SFA model. The results showed water environment is largely dependent on the local phosphorus metabolism, and thus, if the related phosphorus flows, particularly referring to recycling and wastes flows, could not be re-organized towards a more ecological direction, Dianchi Lake would be unavoidably and continuously suffering eutrophication. Aiming at ecological restructuring of the phosphorus metabolism with respect to economic efficiency and environmental effectiveness, critical options are discussed and future directions in relation to both societal and policy aspects are presented.

Keywords

Phosphorus; Metabolism; SFA; Dianchi Lake; China

4.1 Introduction

The industrialization of human societies is closely associated with material processing. In this context, an economic system functions in many aspects like an eco-system: it "ingests" raw materials, which are "metabolized" to produce goods and services, and then "excrete" wastes and pollution (Matthews et al. 2000). Different from the natural processes, however, both the intensity and velocity of material metabolism within modern economies are much more significant (Ayres 1994). It is the material metabolism, through resources extraction and pollution emission, that builds up the link between economy and nature, and consequently often seriously distorts the natural eco-system (Hannah et al. 1994; Wen 1998). There is thus

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no doubt that many of the environmental problems we are facing today are a result of society's processing of materials through the economical activities (Bouman et al. 2000; Kleijn, Huele, and van der Voet 2000).

Since China began to accelerate her economic growth in the late 1970s, both the total direct material input and material consumption intensity have increased steadily. Chen and Qiao (2001), for instance, concluded the total material requirement (TMR) and domestic material input (DMI) of China in 1996 were 1.5 and 1.4 times of those in 1990, respectively. Among the others, mining and excavation for infrastructure, fossil fuels and their hidden flows, soil erosion, metals and their hidden flows, and industrial minerals and their hidden flows were recognized to be the principal contributors to TMR[19]. Furthermore, the authors revealed China's GNP output per unit TMR were rather lower than USA, Japan, Germany and the Netherlands. Recently, other researchers examined the ecological efficiencies of material use at local level (Chen and Zhang 2003; Peng and Hou 2000; Zhang, Shi, and Liu 2003). The results showed that the material inputs into local economies had steadily increased and there is no apparent delinking trends between economic growth and material use occurred during the last a couple of years, wherever in relative rich areas, including Guangdong Province, Jiangsu Province and Shanghai Municipality, or less developing areas, i.e., Guiyang Municipality, in China. Intensive material use and low ecological efficiency lead thus to a substantial emission of pollution loads. Consequently, China, as one of the most rapid growing industrializing economies of the world in the last two decades, has to face serious environmental issues and ecological risks.

Lake eutrophication, one of the major environmental problems caused by local human society's intensified processing and thus strengthened discharge of nutrient and organic substances, has been widely reported in China since the 1980s (Dokulil, Chen, and Cai 2000; Jin 1995; Jin 2001; Si, Wang, and Chen 2000; Yang and Ren 2000). For instance, a study on 31 fresh-water lakes including the largest top 10 in China, based on morning data in 1992 and 1995, concluded 64.5 percent (twenty lakes) of total was hypertrophic, 16.1 percent (five lakes) was categorized as mesotrophic and the rest (six lakes) as eutrophic (Dokulil, Chen, and Cai 2000). Jin (2001) examined eutrophication degree of the five largest fresh-water lakes, fourteen medium lakes and twelve urban lakes in China, and he then came to a conclusion that lake eutrophication has been the most leading environmental problem for China's lakes.

Phosphate overload has been recognized as one of the key limiting factors for leading to eutrophication in most cases (Jin 2001; Moss 2000; Shu et al. 1999), since phosphorus is, on average, the element proportionately scarcest in relation to aquatic organism's need (Stauffer 1998; Dokulil, Chen, and Cai 2000). In contrast to the natural environment where phosphorus

[19] Hidden flows occur at the harvesting or extraction stage of a material cycle. They comprise two components: ancillary flows (for example, plant and forest biomass that is removed from lands along with logs and grain, but is later separated from the desired material before further processing), and excavated and/or disturbed material flows (for example, material that must be removed to get access to a valuable ore, and soil erosion that results from agriculture). It is argued that the size of national hidden flows is closely linked to the presence or absence of a mining sector, the country's geographic scale (which determines the size of infrastructure), and the scale and type of agriculture (which influences soil erosion). There is now increasing recognition of importance of hidden flows in a couple of national environmental-economic accountings. For rich and sophisticated discussions, see Matthews et al. (2000).

is supplied through the weathering and dissolution of rocks and minerals with very low solubility (Steen 1998), however, human activity has significantly shifted the natural phosphorus metabolism, particularly after modern fertilizer industry and phosphate industry was introduced and abroad adopted after the Second World War (Stauffer 1998; Foster 1999; Han, Li, and Huang 1999; Liu, Mol, and Chen 2004).

Phosphorus metabolism within and between economy and nature has been accelerated (Smil 2000). In most countries, including China, most phosphate minerals are processed to fill up the ever-increasing needs of human subsistence, and the rest, about 15 percent, is applied as additives or activators for producing metalwork, leather, dyes, glass, ceramics, medicaments, detergents, etc. (Chen and Tan 1989; Jiang 1999; Steen 1998). As population growth and then increases in production and exports of agricultural produce, more and more external (chemical) phosphate nutrient input to improve and maintain soil fertility has been required (Driver 1998; Steen 1998). Moreover, the transition from traditional agriculture to modern intensive agriculture and the development of urbanization significantly prevent the return to the soil of the constituent element consumed by man and animals in the form of food (and clothing) and thus lead to environmental pollution (Stauffer 1998; Foster 1999). Consequently, traditional metabolic relation between human beings and nature was broken and the industrialized pattern of phosphate metabolism characterized by rapacious extraction of phosphate resource and incontinent discharge of pollution is considerably reinforced.

Towards future elaboration of ecological irrationality of the modern phosphorus metabolism, this study attempts to depict how human society uses phosphorus and thus how environmental problem it causes quantitatively. Different from monetary accounting, substance flow analysis approach is applied to frame a statistical phosphorus flow model in 2000 in the case of Dianchi Watershed, in southwestern China. This paper begins with a brief review of the analytical method itself. After an introduction of the case, the SFA model is built up and discussed in detail. It closes by conclusions drew up from the model and a discussion towards ecologicalizing the phosphorus metabolism.

4.2 The MFA/SFA Method and Phosphorus Metabolism

As traditional monetary accounting cannot generate a complete picture of the economy, physical approaches have been widely applied as a useful complement. Material flow analysis (MFA) or substance flow analysis (SFA), originating from a physical analysis in the context of industrial metabolism (Frosch and Gallopoulos 1989), is an analytical tool for describing the material regime of economy based on the Law of Mass Conservation (Kleijn, Huele, and van der Voet 2000). Mostly, MFA/SFA is applied to elaborate how economies utilize natural resources, by tracking materials from its origin, through different processing steps, to its final destination, with respect to their productive efficiency and the resulted environmental stress (Bouman et al. 2000; Daniels and Moore 2001; Guinee et al. 1999; Kleijn, Huele, and van der Voet 2000; Liu, Mol, and Chen 2004; Matthews et al. 2000; Moolenaar, Temminghoff, and Haan 1998; Van der Voet 1996). Through revealing the physical regime of economical activities, MFA/SFA provides an innovative way for identifying and assessing potential solutions to industrial related environmental problems, and thus it is recognized as an effective tool for physical accounting of national and regional economy.

Since the 1990s, a number of MFA/SFA approaches have been widely applied for three major purposes as given below, i.e.,

(i) to compare the eco-efficiency and examine the eco-restructuring trajectories among nations by accounting input/output intensities of total materials or a specific substance (e.g., see (Janicke 1995); (Adriaanse et al. 1997; Amann et al. 2002; Bartelmus and Vesper 2000; Bringezu 2002; Chen and Qiao 2001; Matthews et al. 2000; Peng and Hou 2000);

(ii) to distinguish the dematerialization potential and policy alternatives by elaborating the social-economic metabolism of a specific substance at a national or regional level (e.g., see) (Jolly 1992) (Daniels and Moore 2001; Fenton 2001; Guinee et al. 1999; Kelly 1998; Kleijn, Tukker, and van der Voet 1997; Kramer 2002; Liu, Mol, and Chen 2004; Llewellyn 1994; Loebenstein 1994; Moolenaar, Temminghoff, and Haan 1998; Ober 2002; Picton and Daniels 1999; Smith 2002; Sznopek and Goonan 2000; Van der Voet, Guinee, and Udo de Haes 1999; Wagner 2002); and

(iii) to optimize industrial distribution and structure towards establishing local industrial "symbiosis" systems or eco-industrial parks by tracking the lifespan of total materials or elements, and further reducing material uses and promoting wastes reuse and recycle within and among enterprises (e.g., see) (Ayres 1994); (Chen and Zhang 2003; Fleig 2000; Fujie and Goto 2002; Posch 2002; Zhang, Shi, and Liu 2003).

All these studies offer advanced experiences and a variety of analytical skills in the regime of physical accounting. As the studies in cluster (i) applied the indicator of ecological efficiency for weighed national economy, however, they barely referred to how independent economic entity ties to the use of total material or selected substances in relation with resources depletion and ecological issues. Moreover, despite a number of specific materials and substances, including some metals, chlorine, cement, sulfur, and several potential hazardous wastes, e.g., arsenic and mercury, were illuminated at either a global or national level in the studies in both cluster (ii) and (iii), most of them were still confined to the realm of industrial metabolism advocated by Ayres, as they hardly presented an insight into modern intensive agricultural production and its negative influences on the nutrient metabolic rift between human society and nature as well as the related environmental pollution, i.e., lake eutrophication.

Phosphorus is a dominant element leading to surface water eutrophication and has attracted a wide attention in China since the 1990s. Considering the dominate role of fertilizers in total phosphate consumption by human societies, different from other industrial substances, it is the agriculture, driven by population growth and ever-increasing food requirement, overwhelmingly determines the utilized intensities and forms of phosphate resource. Within the phosphorus metabolism, the predominant utilization of raw phosphate minerals is refined for the production of chemical fertilizers and then applied to farmlands, from which phosphorus release is closely related to agricultural and living activities. Through the food chain, for instance, phosphorus is transferred from harvested crops, partly through livestock, and ingested and discharged by human beings, to the environment. Accordingly, the phosphorus bio-metabolism by human and livestock are two major sources responsible for local environment. Phosphorus flows with respect to agricultural production, livestock husbandry and people's daily life are thereafter the dominant three aspects in terms of lake eutrophication control in China.

4.3 The Case of Dianchi Watershed

The Dianchi Lake is a senile and shallow plateau lake, located in the middle part of Yungui plateau, southwest of China. Covering a total basin area of 2920 km^2, the lake is regarded as the 6th largest freshwater lake in China. The water surface area of Dianchi Lake is approximately 300 km^2, accounting for 10.3 percent of the basin area, with a cubic content of 1.29 billion m^3 (YPPG 1998). It resembles a *bow* with a length of 45 km along north-south direction and a maximum width of 12.5 km along the other, as shown in Figure 4.1. A constructed lake bank separates Dianchi Lake into two parts, i.e., the Caohai, a relative small Inner Lake in north with a water area of 11 km^2 and a depth of 1-2 m, and the Waihai, a much bigger one in south with a water area of 295 km^2 and a depth of 3-5 m (Lu 1997).

Dianchi Watershed holds quite rich phosphorous resource. Up to 1990, the explored phosphorite reserve summed up to 3.58 billion tons and accounted for 41.2 percent of that of Yunnan province and 17.6 percent of total national reserve (Huang 2001; Wang 1998). Among the total, 0.44 billion tons was categorized as *rich* ores, referring to the related P_2O_5 content is more than 30 percent, which took 46.0 percent of the total rich ores of China, and 1.31 billion tons was regarded as strip-minable ores which is characterized by simple-constitution and is deposited above on the base level of erosion (Luo 1997; Wang 1998).

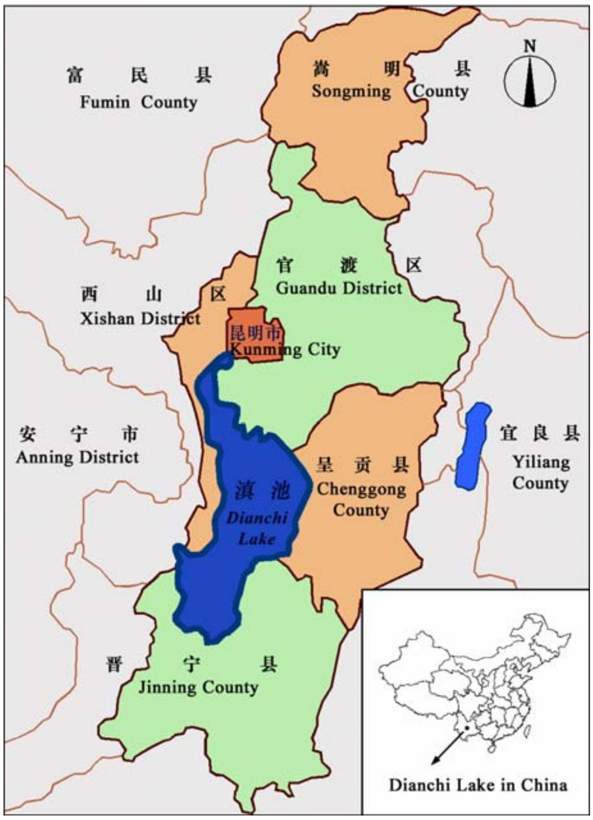

Figure 4.1 Geographic location of Dianchi Lake in China and Yunnan province

Dianchi Lake basin is a densely-populated area with a relative developed economy. The basin includes Wuhua and Panlong districts of Kunming City and 38 villages and towns from 5 suburban areas, i.e., Xishan District, Guandu District, Chenggong County, Ninjing County and Songming County, with a total population of 308 million people in 2001, of which 61.3 per cent lived in cities and towns, accounting for 52.7 percent of that in the whole basin (DESE, KIES, and KWHIDI 2003). While the basin area only takes 13.8 percent of Kunming prefecture, the GDP contributed to 83.0 per cent of that of the whole municipality and was responsible for 26.9 percent of total provincial output in 2001 (KSB 2002; DESE, KIES, and KWHIDI 2003).

The deterioration of water quality in the Dianchi Lake began in the late 1970s and has been rapidly accelerated in the 1990s. Figure 4.2 shows the annual average total phosphorus concentration (TPC) in Caohai and Waihai over the last ten years. Different from the situation in Waihai where the TPC has steadily increased, the TPC in Caohai is high yet largely fluctuated in recent years. The reason for that is that many measures have been implemented to control the phosphorus discharge loads to the Caohai, including the cleanup of contaminated sediments and the construction of wastewater treatment plants.

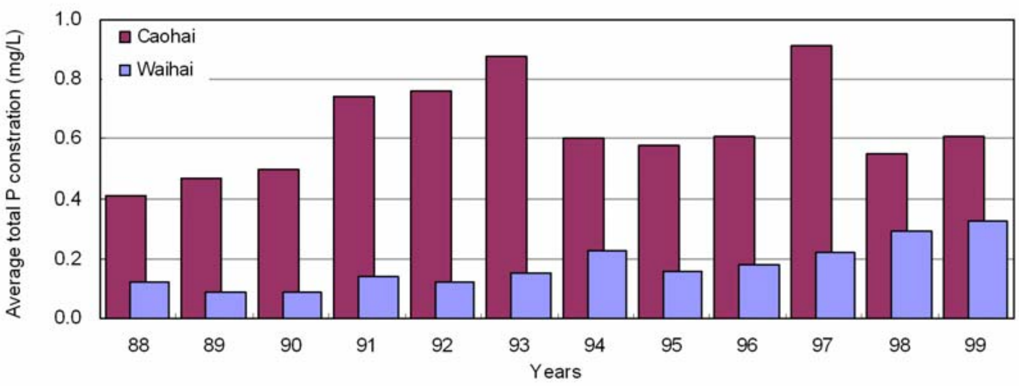

Figure 4.2 The average TPC (mg/L) in Caohai and Waihai of Dianchi Lake

Sources: based on Liu (2001).

Broadly, the discharge of phosphorus can be attributed to three pathways, based on the phosphorus metabolism within local economic society. As phosphate minerals is first of all subject to extraction and refinement, the industrial sector is regarded as one of major pollution sources. Along with application of chemical fertilizer as well as manure to some degree for food production, nutrient loses from agricultural lands and animal husbandry is thus considered as the second source of phosphate load which is closely related to rural farming activities. Finally, human beings ingest phosphate nutrients from vegetal food and animal meat and excrete through bio-metabolism, which partially contributes to phosphate accumulation in local environment.

It is evidenced that the eutrophication of Dianchi Lake is largely attributed to rapid structural change of local agricultural activities and population growth. As shown in Figure 4.3, the phosphorous loads from the agricultural runoff, animal husbandry and human sources have been increased considerably over the last decade, while the load from phosphate industries have been decreasing since 1998 because of successful industrial pollution control. The rapid structural transformation of crops cultivation, i.e., from autarkic grain production to export-oriented intensive flower and vegetable planting, has upgraded the input demand of mineral nutrients including phosphates. Individual farmer's pursuit of stable yield guarantee, thereby, has stimulated the application of chemical fertilizers and, more importantly, has stagnated reuse of night soil. This, together with gradual growth of both livestock and population, inevitably leads to a great deal of manure surplus and subsequently threatens to the water quality of Dianchi Lake. It is notable that mining industry and phosphate detergent consumption are exclusive from the figure because of the lack of historical data. Since a great deal of phosphate wastes generated and contained in tailings and washing wastewater via these two processes respectively could have potentially significant impact on the lake eutrophication, however, the related phosphorus flows should not be ignored in the SFA model, the discussion of which we now turn to.

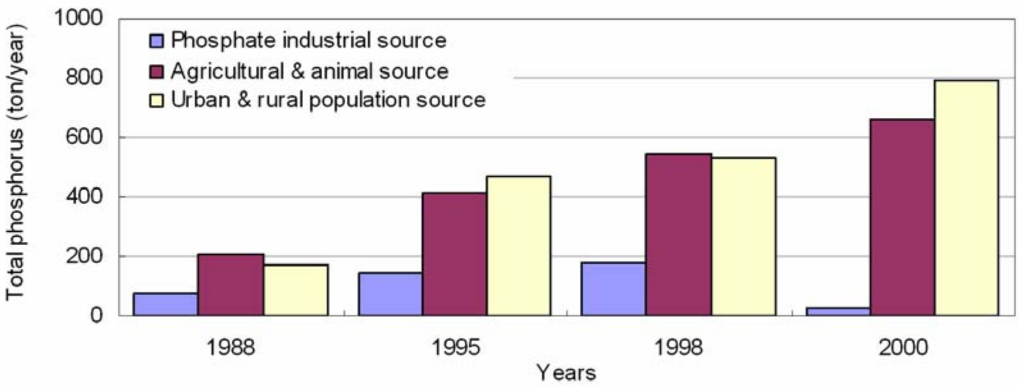

Figure 4.3 Total phosphorus loads from different sources of Dianchi Lake

Sources: based on YPPG (1998).

4.4 Modeling Phosphorus Flow in Dianchi Watershed

Taking 2000 as the base year, a statistical SFA model of phosphorus metabolism was developed for the Dianchi watershed in this section. The major data were derived from the Kunming Statistical Yearbook (2001). The other data were from sources listed in the Appendix III, where they were tabulated into several categories in accord with the social-economic metabolism of phosphorus, which would facilitate the understanding of the structure of our SFA model as below.

4.4.1 Structure of the SFA Model

The phosphorus SFA model consists of six subsystems shown in Figure 4.4. The first one is the mining subsystem, from which the natural phosphorus flows into the economic society and a great deal of wastes in forms of mineral tailings is generated as well. The second is the subsystem of physical processing and chemical refinement of phosphate minerals, i.e., phosphate industry, in which the raw phosphorite ores are turned into phosphate products. In parallel, phosphate-contained industrial wastes are also produced at this stage, which are either reused or released into the environment. Since chemical fertilizers are the dominant phosphate products, agricultural activities, which cultivate crops on soils and produce phosphorus runoff from soils, are thus included in the model as the third aspect. The forth aspect is related to the substantial discharges from breeding animals, i.e., livestock subsystem, from which the phosphorus disposal is to a large degree dependent on the reuse of manure. Finally, there are domestic phosphate-contained wastes, including excreta, washing wastewater and garbage, generated from households, which is operated in urban and rural population subsystem respectively in our model, since the manner of phosphates reuse as well as treatment significantly varies in accord with the life style of people.

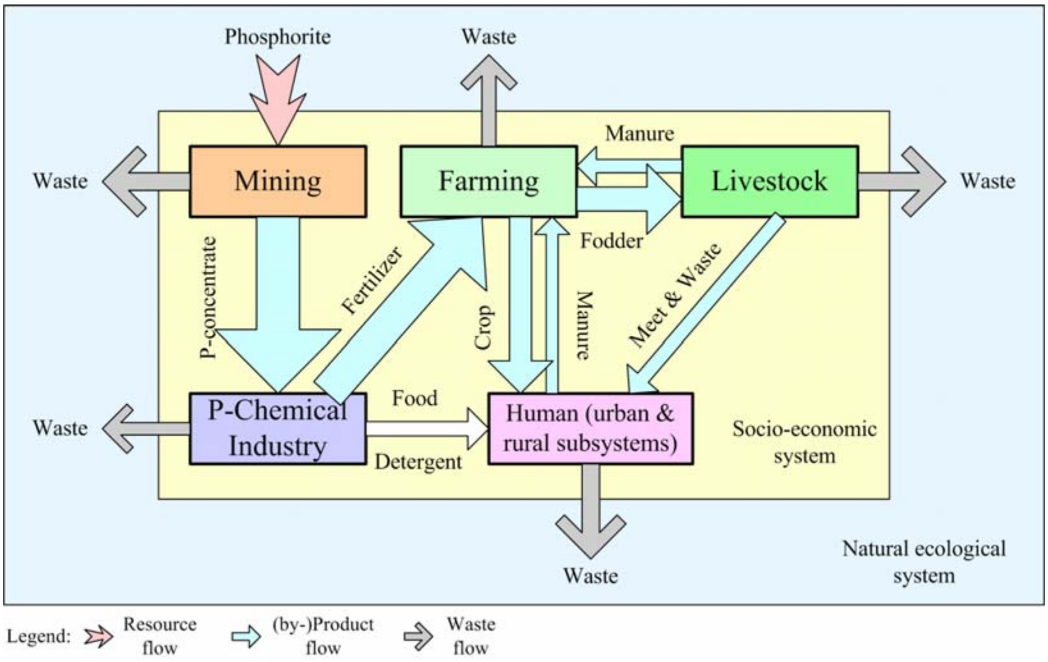

Figure 4.4 Six component subsystems of the static phosphorus flow analysis model

The six subsystems, comprising of the whole phosphorus metabolism of the local economic society, are correlated and interacted, and thus jointly function on the nature. First of all, the extraction of phosphate reserve and subsequently the development of phosphate industry, mostly driven by agricultural production, unavoidably lead to the continuous depletion of resource, and ecological degradation if the massive wastes could not be treated appropriately. Next, the chemical phosphates, the dominate industrial product of phosphate plants, input to agricultural lands does not only facilitate producing crops but also contribute to the output of crop remains, which could either function as phosphorus stores or be leaked to the environment. Third, the farming subsystem connects to human and livestock via complex phosphorus flows of nutrient (re-)distribution, and post-metabolized phosphorus is recycled to agricultural lands directly or after composting as organic manure, or disposal to the nature. Finally, the consumption of detergents also links phosphate industries and people’s daily life, and, more importantly, leads to a great emission of phosphates.

The conceptual framework of the SFA model is presented in Figure 4.5. The skeleton of the model consists of nodes, representing the processes that handle phosphorus, and arrows, denoting the related phosphorus flows from one node to another. The resource flow and waste flows were indicated by RS and S_i (i=1, …, 4) respectively, while the flows into the Dianchi lake, the environmental sink, were labeled as W_j (j=1, …, 12). In order to distinguish the phosphorus flows in our different subsystems, M_k (k=1, 2), I_p (p=1, …, 4), F_q (q=1, …, 7), L_w (w=1, …, 4), R_x (x=1, …, 5), and U_y (y=1, …, 7) were used as indicators in the SFA model respectively. Because of the insufficiency of data, a couple of nominal phosphorus flows, i.e., broken arrows labeled by N_z (z=1, …, 10), which denoted relevant flows but had no fixed value, were supplemented to balance several nodes of the model, particularly to fill up the gap of phosphorus nutrient consumption of human and livestock.

Based on the SFA model, framed by all these nodes and arrows, some relevant results in relation with ecological efficiency of local phosphorus metabolism in the Dianchi Watershed, can be drawn out, as discussion in the next section.

4.4.2 Results Discussion

As shown in Figure 4.5, the estimated total net phosphorus load to the Dianchi Lake was 1693 tons in 2000, of which mining, phosphate industry, farming, livestock, urban and rural population subsystems contributed to 0.1, 1.7, 12.9, 23.2, 33.6 and 28.5 percent respectively as presented in Table 4.1. Apparently, it was the human source, responsible for 1052.1 tons phosphorus release (W_3~W_5, W_7~W_9), and agricultural activities, including both farming and livestock breeding and responsible for 610.9 tons phosphorus load (W_6, W_{10}~W_{12}), dominated the forces for influencing the phosphorous related water environmental problems in the Dianchi Lake. With respect to water quality control, thereafter, the phosphorus metabolism within both population subsystems and farming and livestock subsystems should be the central concern.

Table 4.1 Key Sources of Phosphorus Load to Dianchi Lake in 2000

Source		Phosphorus Load (tons)	Percentage (%)
Mining		2.0	0.1
P-Industry		28.0	1.7
Agriculture	Farming	218.6	12.9
	Husbandry	392.3	23.2
Human	Urban Population	569.6	33.6
	Rural Population	482.5	28.5
	Sum	1693.0	100

Since Dianchi Watershed is one of the important bases of phosphorus resource in China, the fluxes of phosphorus flows within mining and industry subsystems were very intensive, indicated by massive export of phosphorite ores (M_2) and related phosphate products (I_1) in Figure 4.5. But, the sum release to Dianchi Lake from mineral extraction and industrial processing was surprisingly small, i.e., only 30.0 tons (W_1, W_2), being of marginal impact on its eutrophication. Taking consideration of the fact that local phosphorus reserve is rich but certainly limited (0.35 billion tons), however, it is estimated the phosphorite resource would be exhausted after 130 years more or less with the extractive intensity in 2000. On the other hand, as the large amount of solid wastes, mainly including gangues, tailings and gypsums, were generated during the processes of intensive extraction and refinement (S_1, S_2) and were commonly stacked up with few safeguards, ecological degradation, particularly in mining areas, was severe and water environment was also endangered via soil erosion and leaching. The resource depletion and environmental stress attributed to the low ecological efficiencies of mining and phosphate chemical industries, which were only 50.0 percent ($1-S_1/RS$) and 63.7 percent ($1-S_2/M_1$) in 2000, respectively. Hence, reduction of wastes yield and promotion of waste reuses should be the direction towards re-organizing the phosphorus metabolism within mining and industry subsystems, and would thus mitigate the ecological burden of Dianchi Watershed.

Through application of chemical fertilizers, indicated by I_2 in our model, 4049.6 tons phosphorus is transferred from industry subsystem to agricultural lands in 2000. The input of chemical fertilizers, together with manures, contributed to 1387.0 tons phosphorus yield (F_2) and 202.5 tons phosphorus loss (W_6) via agricultural runoff and leaching in 2000. Unutilized phosphorus, accounted for 5089.9 tons as indicated by F_1, was then deposited in agricultural soils for the next cropping season. As the main agricultural by-product, crop stalk was a rich nutrient stock with 383.7 tons phosphorus (F_3). While crop stalks were widely reused by raising family livestock (F_4), composting manure (F_5), and generating methane (F_6), only 4.2 percent (W_{10}/F_3) of contained phosphorus flowed into Dianchi Lake ultimately. It is notable that manure reuse, including directly recycling of human and livestock's excreta (L_1, L_4, R_4) and indirectly reuse of organic wastes by composting (F_7), was responsible for 39.4 percent of total phosphorus input to agricultural lands. Although the phosphate nutrient proportion of manure to total input was rather higher than China's average (Jin and Portch, 2001; Liu et al., 2004), there still was a large gap between recycled phosphorus and required phosphorus by crops production. If not being improved, the gap would absolutely pass on feedback to mining industries and consequently lead to an ever-increasing intensity of resource extraction.

Animal husbandry assimilated a large amount of crop stalks, provided the main source of manure, and also produced a heavy pollution. Among the total phosphorus efflux from livestock subsystem, 79.2 percent, responsible for 1921.2 tons phosphorus, was returned to agricultural soils, while 16.2 percent phosphorus was released into Dianchi Lake. In comparison with animal husbandry in centralized feedlots or farms, livestock breeding in rural households is more of importance for it relates to individuals' farming activities and daily life closely. As presented in Figure 4.5, there was a large amount of 1736.4 tons phosphorus from domestic livestock's excreta (L_1, L_2) returned to agricultural lands in 2000. Another relative small part, responsible for 110.9 tons phosphorus (L_3), was re-utilized to produce domestic methane for household cooking and hearting. While the total recycling ratio of excreta from scaled-breeding animals was only 52.6 percent, that of family-breeding reached 89.1 percent in 2000. Because of the great quantity of excreta generation, however, both the two breeding activities considerably contributed to the eutrophication of Dianchi Lake. This strongly suggests not only the phosphorus flows related to centralized feedlots but also those produced by family-breeding activities should be regulated and re-directed.

With respect to the differentiation of processing phosphate-related wastes between urban and rural areas, the SFA model examined relevant phosphorus flows within urban and rural population subsystems respectively. In rural areas, domestic organic wastes are broadly utilized for living and agricultural production. As shown in Figure 4.5, the majority of human excreta, indicated by a sum of 584.8 tons phosphorus (R_4, R_5), was returned to farmlands. As some rural households in Dianchi Watershed adopt septic tanks, there was a small part of human excreta, responsible for 37.4 tons phosphorus (R_6), was used for generating methane for family cooking and heating. The rests, accounted for 76.1 tons phosphorus (W_7) and 10.9 percent of total, leaked into the lake. Without necessary sewage and treatment systems, domestic wastewater, responsible for 218.4 tons phosphorus (W_8) attributed to the consumption of phosphate-contained detergents, was directly discharged into surroundings and flowed into the lake ultimately along natural canals or dug ditches. Regardless of specific recycling manners and related volumes, the un-utilized rural organic garbage, contained 188.0 tons phosphorus (W_9), entered Dianchi Lake via various approaches. Because of the organic wastes would be decomposed rapidly in water environment, the released phosphorus also stressed on the lake.

Unlike farmers were closely related to farming subsystem and livestock subsystem through the (re-)distribution of phosphate nutrient and reuse of phosphate-related wastes, people live in cities and towns have less association with rural areas, exclusive of their ever-increasing food requirement. Without appropriate recycling approaches, accordingly, the huge amount of urban wastes was subject to treatment via urban infrastructure, which then, to a certain degree, mitigated ecological stress on Dianchi Lake. As shown in Figure 4.5, the prevention of 83.0 percent of human excreta (U_4) and 83.6 percent of domestic garbage (U_6) from Dianchi Lake, responsible for 2490.5 tons phosphorus (S_3), should attribute to the functioning of urban public sanitary system, including wastes collection, transportation and disposal, i.e., incineration or landfill. Likewise, centralized wastewater treatment system, consisting of urban sewages and wastewater treatment plants (WWTP) collected 17.0 percent of human excreta (U_5) and 61.8 percent of washing wastewater (U_7), and transferred 74.1 percent phosphorus of the total into sludge (S_4). Consequently, total 569.6 tons phosphorus (W_3-W_5) was released to the lake, and the rests, accounted for 2920.8 tons metabolized phosphorus (U_3-U_4) from urban population subsystem, were disposed as worthless wastes and, if not being reused, it would result in a significant secondary pollution.

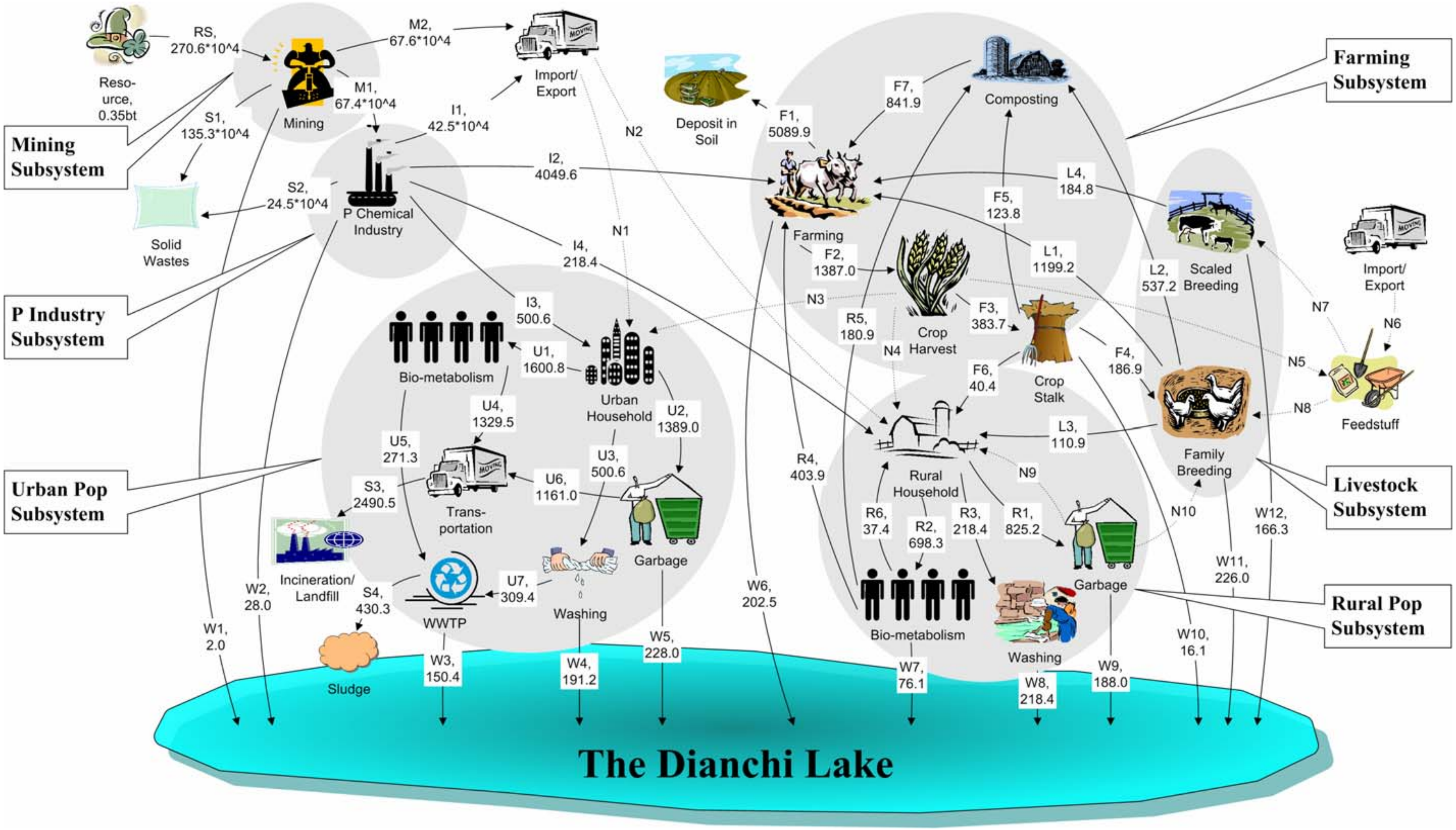

Figure 4.5 A SFA model for local phosphorus metabolism in Dianchi Watershed in 2000

4.5 Conclusion: Towards Prospects

Environmental pollution is commonly caused by irrational and inefficient distribution and utilization of materials. By the SFA approach, this paper quantitatively describes how the local human society extracted resource from the nature, metabolize it via various economic activities, and excrete pollution into the environment, in the case of phosphorus within the boundary of Dianchi Watershed. In general, the results suggested water environment is largely of dependence on the local phosphorus metabolism, and thus, if the related phosphorus flows, particularly referring to recycling and wastes flows, could not be re-organized towards a more ecologically sound direction, Dianchi Lake would be unavoidably and continuously suffering eutrophication.

Towards more economic efficiency and environmental effectiveness, some kinds of (radical or gradual) shifting of the local phosphorus metabolism are desired. First of all, as it is apparent the linear and open-ended metabolic manner of human society is functioning relying on massive and continuous input of phosphorus resource, the organic wastes from human and livestock by all means should be recycled to agricultural lands towards a more closed-loop. Second, cautiously balancing phosphate in soils with those in removed crops and chemical and organic fertilizers would maintain soil fertility precisely and then stimulate structural re-organization of phosphate industries and benefit agricultural pollution control as well. Third, improvement of productive efficiencies and wastes reuse within mining sectors and phosphate industries should also be one of future directions, although a success of industrial wastewater control was achieved. Forth, except for accelerating construction of urban infrastructures for which facilitate urban pollution control, rational disposal of secondary wastes, including a large amount of sludge from WWTPs, filtrate of land-fill plants and leaching liquid caused by rainfall, should be taken into consideration. Fifth, reducing consumption quantity of phosphate-contained detergents via encouraging development and application of substitutes is required since a considerable part of phosphates came from washing activities in our model.

Based on the illumination of future directions, smart policy-making should take a holistic view with respect to both economic efficiency and environmental effectiveness. Towards ecologicalizing the local phosphorus economy, on the one hand, reduction of phosphorus, particularly chemically synthetic phosphate input to agricultural lands and consumed by human, and transferring of organic phosphate from wastes to secondary resource should be central concerns. On the other hand, taking account of the notion of economizing the local ecology, environmental improvement would be achieved by a number of incentives which would not harms individuals' interests, for instance, manure reuse would never be predominate if rural farmers would persist pursuing short-term's economic profits and urban citizens would not give up water-flushing toilets; and ban of phosphate-contained detergents would not be advocated if people were unwilling to buy substitutes because of economic or customary reasons. Harmonizing local economic development and ecology, thereby, through exploring societal dynamics of the phosphorus metabolism, would be taken seriously into consideration.

References

Adriaanse, A., S. Bringezu, A. Hammond, *et al.* 1997. Resource flows: the material basis of industrial economies. Washington, D.C.: World Resources Institute.

Amann, C., W. Bruckner, M. Fischer-Kowalski, *et al.* 2002. Material flow accounting in Amazonia: A tool for sustainable development. Social Ecology Working Paper 63, IFF.

Ayres, R. U. 1994. Industrial metabolism: theory and policy. In *Industrial Metabolism: Restructuring for Sustainable Development*, edited by R. U. Ayres and U. E. Simonis. Tokyo/New York/Paris: United Nations University Press.

Bartelmus, P., and A. Vesper. 2000. Green accounting and material flow analysis: Alternatives or complements? Wuppertal, Germany: Wuppertal Papers No.106 (ISSN 0949-5266), Wuppertal Institute.

Bouman, M., R. Heijungs, E. Van der Voet, *et al.* 2000. Material flows and economic models: an analytical comparison of SFA, LCA and partial equilibrium models. *Ecological Economics* 32 195-216.

Bringezu, S. 2002. Towards sustainable resource management in the European Union. Wuppertal, Germany: Wuppertal Papers No.121 (ISSN 0949-5266), Wuppertal Institute.

Chen, J. F., and G. X. Tan, eds. 1989. *Production and Application of Phosphate*. Chengdu: Chengdu Science and Technology University Press. (in Chinese).

Chen, J. N., and T. Z. Zhang. 2003. General Plan for Ecological Economy and Urban Development of Guiyang Municipality in Southwestern China. Beijing: Department of Environmental Science and Engineering, Tsinghua University.

Chen, X. Q., and L. J. Qiao. 2001. A preliminary material input analysis of China. *Population and Environment* 23 (1): 117-126.

Daniels, P. L., and S. Moore. 2001. Approaches for quantifying the metabolism of physical economies - part I: methodological overview. *J. Industrial Ecology* 5 (4): 69-93.

DESE, KIES, and KWHIDI. 2003. Plan on Ecological Protection for Dianchi Lake Basin. Beijing: Department of Environmental Science and Engineering, Tsinghua University. (in Chinese).

Dokulil, M., W. Chen, and Q. Cai. 2000. Anthropogenic impacts to large lakes in China: the Tai Hu example. *Aquatic Ecosystem Health and Management* 3 81-94.

Driver, J. 1998. Phosphates recovery for recycling from sewage and animal wastes. *Phosphorus and Potassium* 216 (Jul/Aug): 17-21.

Fenton, M. D. 2001. Iron and steel recycling in the United States in 1998. Denver: Open file report 01-224, U.S. Geological Survey, U.S. Department of the Interior.

Fleig, A.-K. 2000. Eco-industrial parks: A strategy towards industrial ecology in developing and newly industrialized countries. German: GTZ.

Foster, J. B. 1999. Marx's theory of metabolic rift: Classical foundations for environmental sociology. *American Journal of Sociology* 105 (2): 366-405.

Frosch, R. A., and N. E. Gallopoulos. 1989. Strategies for manufacturing. *Scientific American* 261 (3): 144-152.

Fujie, K., and N. Goto. 2002. Materials flow analysis and modeling to establish a material flow network in regional area. *Sino-Japan Workshop on Circular Economy and Society*, at Beijing, Oct. 10-11.

Guinee, J. B., J. C. J. M. van den Bergh, J. Boelens, *et al.* 1999. Evaluation of risks of metal flows and accumulation in economy and environment. *Ecological Economics* 30 47-65.

Han, X. G., L. H. Li, and J. H. Huang, eds. 1999. *An Introduction to Biogeochemistry*. Beijing: Higher Education Press. (in Chinese).

Hannah, L., D. Lohse, C. Hutchinson, *et al.* 1994. A preliminary inventory of human disturbance of world ecosystems. *AMBIO* 23 246-250.

Huang, Z. Q. 2001. Phosphate resources and its industrial development in Yunnan province. *Conservation and Utilization of Mineral Resources* (2): 17-21. (in Chinese).

Janicke, M. 1995. Green industrial policy and the future of "dirty industries". Free University Berlin: Unpublished paper.

Jiang, S. X., ed. 1999. *Phosphoric Acid, Phosphorous Fertilizer and Compound Fertilizer*. Beijing: Chemical Industry Press. (in Chinese).

Jin, X. C. 1995. *Lakes in China - Research of Their Environment*. Vol. I & II. Beijing: China Ocean Press. (in Chinese).

Jin, X. C., ed. 2001. *Techniques for Control and Management of Lake Eutrophication*. Beijing: Chemical Industry Press. (in Chinese).

Jolly, J. H. 1992. Material flow of zinc in the United States 1850-1990. Denver: Open file report/1992, Bureau of Mines, U.S. Department of the Interior.

Kelly, T. 1998. Crushed cement concrete substitution for construction aggregates - A materials flow analysis. Denver: U.S. Geological Survey Circular 1177, U.S. Department of the Interior.

Kleijn, R., R. Huele, and E. van der Voet. 2000. Dynamic substance flow analysis: the delaying mechanism of stocks, with the case of PVC in Sweden. *Ecological Economics* 32 241-254.

Kleijn, R., A. Tukker, and E. van der Voet. 1997. Chlorine in the Netherlands - part I: An overview. *J. Industrial Ecology* 1 (1): 91-112.

Kramer, D. A. 2002. Magnesium recycling in the United States in 1998. Denver: U.S. Geological Survey Circular 1196-E, U.S. Department of the Interior.

KSB. 2001. *Kunming Statistical Yearbook 2000*. Beijing: China Statistics Press.

Liu, L. P. 2001. Analysis on eutrophication of Dianchi Lake and its control measures. *Yunnan Environmental Science* 20 (supplement): 25-27. (in Chinese).

Liu, Y., A. P. J. Mol, and J. N. Chen. 2004. Material Flow and Ecological Restructuring in China: the Case of Phosphorus. *J. Industrial Ecology* 8 (3).

Llewellyn, T. O. 1994. Cadmium (material flow). Denver: Information Circular 9380, Bureau of Mines, U.S. Department of the Interior.

Loebenstein, J. R. 1994. The material flow of arsenic in the United States. Denver: Information Circular 9382, Bureau of Mines, U.S. Department of the Interior.

Lu, S. L. 1997. Pollution and control countermeasure in Dianchi Lake. *Chongqing Environmental Science* 19 (3): 1-4. (in Chinese).

Luo, F. Y. 1997. A discussion on the position and the development of the phosphate rock in Yunnan. *Yunnan Chemical Industry* (4): 8-12. (in Chinese).

Matthews, E., C. Amann, S. Bringezu, *et al.* 2000. The weight of nations: material outflows from industrial economies. Washington, D.C.: World Resources Institute.

Moolenaar, S. W., E. J. M. Temminghoff, and F. A. M. D. Haan. 1998. Modeling dynamic copper balances for a contaminated sandy soil following land use change from agriculture to forestry. *Environmental Pollution* 103 117-125.

Moss, B. 2000. Phosphorus, detergents and eutrophication control - A summary of experience in Europe and North America. *Detergent and Cosmetics* 23 (2, Supplement, July): 117-129.

Ober, J. A. 2002. Material flow of sulfur. Denver: Open file report 02-298, U.S. Geological Survey, U.S. Department of the Interior.

Peng, X. Y., and X. R. Hou. 2000. The theoretic and practical analysis of relationship between material input and income comparison between China and other Countries. *Chinese Soft Science (Zhongguo Ruankexue)* (9): 33-38. (in Chinese).

Picton, T., and P. L. Daniels. 1999. Ecological restructuring for sustainable development: evidence from the Australian economy. *Ecological Economics* 29 (3): 405-425.

Posch, A. 2002. From 'industrial symbiosis' to 'sustainability networks'. The Environment and Sustainable Development in New Central Europe: Austria and Its Neighbors, at Minneapolis, USA.

Shu, J. H., W. Y. Huang, X. Y. Gao, *et al.* 1999. The evaluation of the effect of powdered detergent containing phosphate on Tai Hu Water Area Eutrophication. *Detergent and Cosmetics (Riyong Huaxuepin Kexue)* 22 (Supplement, August): 129-145. (in Chinese).

Si, Y. B., S. Q. Wang, and H. M. Chen. 2000. Nitrogen and phosphorus losses from agricultural soil and surface water eutrophication. *Journal of Soil* 37 (4): 188-193. (in Chinese).

Smil, V. 2000. Phosphorus in the environment: Natural flows and human interferences. *Annual Review of Energy and Environment* 25 53-88.

Smith, G. R. 2002. Lead recycling in the United States in 1998. Denver: U.S. Geological Survey Circular 1196-F, U.S. Department of the Interior.

Stauffer, J. 1998. *The Water Crisis: Constructing Solutions to Freshwater Pollution*. London: Earthscan.

Steen, I. 1998. Phosphorus availability in the 21st century: Management of a non-renewable resource. *Phosphorus and Potassium* 217 (Sep/Oct): 25-31.

Sznopek, J. L., and T. G. Goonan. 2000. The materials flow of mercury in the economies of the United States and the world. Denver: U.S. Geological Survey Circular 1197, U.S. Department of the Interior.

van der Voet, E. 1996. Substances from Cradle to Grave: Development of a methodology for the analysis of substance flows through the economy and the environment of a region. Ph.D. Thesis, Centre of Environmental Science, Leiden University, Leiden, the Netherlands.

van der Voet, E., J. B. Guinee, and H. A. Udo de Haes. 1999. Metals in the Netherlands: a problem solved? *CML-SSP Working Paper* 99.003, the Centre of Environmental Science, Leiden University, the Netherlands.

Wagner, L. A. 2002. Materials in the economy: Material flows, scarcity, and the environment. Denver: U.S. Geological Survey Circular 1221, U.S. Department of the Interior.

Wang, Y. F. 1998. Apprehension and foresight of phosphorous resource in Dianchi area, Yunnan province. *Geology of Chemical Minerals* 20 (2): 195-198. (in Chinese).

Wen, Y. 1998. A preliminary study on quantitative assessment approach to intensity of human activity. *Impact of Science on Society* (4): 55-60. (in Chinese).

Yang, H., and Z. Y. Ren. 2000. Study on nutrition evaluation of Chinese Lakes. *Jounral of Shaanxi Normal University (Natural Science Edition)* 28 (4): 104-107. (in Chinese).

YPPG. 1998. National Ninth-Five-Year Plan and Long-Term Program on Water Pollution Prevention and Control for Year 2010 for Dianchi Lake Basin. In *Water Pollution Prevention and Control Plans and Programs for Huaihe River, Haihe River, Liaohe River, Taihu Lake, Chaohu Lake and Dianchi Lake in China*, edited by SEPA. Beijing. (in Chinese).

Zhang, T. Z., L. Shi, and Y. Liu. 2003. Plan for Industrial Ecological Economy of Jiangsu Province in China. Beijing: Department of Environmental Science and Engineering, Tsinghua University. (in Chinese).

CHAPTER 5

Reflection on the Substance Flow Analysis Approach

5.1 Introduction

SFA approaches have been developed to depict the flows of substances through the economy. Different from numerical models which are constructed mainly relying on natural science-based monitoring and experimental data, SFA models follow the Mass Conservation principle in a more conceptual and experiential way. In general, SFA models should be complete, in the sense of covering extraction, production, consumption, waste treatment and recycling, and so forth (Bouman et al. 2000). However, it is not surprising that framing and fleshing SFA models is rather difficult for two reasons. First, physical interactions between human societies and environment are more complicated than people imagine. And second physical flows, unlike capital or cash flows, are seldom subject to full recording and monitoring of all levels of economies.

While a few industrialized countries started to include material accounting into their national statistics in the late 1990s, it is not the case yet in China. Moreover, operation of the official statistical system and the completeness and validity of data in China have been criticized (e.g., see Rawski and Mead 1998; Zhou and Ma 2003; Holz 2003, 2004). In this sense, handling China's economy by MFA/SFA approaches would not be as easy as it has been proclaimed (Kleijn 2000). Except for a few academic outputs on national bulk-material consumption (e.g., Peng and Hou 2000; Chen and Qiao 2001; Deng and Li 2002), application of these approaches in China has been barely reported up till now.

The construction of static SFA models in a given year, instead of time-related dynamic models, is an appropriate start-point to acquire an insight into P throughput within China's economy, as well as into the environmental consequences it causes. Even so, however, incomplete and invalid data raise vast difficulties in dealing with P flows in China. Huge gaps in data, ranging from resource production by mining, through distribution of manufactured products and semi- or by- products, to end-of-pipe treatment and reuse of wastes, significantly hampers determination of accurate quantities associated with each flow. Furthermore, inconsistencies among data from different sources undermine mass equilibrium calculation at the nodes. The processed raw ores, for instance, summed up to 3709 metric thousand tons in terms of pure P (mkt P) in 2000 according to various sources (see Appendix IV). But meanwhile total mineral output accounted for only 2536 mkt P according to the China's Statistical Yearbook 2001 (abbr. CSYB, NBS 2001). Taking import and export of P ores into consideration, the flux of actual extracted minerals ought to be 4218 mkt P which is 1.6 times the officially reported yield. This implies that a change in stock of 1682 mkt P would balance extraction and consumption, accounting for 66% to mining output in 2000. However, insight in historical data makes the enormous amount of stock release unbelievable, as national production and consumption of phosphorite has been relatively stable during the last decade, as presented in Figure 3.2 and 3.3.

Therefore, there is a need to turn back to the methodology of data processing, the crucial pivot for applying physical-based approaches for China. In moving towards a robust reduction of overall uncertainty associated with the models, subsequently, we present and discuss the main principles underlying data processing in section 5.2.

5.2 Principles to Reduce Data Uncertainty

Here we discuss six principles that can be applied in declines with data uncertainty in P flow models: data reliability, scale-related uncertainty, structural uncertainty, balancing technique, and model validity are taken into consideration, respectively.

First of all, basic socio-economic data coming from various statistical references must be verified before applying them into the models. In doing so, the official comprehensive databases provided by the National Bureau of Statistics (NBS) at the national level and the Kunming Statistics Bureau (KSB) at the local level, give the highest reliability. This approach, offering a relative stable output of models by shifting among individual estimates, can be appropriate, because it opens to systematic uncertainties introduced by official statistical system, but meanwhile it excludes stochastic errors, which are more likely to raise difficulties in data processing.[20] Moreover, it also allows a reasonable comparison with – or inclusion of – provincial configurations, since the corresponding data are in general consistent.

Second, aiming at remediation of spatial scale-related uncertainties, models should take as much as possible specific information on P flows at sub-geographic levels into consideration. Keeping in mind that modeling at a large spatial scale generally inclines to a magnification of uncertainty caused by adopting one *aggregated* (e.g. nation-wide instead of province-level) variable or parameter, this principle of reduction of spatial uncertainty is aiming at integrating regional configurations of scale varieties. For instance, one single leaching parameter applied for estimated agricultural runoff is replaced by a set of regional-based indicators according to new academic insights and researches.

Third, by adopting the assumption that economies operate in a steady state with static models, stock change is excluded. Accounting materials accumulated within the economy in terms of durable products usually takes place, where the release of materials would lead to direct

[20] For instance, all experts recognize that the total amount of national cultivated areas has been underestimated in the official statistics for years. However, it is rather difficult to determine the exact level of the underestimation from various professional studies (e.g., Smil 1995; Heilig 1999), till the promulgation of the national agricultural census in 2000.

environmental damage, e.g., metals (Guinee et al. 1999) or PVC (Kleijn et al. 2000). Regarding the P cycle, those soils on which human beings cultivate and harvest crops is a tremendous nutrient sink. However, the cultivated soil cannot be simply categorized into economic accumulation, because it is the indispensable physical basis of agricultural production as the acceptor of enormous P fertilizers application as well as one of vital sources responsible for P leakage. Treating cultivated soil as an essential component of crop farming sector, therefore, only input and output are taken into consideration in our P flow models.

Fourth, to obtain a reasonable solution of data inconsistency between input and output, we introduced so-called *derived* variables instead of officially *promulgated* production yields into the static P flow models.[21] This implies that models follow a demand-driven dynamic rather than a production-dominant one. In doing so, input quantity is derived via input-output balancing instead of directly extracted from statistical databases on production. We pose our trust on this method, because the data of human consumption of ultimate products, which comes up via social surveys conducted by central governmental agencies, is reliable rather than the local-central gathered data on production. Moreover, as waste flows are basically excluded from official statistics, this simplification allows us to combine environmental damages with economic activities, which is one of the essential liabilities of physical-based approaches. The input quantity thereby can be formulated as 'input = consumption + waste'.

The fifth principle focuses on another aspect of data balancing. As waste flows are barely subject to monitoring, the amounts of wastes are generally estimated based on production quantity and efficiency of resources use or, in most cases, even by emission coefficients only. However, this kind of *bottom-up* approaches may introduce considerable uncertainties because of ignorance of individual varieties among the same category.[22] More importantly, it cannot reveal a causally deduced relationship of resources utilization or nutrient ingestion and associated waste discharge. In contrast, we develop *top-down* techniques and apply for distributing crops as well as livestock products into consumption sections. It is apparent that this approach has overwhelming advantages, because it allows us to account animal and human wastes from a consumption perspective and thus reveals potential solutions in terms of dematerialization. However, the approach is only applied for illustration of the national P flows because of data limitation at the regional level. In that sense, its application runs counter to the second principle.

Last but not least, the main outputs of P flow models should be compared with other academic discoveries. Since there are barely specific studies on China's P flows, independent output

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[21] Derived variable is a statistic concept and broadly applied for quantitative analysis at macro level. For instance, social consumption is rather difficult to be quantified in comparison with production yield, because of the underlying diversity of consumption behavior as well as the limitation of statistical methods. In this case, 'derived consumption' is an applicable substitute for the real quantity. It is calculated by the formula "Production + Import – Export", regardless of accumulation and release.

[22] Pollution emission significantly varies among technology, scale, management and other characteristics of enterprise. There is also a distinction of daily nutrients discharge between adult and young animals. In general, the total amount of production is applicable but detailed information associated with subcategories is unavailable in China. Thus, the conventional method which poses its legitimacy on production-emission relation described by a single integrated parameter cannot generate an appropriate insight without taking individual variety into account.

data are desired as indicators to validate our models. We use 1) the percentage of chemical fertilizer production in total minerals utilization to validate the introduction of derived variables into mining and phosphate industries; 2) the ratios of P accumulation into and leachate from agricultural soils as indicators for farming subsystems; 3) the percentage of feed consumption in the total grain yield for animal husbandry; 4) the percentage of total organic fertilizers application in relation to chemical fertilizers to verify overall uncertainties associated with wastes recycling; and 5) the P ingestion per capita per day as a dietetic indicator for urban and rural residents to examine food consumption and related waste generation. Again, because of data insufficiency, our analysis of model validity is limited to the national P flows.

By using these new methodological principles of data processing, we return to our P flow models. We do that also with more recent data of the year 2000.

5.3 Development of PHOSFLOW Model in 2000

Departing from the former version of the "National Static Phosphorus Flow Analysis Model of China" (abbr. PHOSFLOW 1.0) as discussed in Chapter 3, in this section we develop the model for the year of 2000. The model is demonstrated in Figure 5.1 and the main data sources associated with each subsystem are presented in Appendix IV.

Insufficient data for China's cultivated area was the main reason for confining our earlier PHOSFLOW model to the year 1996. Before the original promulgation of the national agricultural census data in 2001 by NBS (refer to endnote 12 in Chapter 3), it has been argued that the amount of national cultivated area was significantly underestimated by 20% – 30% (Smil 1995; Heilig 1999). This would absolutely lead to an inevasible misunderstanding on fertilizing intensity and land carrying-capacity, among others. After CSYB (2001), however, the following national statistical yearbooks (2002 and 2003) repeat these figures again and no additional data were given on cultivated areas in other years.[23] In this sense, we turn to apply a database developed by the Institute of Soil and Fertilizers, the Chinese Academy of Agricultural Sciences (ISF 2003) for renovation of PHOSFLOW model to the year 2000 (remarked as version 3.0). This database is available with provincial configurations of cultivated area as well as P content in soils, and the national cultivated area was summed up to 129,986 thousand hectares in 2000.

As accumulation of P in the economy and stock releases of P to the environment are not taken into consideration in our static model, mining and fertilizer productions are balanced according to international trade, consumption and waste. Based on national statistics (NBS 2001; CNLIA 2001; CNCIC 2002), the derived quantities of domestic P extraction and manufactured fertilizers reach to 4885 mkt P (RP) and 3509 mkt P, increasing by 93% and

[23] The CSYB 2003 provides a primary estimate on national cultivated area in 2001 by an endnote on page 415. It remarks the area of cultivated land (total area) in 2001 is 127082 thousand hectares (th) according to NBS. In addition, the Ministry of Land and Resources of China (MLR) started to promulgate the Communiqué on Land and Resources of China in 2002. According to the last three communiqués, cultivated areas are 127615 th, 125930 th and 123392 th respectively, corresponding to the year of 2001, 2002 and 2003. However, all of them didn't provide data at the provincial level.

20% in comparison with officially reported yields (NBS 2003), respectively. The result seems reliable since the quantity of minerals manufactured as fertilizers accounts for 79% in total P ore output, which is consistent with other studies (Jiang 1999; Cui 2000). In addition, generation of gangues accounted for 392 mkt P in 2000 is estimated according to national mining productivity (JSDLR 2003). However, no specific data is available for development of leaching from mining sources.

Nutrient leaching from cultivated soils is refined towards reducing spatial scale-induced uncertainty. Taking varieties of fertilizing intensities, ratio of nitrogen to P uses and application of organic fertilizers among different regions into consideration, Lai (2004) proposed an entire set of leaching coefficients for each province. Relying on this study, we extend the aggregated parameter (refer to K_8 in Table 3.3) to province-level coefficients, which in turn improves the reliability of our model. The result shows agricultural runoff accounts for 245 mkt P in which chemical fertilizer is responsible for 162 mkt P, reaching 4% of application volume. In comparison with relevant studies (Waddell and Bower 1988; Haygarth 1997), this estimate falls in a reliable interval. Likewise, distribution of crop residues is enriched according to province-based studies (Han et al. 2002; Gao et al. 2002). Concerning all related sources, organic fertilizers sum up to 1672 mkt P and contribute 28% to total P applied for farmland, which is consistent with professional estimates (Wang et al. 1996; Jin and Portch 2001). By balancing the node of crops farming, the nutrient deposits in cultivated soil subsequently accounts for 2084 mkt P. According to the implication of steady-state models as discussed elsewhere earlier, therefore, the accumulated utilization ratio of P absorbed by plants can be balanced as 61%, regarding various nutrient losses. This result can be further validated by some agricultural researches (Cui 2000; Chen 2002).

Using a top-down approach, P contained in various agricultural products is distributed to animal husbandry as well as residents, on the basis of China's Food Balance Sheet (abbr. CFBS, FAO 2003a)[24]. Feed consumption is estimated based on an approach in relation with feed-meat conversion ratios. In doing so, yields of various kinds of livestock products (NBS 2001), including beef, pork, mutton, poultry meat, milk and eggs among others, are taken into consideration. The corresponding animals account for an overwhelming proportion to overall husbandry in terms of both products yield and output value. Furthermore, feedstuff consumption structure (Luo 2000) and feed-meat ratios (Lu 1997) are adopted. As a result, animal husbandry consumed 29% of harvest grains which is consistent with the consensus of Chinese scholars (Cheng et al. 1997; Qin 1997; Zhou 2001). If we keep coefficients of nutrient loss unchanged (SEPA 2002; Jia et al. 2002; Chen 2002; Shen 2001), the approach leads to a waste yield of 2148 mkt P by breeding animals and 387 mkt P that leaked to the water environment in 2000. Thus, general P loss ratios of intensive feedlots and family-based farms accounted for 25% and 10% respectively.

[24] CFBS provides a comprehensive database on national agricultural production, consumption, international trade and stock on domestic supply side, and feed, food, manufacture, waste and other uses on domestic utilization side. All main cereals, starchy roots, sugar crops, pulses, oil crops, vegetables, fruits as well as meets, milk and eggs are independently accounted. Among which, basic productive figures are provided by NBS, and the others are estimate by FAO itself. According to CFBS 2000, feed grain accounts for 30% of total domestic utilization of grain in terms of pure P. This result is principally consistent with relevant studies (e.g., Zhou 2001).

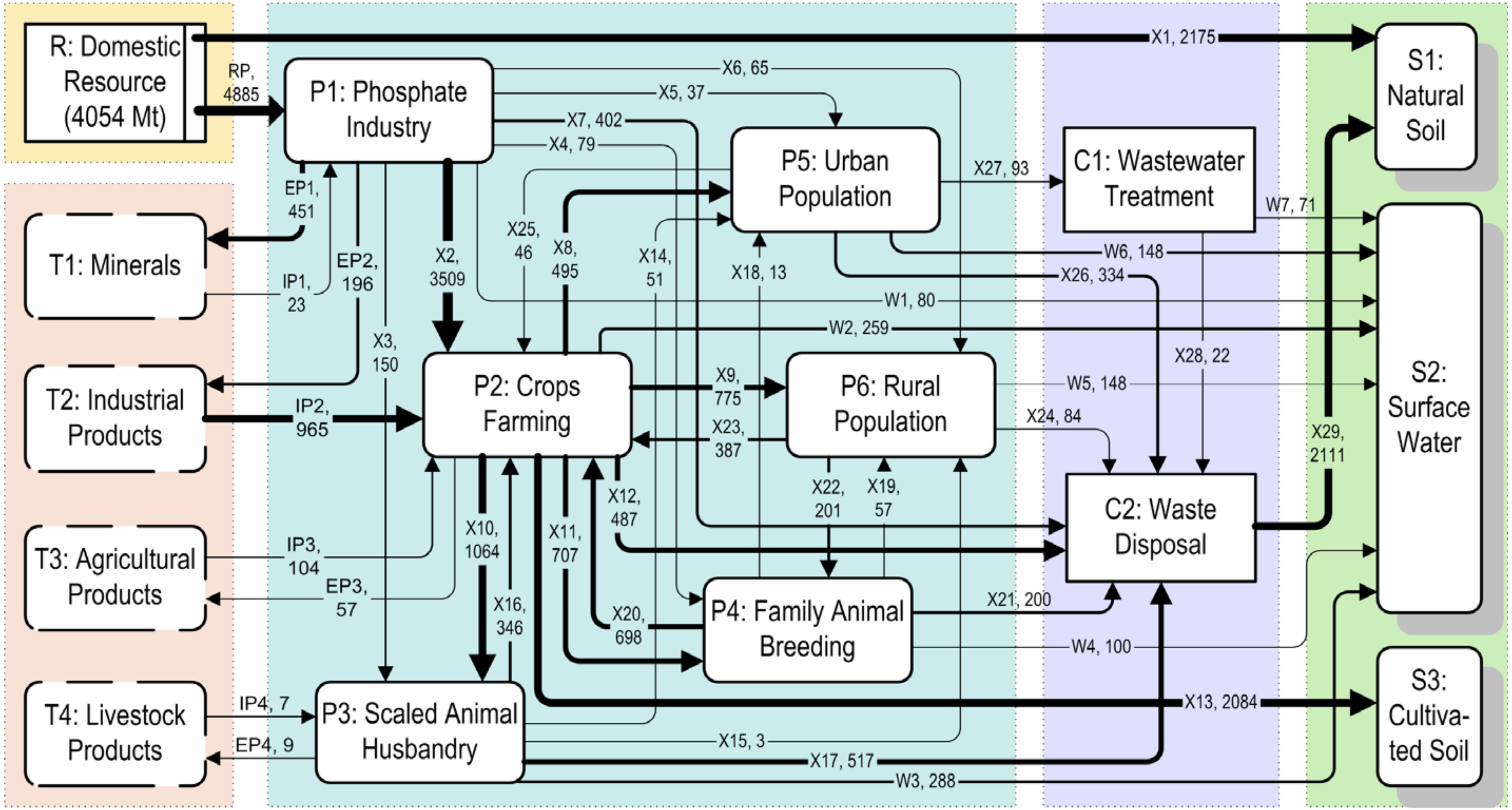

Figure 5.1 The updated national static P flow analysis model in 2000 (PHOSFLOW 3.0)

The national food supply configuration is built up by combining CFBS 2000 within the statistics of food consumption by urban and rural residents (NBS 2001). In addition, we assume that the food demand of urban population has more influence on the output of intensive feedlots than that of rural consumers. A main output of this approach is a daily nutrient ingestion of individuals, i.e., 1402 mg P/capita*d of urban residents and 1470 mg P/capita*d of rural residents. According to the last national census in 1992, these figures are 1077 mg P/capita*d and 1044 mg P/capita*d respectively (MOH 2004). Due to the rapid increase in livestock products consumption during the last decade (refer to Figure 3.4), the results are in need of further discussion.

There are some structure-related modifications of the new model compared to the early version. First, international trade is given close attention in our updated and restructured model. Import and export of P-containing products are classified into four groups: minerals, industrial products (fertilizers and STPP), agricultural products and livestock products. We will discuss the related influences on the domestic environment at national and local levels in section 5.5.

Second, P-related industries are simplified in terms of their physical structure. In Chapter 3, fodder industry and food industry were introduced to account for pollution loads. Whereas those industries posed little influence on national P flows from a physical point of view, they are skipped in this updated version. Meanwhile, productions of phosphoric acid, phosphor and feedstuff additives are articulated, which were treated as one category of 'other P chemical industries' in Chapter 3 (as indicated in Table 3.3). Together with the production of chemical fertilizers, they subsequently compose the subsystem of "Phosphate Industry"[25] in the updated model.

Third, for a better understanding, the PHOSFLOW 3.0 model separates the node "Food Consumption" into "Urban Population" and "Rural Population", along with the conceptual framework as shown in Figure 4.4. For the same reason, mass flows and related impacts of industrialized feedlots and family-based farms are taken into account independently.

Forth, the composting process is a difficult sector in the PHOSFLOW model, because it is impossible to find additional information on the structural features of recycling of organic fertilizers, which consist of reused human excreta, animal manure and crop residues. Whether the recycled masses are subject to composting or not, the eventual destination is the agricultural soil. The definition of composting in the earlier version of our model was also vague itself. Although advanced composting infrastructure in terms of both know-how and facilities, has been underdeveloped in rural areas, some ordinary composting activities, e.g., water-logged composting in manure pits and retting on farm-fields beside roads, are regularly found since farmers are used to apply these conventional approaches to decompose organic mass and increase nutrients absorptivity of crops. The node 'composting', thereafter, is removed from the updated version and related flows are redirected into 'farming' node.

[25] Phosphate industry refers to a chain of industries involved in concentration of raw ores and a series of physical and chemical processes of refinement of phosphor, synthetic fertilizers, feedstuffs, food additives, industrial phosphate and other commodities of phosphatic compounds. For a sophisticated discussion, see Jiang (1999).

5.4 Modification of PHOSFAD Model

The former version of the "Regional Static Phosphorus Flow Analysis Model of the Dianchi Watershed" (abbr. PHOSFAD 1.1) as discussed in Chapter 4 was built up early 2003. The shift in spatial scale from the national territory to a regional watershed considerably increases the difficulty in data processing. This is due to the limitation of regional official statistics, which has been mainly conducted on the basis of administrative districts but not for a geographical watershed. This significantly weakens data verification. Secondly, in comparison with data accessibility regarding national SFA modeling, regional data for the Dianchi Basin is less transparent and self-evident, because of political reasons[26]. Third, at a regional level some factors become more sensitive and influential in analyzing local P metabolism. Cultivation of flowers and vegetables compared to grains, for instance, inclines to more intensive use of fertilizers and plays a dominant role in agricultural leaching in Dianchi Basin. Therefore, persistence in a single leaching coefficient could considerably undermine model quality.

Fortunately, some valuable investigations and publications on Dianchi Basin raise the possibilities to improve the PHOSFAD model, as well as our knowledge on local P flows. By applying these principles, the modification of the first PHOSFAD 1.1 model is conducted basically along four directions: 1) refining physical flows with supplementary data collected after 2003, including inserting *missed* flows of the earlier version, revision of *incorrect* flows and removal of *redundant* flows associated with composting processing which has only a slight influence on the structure of material throughput; 2) to convert weight to metric unit; 3) to correct all percentages and tonnages into integers regarding the significance of digits and consistency with the output of PHOSFLOW model; and 4) to re-label flow names in parallel with the changes in flows. The updated model version, labeled as 2.2, is inserted as Figure 5.2. In comparison with the discussion of model refinement at the national level, this section details the modifications of the PHOSFAD model.

5.4.1 Mining and Phosphate Industry

The quantity of phosphorite mining yield in PHOSFAD 1.1 version was determined by a governmental plan of mining industrial development (Wang 1998). According to the Kunming Statistical Yearbook 2000 (abbr. KSYB, KSB 2001), however, actual flux was 7805 mkt in 2000 containing 30% P_2O_5, i.e., 1022 mkt P. Together with additional evidence on mining productivity and the quantity of transported raw ores for domestic use in other regions (Tao 2002), the extracted minerals (R_S) and related flows (i.e., M_1, S_1, E_1 and E_2) are subsequently changed, while other variables and parameters in relation to mining and industrial sectors have remained constant.

[26] Local governmental departments are more likely to be conservative in providing additional information besides the public data. They reject to do so, because they don't exactly know the political consequences and sometimes their interests are not involved in related governmental programs. In contrast, governmental agencies employed at national level have become more enlightened to open and maintain communications with academics under pressures from the central government as well as the public.

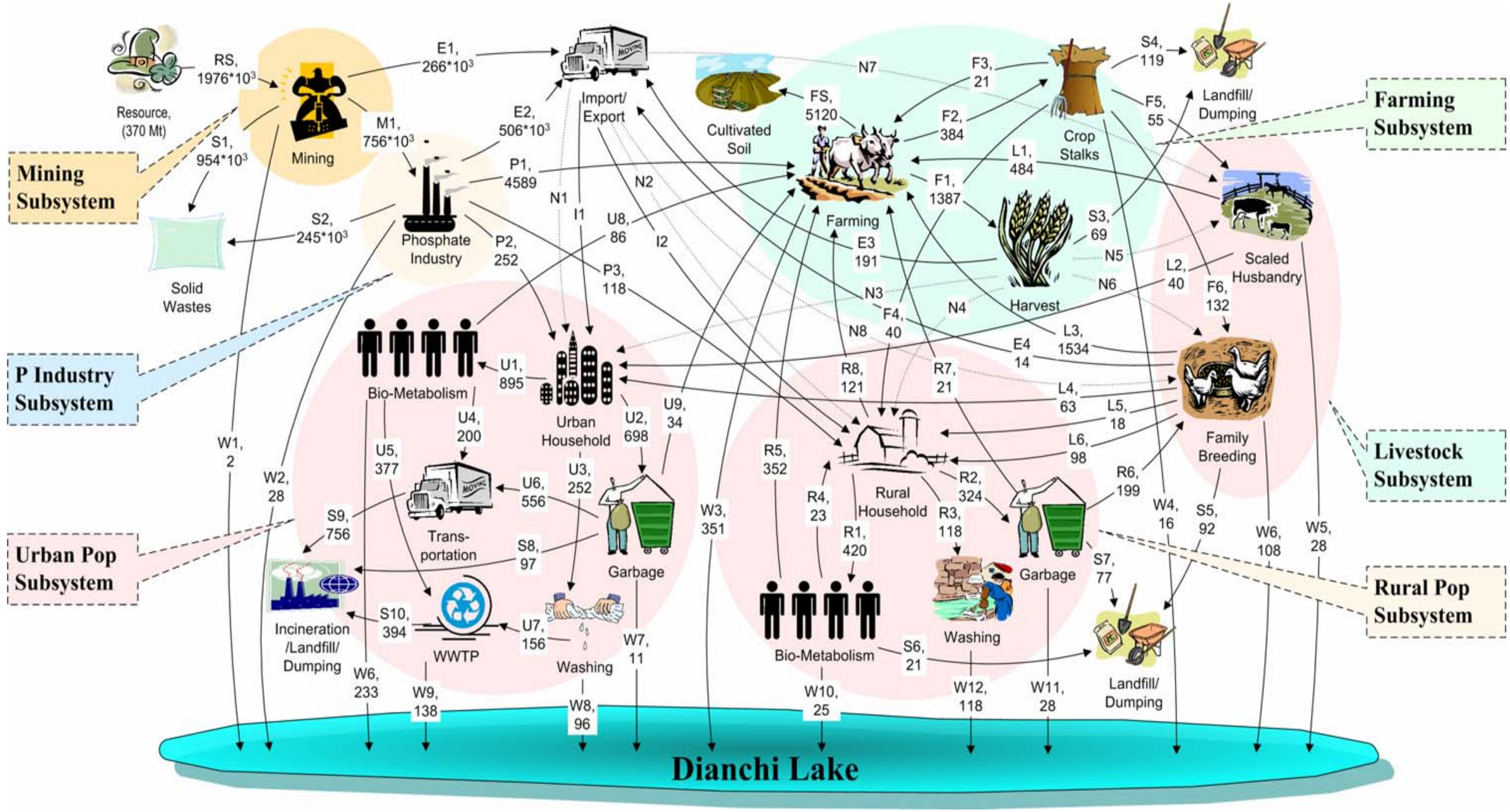

Figure 5.2 The updated static SFA model of Dianchi Watershed in 2000 (PHOSFAD 2.2)

No new findings appear to support further improvement of phosphate industry in our model. While leakages (W_1 and S_2) are kept consistent with those of version 1.1, the produced quantities of chemical fertilizers (P_1) and detergents (P_2 and P_3) are changed since they are derived variables depending on related consumption quantities, which we will discuss later. Accordingly, export of various industrial products (E_2) is rebalanced to keep the node equilibrium. Notably, due to data gap, the PHOSFAD model hasn't addressed sufficient attention to the detailed physical structure of exported commodities, compared to what has been done at the national level.

5.4.2 Crop farming

In comparison with crop utilization at the national level, it is extremely difficult to correctly distribute crops between animals and residents and thus to quantify related nutrient losses with respect to the ultimate destinations within the Dianchi Basin. This is due to the substantial information is lacking on products trading among different regions as well as on consumptive configurations of both animals and residents. It is nearly impossible to distinguish, for instance, the original production place of flour by which breads are produced and consumed. This suggests that our top-down analytical method is not an applicable tool in this case.

Keeping the yields of gross harvests and residues – two principal benchmarks – consequently constant, the model of PHOSFAD 2.2 uses the conventional method by which the quantities of crops utilized as feed and food are reversely deduced on the basis of emission coefficients of animals and residents, respectively. Again, because the quantities of trans-boundary P flows are unclear, we develop eight nominal variables (N_1 ~ N_8) to qualitatively indicate crops distribution. As presented in Chapter 4, nominal variables are those flows with unknown values. Since nutrient demand can be balanced using the principle of Mass Conservation for each relevant consumption node, a certain combination of a couple of nominal flows has a fixed value. For instance, N_1 and N_3 denote the demand of urban population in the model of PHOSFAD 2.2 and account together for 1485 metric tons in terms of pure P (mt P). Likewise, three other couples of nominal variables, i.e., N_2 and N_4 for rural population, N_5 and N_7 for intensive livestock husbandry and N_6 and N_8 for family breeding, are applied to indicate related demands in the model.

The crops-related physical structure is further complemented by two significant flows which lacked in the earlier version, i.e., the export flow of agricultural products, mainly including vegetables and flowers, to the outside of Dianchi Watershed (E_3), and the outflow of processing wastes to the environmental sink, i.e., natural soil (S_3). The former has a remarkable impact on the water environment, which we will discuss later. It is quantified according to official statistics on agricultural production and export (KAB 2001). The latter is estimated based on China's food balance sheet (FAO 2003a) and Zhan (1995).

As discussed above, P leaching from agricultural soils (W_3) is re-assessed. Taking crop variety and fertilizing intensity into consideration, one average leaching coefficient (kg P/ha) is replaced by four parameters based on in-situ monitoring results (Lu et al. 2003) and newly released academic researches (Zhang et al. 2003; Duan and Ming 2003). Apparently, the flux increases from 203 mt P to 351 mt P, and was thus remarkably underestimated in the earlier version of the model.

There are two changes in the distribution of crop residues. Utilization of crop residues include recycling to farmland, padding stalls, burning for household cooking and heating, and use for breeding animals (KAB and KAA 1999; Duan and Ming 2003). Keeping in mind that barnyard manure is hardly subject to recycling to farmland,[27] the updated model introduces an independent flow to natural soils (S_4) accounting for 119 mt P, combing wastes of stalls padding and un-utilized residues, which was ignored in the earlier version. The recycled nutrient to agricultural soils (F_3) thereby decreases to 21 mt P, deducting from the quantity used for animal barnyards. Second, the quantity of crop residues reused as feedstuff is separated into two independent flows, corresponding to intensified husbandry (F_5) and family-based breeding (F_6) respectively. It includes the assumption that feedstuff consumption relates linear to the number of animals. In addition, the composting process is removed for the same reason as discussed in section 5.3.

5.4.3 Livestock husbandry

The PHOSFAD models take the same animal categories into consideration as the national model. Due to data limitation as discussed early, a bottom-up method is applied for accounting the generation of animal wastes on the basis of daily P load per capita of each kind of livestock and poultry. Aiming at a comparable consistency with other domestic studies on animal wastes, the updated PHOSFAD model adopts the indicators officially advocated by SEPA (2002)[28]. The results show a slight shifting from 2425 mt P to 2344 mt P while the total amount of waste varies only 3%. But the configurations of industrialized husbandry and family farms breeding activities have changed considerably: animal waste from intensive feedlots increases from 351 mt P to 512 mt P and that generated by family farms falls to 1832 mt P from 2073 mt P. This is not surprising, seeing the large varieties in P load indicators of each kind of animal newly proposed by SEPA and those quoted in different literature sources.

Animal waste is subsequently re-allocated, keeping related utilization ratios unchanged compared to the earlier version (KAB and KAA 1999; KIES 2002). Among others, an additional waste flow from family breeding node pointing to natural soil (S_5) is discriminated from total nutrient loss to surface water (W_6), relying on our fieldwork with respect to utilization of animal wastes (Wang 2003).

The distribution of livestock products was previously hardly articulated. By following the same allocation mechanism as discussed in the former section, we distinguish three flows

[27] Based on the field investigation in 2002.

[28] The SEPA implemented the first national investigation on livestock husbandry between April 2000 and March 2001, and published the main findings in 2002. It is also the first time by which P emission coefficients of animals were proposed by the government. Although these indicators are inconsistent with existing estimates promoted by different organizations or experts, it is not surprising that they have been broadly accepted and cited by academics as soon as they were published, because it officially offers opportunities for comparative analysis among various studies on the environmental impacts of China's livestock husbandry.

bounded within the local economy (L_2, L_4 and L_5), two import flows (I_1 and I_2) and one export flow (E_4) crossing the geographical boundary of watershed, on the basis of factual consumed volumes of meats, eggs and milk per urban and rural resident (KSB 2001). These flows in relation with livestock products flesh the local physical structure, since it brings a possibility to connect husbandry activities to environmental impacts from a consumptive perspective.

5.4.4 Population

The reliability of population statistics in China has been discussed (e.g., Zhou and Ma 2003). The main focus is population classification and statistical calibers. Officially, population census has been conducted based on the system of “domicile register” (*huji zhidu*) since the early 1950s, which basically differentiates urban and rural population based on where people settle. However, this categorization is rather ambiguous, also because of the repeating changes of definitions of China’s cities and towns and the prevalent population migration. Moreover, because of the inconsistency between urban planning and construction with the operation of domicile register system, residents living in newly extended urban area cannot automatically obtain urban identities (domiciles). In other words, the amount of population living in urban areas is not equal to that of urban population in Chinese statistics. In addition, there are four geographic concepts attached to urban population statistics, i.e., city zone, constructed zone, urban area and municipal area. And the respective territories in principal become larger, which are also likely to lead to a puzzle of numbers.

As indicated in Chapter 4, the geographical territory of Dianchi Basin covers a number of sub-districts, involving Wuhua District, Panlong District, Xishan District, Guandu District, Chenggong County, Jinning County and Songming County, affiliated to Kunming Municipality. The former two districts constitute the so-called city zone and together with the other two districts comprise of the urban area of Kunming City. The three counties belong to part of municipal area. In the model of PHOSFAD 1.1, the amount of population within Dianchi Basin was accounted twice by summing up the number of population living in the urban area and that of each sub-district. In the new model, the overlapping number is deleted. The urban and rural populations of Dianchi Watershed thus decrease to 1518 thousands from 3020 thousands and 712 thousands from 1318 thousands, respectively, and the total population of the whole basin accounts for 2230 thousands. Since a bottom-up method relies on the population and the coefficient of nutrient load per capita, flows related to generation, disposal and reuse of human wastes are revised accordingly.

5.4.5 Wastes treatment/ disposal

The treatment of domestic wastewater and garbage in urban and rural areas are refined according to additional evidence. The development of urban infrastructure has reached a remarkable achievement in Kunming City compared to the whole country. However, apparently construction of urban infrastructures still lags behind the rapid urbanization and population growth, and rural areas overall lack fundamental environmental facilities. Only

62% of total domestic wastewater was collected by the urban sewage system and was thus subject to centralized treatment (U_5). Consequently, 74% of the contained P in this treated wastewater was removed and transferred into sludge via four treatment plants in Kunming City in 2000 (Ma et al. 2003). This indicates that a large part of wastewater is directly discharged into surrounding surface water (W_6), which we missed in the earlier model. Rich P-contained sludge deposited in septic sinks established in residential communities (between residential drainage system and municipal sewage) and sediment chambers of non-flushing sanitary facilities is periodically collected and transported out of cities. These are recorded by U_4 in both versions. However, the absolute quantity was over-weighted in the earlier version, as the P content carrying by flushing water was not discounted. A revised allocation factor, comparing the human bio-metabolic efficiency (Chen 2002; Shen 2001; Shu et al. 1999; Zhang and Qian 1998) and sedimentation efficiency of P in the septic sinks (He 2002), is applied to refine it.

The main destination of urban domestic sludge, as well as urban domestic garbage, is to be incinerated, land-filled or dumped. Only a small part can be recycled to agricultural soil after composting processes. The recycling flows, i.e., U_8 and U_9, are supplemented in version 2.2 to indicate reuse of urban excreta and garbage to farmland based on official statistics (MOC 2001). Outflow of urban garbage is complemented by flow (S_8) responsible for dumping of uncollected waste.

As sanitary facilities and related agencies are weak in rural areas, human excreta are periodically collected by farmers themselves and a majority is applied for maintaining soil fertility (R_5). Besides nutrient loss (S_6 and W_{10}), the remaining human excreta (R_4) together with family-breeding animal dung (L_6) are utilized to generate biogas for rural household lighting, cooking and heating. The decomposed outcomes, including sediments and liquid via fermentation, are dominantly reused on farmland (R_8), again because of their high organic fertility.

In addition, the quantities of rural garbage reused as feedstuff for family-breeding animals (R_6) and organic fertilizer are revised, based on a field survey in 2002 (Wang 2003). Accordingly, the reuse of rural garbage as household fuel material is balanced and redirected as waste into natural soil (S_7), instead of pointing to rural households.

The refinements we discussed above are presented in Appendix V in detail.

5.5 Comparative Analysis of Physical Characteristics

We now turn our attention to a re-examination of the physical profiles of the national and regional P flows. Unlike LCA, PIOT and bulk-MFA approaches, however, interpreting results of SFA models has not been normalized yet and differs between studies. This undermines the explanatory capacity of the SFA approach and limits its application as an analytical tool for decision making. In order to facilitate an understanding of the similarities and differences of physical features of the national and the local P metabolisms, we develop an approach to enable a comparative analysis. In parallel with the discussions in Chapter 3 and 4, we first analyze the aggregate structure of the national and local P flows in China, by applying the standard bulk-MFA framework (Matthews et al. 2000) as presented in Figure 2.2. Second, we will discuss the individual P metabolic features of involved societal production and consumption sectors, relying on a definition of four P efficiencies in relation with production, accumulation, recycling and emission.

5.5.1 Aggregate Physical Structure

As demonstrated in Figure 5.3, the national DMI of P accounted for 5317 mkt P in 2000. Domestic extraction, responsible for 82% of the DMI, was a dominant contributor. Most of these extracted P ores were consumed domestically and only 9% of them were exported. Notably, a huge amount of waste was generated, i.e. 45% per unit marketable ore yield, through processes of washing and flotation. As these wastes are usually dissipated and discarded as scrap-heaps, an increasingly ecological risk has risen, although their environmental impacts are not immediate.[29] Moreover, this accelerates a depletion of the non-renewable mineral resource. Based on a new official estimate of domestic P reserve (2003) and a foreign estimate (USGA 2004), the basic reserve of Chinese P, i.e. 4054 million tons with average P_2O_5 content of 17% – 22%, could be exhausted after 64 – 83 years.[30]

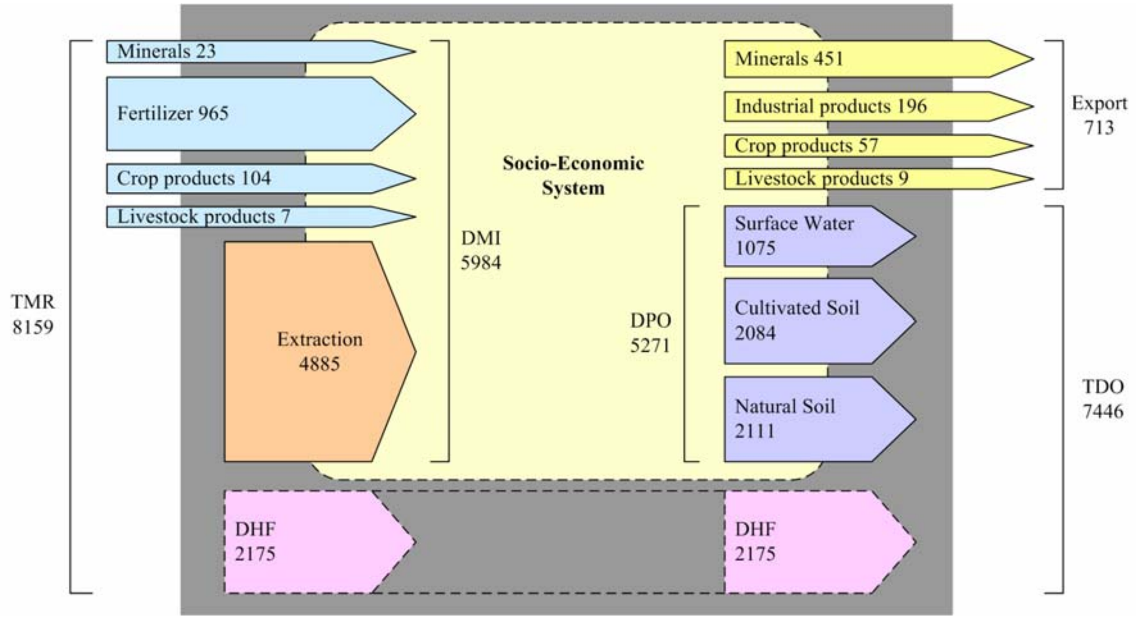

Figure 5.3 Aggregate structure of societal P metabolism in China in 2000 (unit: mkt P)

[29] According to a national investigation “Ecological Deterioration and Restoration of Mining” during Augest 1995 – March 1996 (refer to an official document of SEPA, *huanfa*[1997] no.448), only 5.2% gangue were subject to appropriate disposal.

[30] As complement to endnote 5 of Chapter 3, China has shifted from the conventional ABCD classification to an innovative accounting system since the late 1990s. On 8 June 1999, the State Bureau of Quality and Technical Supervision of China promulgated the national standard “Classification for Resources/Reserves of Solid Fuels and Mineral Commodities” (GB/T 17766-1999) proposed by the Ministry of Land and Resources of China (MLR). According to this new standard, solid mineral resources are discriminated into three categories: extractable reserve, basic reserve and resource. Among others, the basic reserve is principally comparative to the “reserve base” developed by the Department of the Interior of United States. In comparison with the conventional measurement, economic cost, feasibility assessment and geologic reliability are introduced. This standard was implemented on 1 December 1999 and national resources were accordingly re-evaluated.

Commodity import and export accounted for 1099 mkt P and 713 mkt P in 2000, and together contributed to a net import of 386 mkt P, which was mainly attributed to the huge input of chemical fertilizers. Since domestic demand on high quality compound fertilizers has been increasing annually as shown in Figure 3.3, a trade deficit in terms of P, which ultimately accumulates in domestic environments, is likely to remain stable. While the trading of agricultural commodities slightly affected the Chinese throughput in terms of P, the large imports of cereals and soybeans, accounting for 3% and 83% of domestic yields and 4% and 26% of global trading market (FAO 2003a; 2003b), respectively, triggers a worldwide controversy in food security of China (Brown and Halweil 1998; Heilig 1999; Wang and Davis 2000; Rosegrant et al. 2001). Export-oriented livestock production accounted for 7% of domestic yield and was responsible for 8% of animal waste emission. As over one half of the intensive feedlots, the main contributors to the export, are concentrated in a limited number of localities in China, the export of livestock products could become one of the main causes of local eutrophication.

With respect to the impacts of national P throughput on domestic environments, the Chinese DPO was 5271 mkt P in 2000, accounting for 88% of DMI. The P effluent to cultivated soils, natural soils and surface water was responsible for 40%, 40% and 20% of the DPO, respectively.

As one of three largest mining bases in China, P flows in Dianchi Basin are rather intensive.[31] As shown in Figure 5.4, the DMI of P to Dianchi Basin was 1025 mkt P and accounted for 460 kg P/capita in 2000, almost one hundred times to the intensity of national gross input. The local P ores yield, of which 26% of the raw ores were exported, was responsible for 21% of national total output in 2000. The huge magnitude of domestic hidden flow accounted for one half of the yield (Chen 2000), 5 per cent higher than the national average mining productivity. Keeping mining intensity constant, the estimated duration of local P extraction would sustain roughly 50 years, even sooner than the national figure.

Due to data insufficiency on product circulation on domestic markets, only grains and livestock products were considered on the input side, together less than one percent of the local DMI. In contrast, the huge magnitude of raw ores and chemical products, accounting for 75% of the DMI, dominated the aggregated structure of local P throughput. Compared to the national configuration, this suggests that the mobilization of local P resources was in principal oriented to export instead of local consumption.

Notably, the environmental impacts of the trade of agricultural commodities are twofold. As the import of grains was two times the local grain yield, it suggests that Dianchi Lake did not suffer from a P leaching comparable with what would happen if the imported grains were produced locally. However, this positive effect was largely counteracted by the intensive cultivation of flowers and vegetables around the lake, which occupied 28% of local farmland and contributed for 30% of local agricultural output value in Dianchi Basin in 2000 (KSB

[31] In parallel with implementation of the national standard for classification and evaluation of mineral resources, phosphorus reserve of Yunnan Province was accordingly re-evaluated in 2000. The results show the basic reserve of Dianchi Basin is about 370 million tons, accounting for 9% of national total. see Huang (2001); Tao (2002).

2001). Moreover, 40% of the produced flowers and 55% of the vegetables were exported (KAB 2001), which accounted for 86 g P/capita and almost doubled the P export intensity of all agricultural products at the national level. As characterized by a high fertilizing intensity and a mature drainage system, the production of flowers and vegetables put a strong stress on the lake, although the absolute quantities of these P flows were small compared to the overall P turnover in Dianchi Basin. According to the PHOSFAD model, export-oriented farming activity was responsible for 41% of agricultural runoff and 8% of total P load to Dianchi Lake.

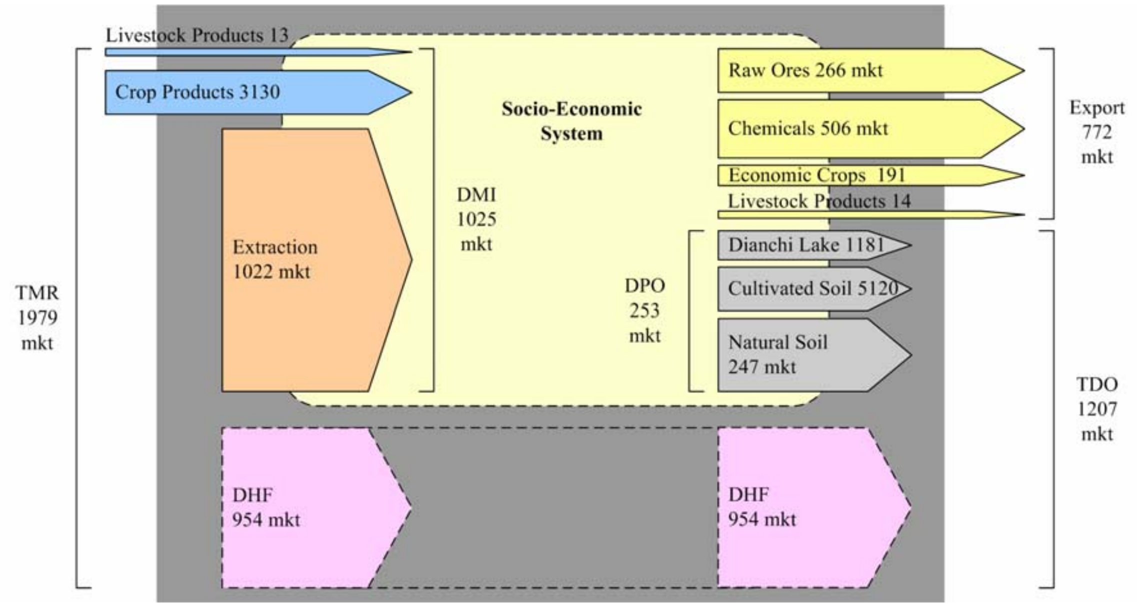

Figure 5.4 Aggregate phosphorus metabolic structure in Dianchi Basin in 2000 (unit: mt P)

The local DPO significantly differed from its national counterpart. Because of the huge amount of P export, the DPO only accounted for 25% of the DMI in 2000. Hence, the P accumulated in natural soils dominated the overall magnitude of the DPO. Because of local intensive farming activities, the nutrient enriched local cultivated soils with 2 kg P/capita, 40% higher than the average of whole country. In contrast, Dianchi Lake suffered a P load of 0.5 kg P/capita, which was 38% lower than its national counterpart.[32]

5.5.2 Sectoral Metabolic Efficiency

In this section, we use four indicators to describe the physical interrelation of sectoral P metabolism, i.e. production efficiency, emission efficiency, accumulation efficiency and recycling efficiency. On the basis of the P input, these P efficiencies of each sector can be

[32] Carrying capacity of Dianchi Lake in regard to P load has been discoursed. Recent results show the maximum load on Dianchi Lake should not exceed 38 – 75 mt P/yr to function as a desired requirement of Class III based on national standard on surface water quality (GB3838-2002), and a load range of 389 – 535 tons/year is estimated for the lowest requirement by Class V (DESE, KIES, and KWHIDI 2003; Yang and Yang 2002). In this sense, it is not surprising that that Dianchi Lake has experienced eutrophication with relative low P load.

defined as a percentage of the produced, emitted (to surface water), accumulated (in soils) and recycled P to the P input, respectively. As human being is the ultimate consumers, the production efficiency cannot make sense. We also ignore here the sectoral features of mining industry, which were included in the discussion on the aggregate structure in the former section.

The manufacturing of chemical fertilizers processed 79% P ores of domestic consumption and produced 3509 mkt P. Combining all Chinese phosphate industries, the production efficiency reached 88% in 2000 as shown in Figure 5.5. The huge amount of un-reused P in the form of bulk gypsum,[33] responsible for 84% of industrial solid waste, led to a low recycling efficiency of the P industrial sector.

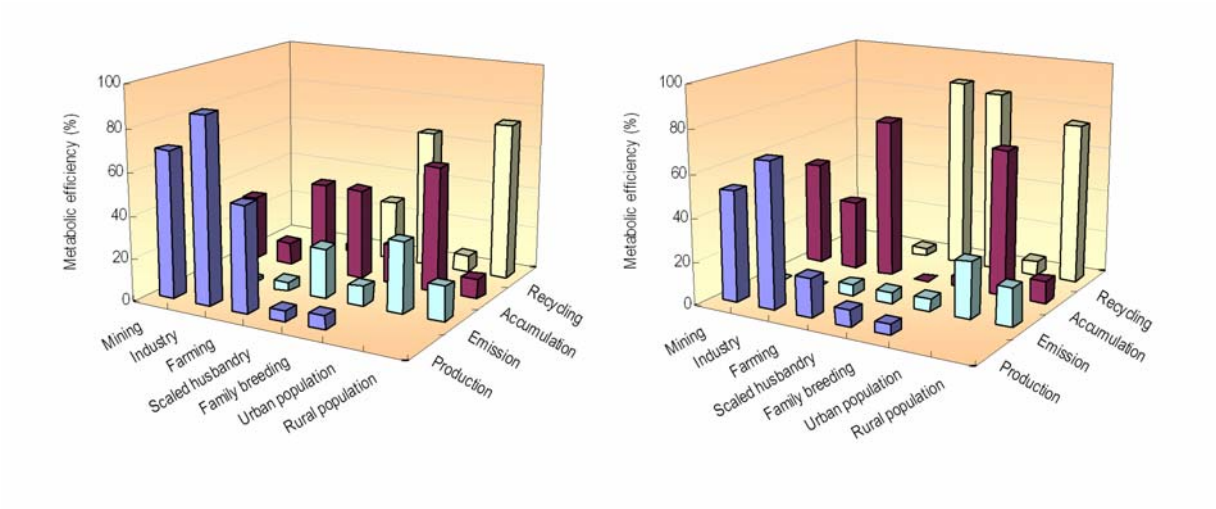

Figure 5.5 Sectoral metabolic efficiencies in China (left) and in Dianchi Basin (right) in 2000

With domestic production and import, the gross quantity of synthetical fertilizers consumed by agricultural sector added up to 4348 mkt P in 2000, accounting for 73% of DMI. While the input of inorganic and organic P to farmland achieved 47 kg P/ha, over one third of P, roughly equal to the amount of manure application, was immobilized in cultivated soils. Because of various P losses and reuse of crop residues, only 2995 mkt P was ultimately transferred to food, feed, seed and primary materials of industrial manufacture. Consequently, the (net) production efficiency and accumulation efficiency accounted for 50% and 43%, respectively, while agricultural runoff and residues reuse were marginal in quantity, as shown in Figure 5.5.

The feed structure, manure reuse and waste emission of intensive feedlots considerably differ from those of dispersed family farms. Based on the PHOSFLOW model, the intensive and dispersed livestock consumed 1214 mkt P and 1067 mkt P in 2000, accounting for 25% and

[33] The main component of phosphor-gypsum is calcium sulphate (main compound of natural gypsum), accounting for 92% - 96% of gross weight. The phosphor-gypsum is generated through the processing of phosphate rocks into phosphoric acid. In principle, the component of the waste is recoverable and could be used for production of sulfuric acid and building materials. However, reuse of phosphor-gypsum is prohibited in many countries because of the presence of trace radio-active elements (Xi 1990; Ayres and Ayres 1996; Smil 2000).

18% of DMI, respectively. The production efficiencies, indicating the nutrient conversion ratios of P, accounted for only 5% and 6%, respectively. As a consequence, a huge amount of animal waste was generated by animal husbandry as a whole, of which 717 mkt P was dumped in the countryside, 1044 mkt P was recycled to farmland and 387 mkt P was discharged into surrounding waterbodies. The intensive feedlots and farms contributed 74%, 72% and 33% to these P flows, respectively, which significantly differed from the P configurations of family farms. Measured by the efficiencies of P use, the emission, accumulation and recycling of P accounted for 24%, 43% and 28% of the input to the intensive feedlots, while the corresponding figures of family farms were much better as shown in Figure 5.5. The results, parallel to the discussion in Chapter 3, suggested that the intensive feedlots had more substantial environmental impacts.

In the year of 2000, the Chinese population consumed 1502 mkt P, consisting of 1401 mkt P in food and 101 mkt P in detergents which accounted for 23% and 2% of national DMI respectively. Whereas recycling of urban waste in forms of human manure and composted garbage accounted for only 8% of the P consumption (as shown in Figure 5.5), a majority of the urban waste was directed to environmental sinks and contributed to high accumulation efficiency and emission efficiency, i.e. 59% and 33%, respectively. In contrast, 74% of the consumed P in rural areas was subject to recycling, of which 58% was applied as manure for agricultural soils and 42% was reused as feed for breeding animals. Consequently, the accumulation efficiency and emission efficiency accounted for 9% and 16%.

The sectoral efficiencies of P throughput in Dianchi Basin considerable differed from the national configurations. Regarding local chemical industries, the production efficiency was 68% in 2000, 10 per cent lower than the national average. Notably, most of the P waste accumulated on natural soils, accounting for 32% of the P input and threefold over national average, instead of leaked into the lake. The low emission efficiency of P, as shown in Figure 5.5, was mainly attributed to a successful local industrial environmental regulation (Xu 1996; Wang 1998; Zhang 2001).

Crop farming in Dianchi Basin was remarkably materialized in term of P use. The application of chemical fertilizers reached 105 kg P/ha in 2000, three times the national average. Together with extensive use of organic manure, gross fertilizing intensity increased to 165 kg P/ha, 250% higher than the agricultural application at the national level. Not surprisingly, the accumulated P utilization of plants was only 29%, half of the national average. Consequently, compared to the national average, the local farming sector was less efficient in P use, as the production, accumulation and emission efficiencies accounted for 18%, 74% and 5% in 2000, respectively. The recycling efficiency was roughly equal to its national counterpart.

Total 552 mt P and 1927 mt P in various forms of feeds formed input for the local sectors of intensive animal husbandry and dispersive family breeding in 2000, respectively. A huge demand for organic fertilizers of the local farming practices, in particular the cultivation of flowers and vegetables, made that the majority of animal manure generated from both intensive feedlots and family farms was applied on farmland. This then contributed to a better environmental performance of the local animal husbandry. The recycling efficiency of the intensive livestock production in Dianchi Basin, among others, reached 88%, three times the national average.

As indicated in Figure 5.5, most physical features of P efficiencies in relation with urban and rural household consumption were relatively similar at the national and local level, except for the urban P emission. In Dianchi Basin, the rates of urban wastewater treatment and P removal from wastewater reached 62% and 74% in 2000, respectively, roughly for a factor two and three of the national average. This contributed to relative low emission efficiency at the local level. The not-removed P in a quantity of 138 mt, together with the uncollected urban wastewater and the dispersive sources of rural households, contributed to the eutrophication of Dianchi Lake.

5.6 Discussion on the Changes in P Loads to Water

It is interesting to compare the two new models with their original counterparts to track how data outcomes have changed as well as to analyze the trajectory of development of P flows. Taking our concerned environmental issue, i.e. eutrophication of surface waterbody into consideration, we discuss the variety of pollution loads from various sources.

Table 5.1 Comparison of multi-sources loads to surface water in China in 2000

Version	Industry	Agriculture	Composting	Livestock	Detergents	Human source	Garbage	Total
P loads in quantity (unit: mkt)								
PHOSFLOW 1.0	117	164	90	602	105	58	n.a.	1136
PHOSFLOW 3.0	80	259	0	387	98	198	53	1075
Changes in loads (unit: %)								
Data-induced	13%	58%	0	-36%	-4%	-26%	-	-11%
Structure-induced	-45%	0	-100%	0	-3%	267%	-	6%
Change in total	-32%	58%	-100%	-36%	-7%	241%	-	-5%

Notes: n.a. refers to no accounting in the PHOSFLOW 1.0 model.

As presented in Table 5.1, the contributions of various sources vary significantly between the two national models, but the aggregate influence of their differences brings forward a slight shift in the total P loss. Regardless of time leap from the year 1996 to 2000, the fact that industrial emissions decrease by 32% is due to the removal of food industry which accounts for 45% of industrial load in the early model, and purification of emission coefficients according to production chain of phosphate industries. Second, the increase of the percentage of agricultural runoff to the total load attributes to the switch from a single leaching coefficient to an elaborate estimation which takes fertilizing intensity, nutrient ratio and manure application into account. Third, the use of a top-down approach leads to a reduction of wastes leakage from livestock husbandry by which the total yield and the generation structure of animal waste change. Keeping emission efficiencies of animal husbandry unchanged, this accordingly results in an increase in pollution/emission. For the same reason, the quantity of human excreta becomes a food-induced variable, which subsequently influences nutrient loss of anthropogenic sources by a net percentage of 26%. In contrast, the introductions of directly discharged wastewater in urban areas and of nutrient loss from dumped human excreta in rural areas lead to a significant increase in pollution loads from human sources. Forth, while determination of detergents-induced emission follows the same assumption with the former model, introduction of domestic wastewater flows into treatment plants alleviates the environmental burden. Finally, whereas the removal of composting process and the supplement of domestic garbage accounting in our new model contribute to the validity and completeness of the physical structure of national P flows, they are not influential regarding freshwater eutrophication.

While the fundamental structure of the regional PHOSFAD model doesn't change as much as the renovation of national P framework, a considerable shift takes place in terms of pollution load to water environment. As presented in Table 5.2, only the emissions of mining and phosphate industries remain constant, since no new available data contribute to further improvement. The introduction of spatial diversity of corps variety and fertilizing intensity leads to an increase in agricultural runoff. As loss of solid residues keeps the same value compared to the earlier version, the total load from agriculture sources varies in a positive and linear correlation with nutrient leaching from cultivated soils. The loss of animal wastes decreases because of the adoption of SEPA's indicators instead of what we developed in the early version. This leads to 42% reduction in P emission from the sectors of animal husbandry. Together with the influence of supplement of the solid waste flow from family farms to natural soils, the total load of animal husbandry decreases by 65%. The adjustment of population poses an impact on the P emissions from detergents, human excreta and domestic garbage. Only the P load caused by detergent consumption varies linearly to the change in population, i.e. 48% less than its accounted value in the earlier model. In contrast, because a couple of uncontrolled waste flows are newly supplemented as the outflow paths of human excreta and domestic garbage, the changes in the P loads from human excreta and domestic garbage are mixed outcomes. The directly discharged wastewater in urban areas is the main factor which leads to a significant increase in the P load from human sources. Together with the supplement of the P flow related to dumped excreta in rural area, structure-induced changes in P load accounted for 145% compared to our original estimate, while data-induced change was only responsible for -2%. In terms of garbage disposal, dumping in the countryside and villages is introduced, which leads the structure-induced change of -42%. All those modifications contribute to the ultimate varieties in pollution loads to the Dianchi Lake as presented in Table 5.2. To summarize, the increase of nutrient losses from farming activity and human sources are compensated by the decrease of quantities of detergents and garbage. The total load thereby reduces to 1181 mt P, 30% less than that of the former version.

It is apparent that our refinement leads to significant shifts regarding model output of pollution loads. Not surprisingly, since P use relates to almost all social and economic aspects, reliability of the models depends on to what extend involved flows have been monitored and recorded, and to what extend those data can be extracted from existing statistics as well as literature references. No doubt the models PHOSFLOW 3.0 and PHOSFAD 2.2 are more reliable constructions of reality, compared to their earlier counterparts because of our continuous pursuit of data refinement as well as reflection on the physical relation of P flows and economic activities. However, since data gap still exists and our understanding on the P regime needs further improvement, continuous modification of these two models is desired.

Table 5.2 Comparison of multi-sources loads to the Dianchi Lake in 2000

Version	Mining	Industry	Agriculture	Livestock	Detergents	Human source	Garbage	Total
P loads in quantity (unit: mt)								
PHOSFAD 1.1	2	28	219	392	490	146	416	1693
PHOSFAD 2.2	2	28	367	136	254	355	39	1181
Changes in loads (unit: %)								
Data-induced	0	0	68%	-42%	-48%	-2%	-49%	-27%
Structure-induced	0	0	0	-23%	0	145%	-42%	-3%
Change in total	0	0	68%	-65%	-48%	143%	-91%	-30%

Nevertheless, it is very important to keep in mind that models by all means are no more than intelligent reflections of the real world. This implies, on the one hand, that the overall modeling procedure is inevitably accompanied by various uncertainties, arising from data unreliability, data processing, parameter selection, structure formation (e.g., Beck 1987), to the subjective uncertainties constructed by various individuals' understandings on reality (Hogenboom et al. 2000). This accordingly leads to a doubt of models' reliability. However, we should not be frustrated. The principal distinction between reality and man-made data and models justify our continuing search for elaboration of models and for ongoing knowledge improvement. Critics and pessimists are of course right in stating that models should never be confused with reality and we cannot even know how far we are with models from reality. The underlying proposition against these critics and pessimists is that, although models can never be validated with respect to questions how long (or short) the distance is between them and the real world (Konikow and Bredehoeft 1992; Oreskes et al. 1994), modeling approaches are and remain an indispensably strategy to acquire and become closer to reality through ongoing progress of scientific knowledge (Quade 1997). In that sense, compared to traditional techniques for accounting pollution loads, a P flow analysis approach has the advantage that provides systematic insights for developing an effective and efficient eutrophication control strategy, which we will discuss in the following chapters.

5.7 Conclusions: Setting an Agenda for Policy Evaluation

Departing from analyzing and describing the physical configurations, we will turn our focus to the environmental regulation to inquire how current environmental policies have contributed to the physical profile of societal P throughput. With that we aim to assess to what extent current environmental policies have contributed to an environmental improvement of surface water in China, at both the national and local levels. However, as P uses are connected to many – if not all – production and consumption activities within economic societies, it is nearly impossible to examine all potential relevant policies in relation to physical P flows one by one. Therefore, there is a need to discriminate which societal activities associated with P flows are environmentally crucial and thus worthy to be addressed for policy evaluation. In this sense, the SFA approach shows its valuable competence, as it analytically tracks environmental problems related to their social and economic origins from a physical point of view. Moreover, the discussions on the aggregate structure and sectoral efficiencies facilitate a systematic understanding of the causal relations among P production (consumption), recycling and emission. This suggests that a holistic perspective is desired for evaluating environmental policies, although only the P leakage does matter waterbodies directly. As such it can generate a comprehensive insight of the policy-induced changes of P flows, rather than only evaluating the efficiency and effectiveness of the end-of-pipe measures.

Focusing on our core environmental issue – rich fertilization of P in surface water – the static SFA models revealed that crop farming, animal husbandry, human excreta and detergents consumption are dominant contributors to the total P emissions. As shown in Figure 5.6, these four sources are together responsible for 88% and 94% of the national and the local total P load to the waterbodies, respectively.

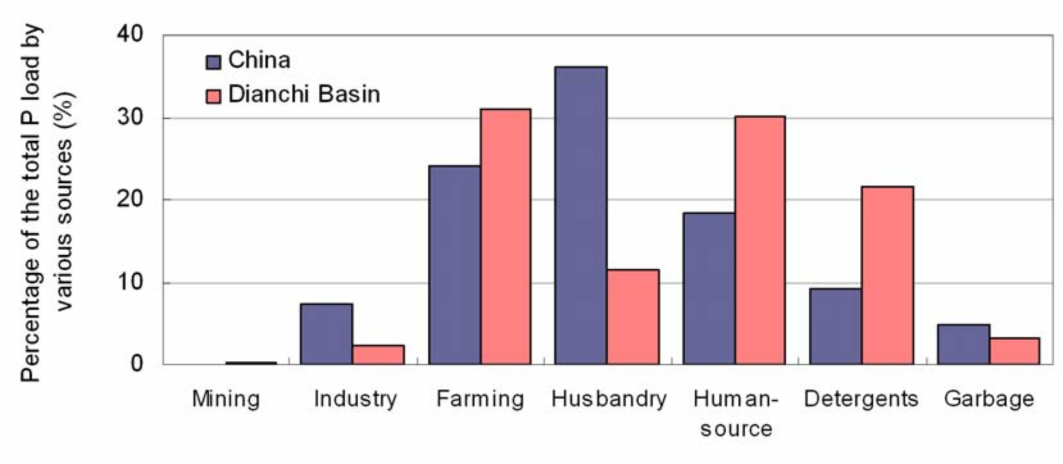

Figure 5.6 A comparison of multi-sources P loads

Instead of a complete examination of all environmental policies employed in forming and directing the societal P flows, this study focuses on regulation of three typical social practices: intensive livestock regulation, urban centralized sewage strategy and P-containing detergents ban. Together they represent the main strategies of P control in modern economies, respectively: the enforcement of productive process, the removal of P from bulk waste flows and the pursuit of green consumption.

The physical characteristics of the intensive livestock husbandry considerably differ from those of dispersive breeding activities, with respect to the disposal and discharge of P in forms of bulk animal waste and wastewater. While the intensive feedlots and their caused environmental problems have received increasing attentions in China, the family farms have not been confronted with environmental regulation yet. These fundamental differences justify the narrowing down of our attention to the large-scale and industrialized feedlots where thousands of livestock and poultries are kept. As a huge amount of un-reused P significantly undermines the sectoral P productivity and could also threatens surrounding waterbodies by P leaching, manure application should be one of the central concerns of the environmental regulations at both the national and local level.

Compared to the dominant agricultural application of human manure in rural areas, the societal arrangement of the P contained urban excreta, called centralized P control strategy, has distinct traits. This approach, aiming to transform or transfer P in order to reduce P escaping in the end of pipes, can by no means be regarded a clean option, as it results in severe secondary environmental problems. Therefore, the environmental policies in conjunction with the construction and operation of urban sewage infrastructure consisting of sewer networks and a number of centralized treatment facilities are worthy of evaluation.

Besides the above mentioned end-of-pipe technical solution that is responsible for a partly removal of the consumed P-detergent, there is one other societal reaction. Because the functions of P substance as one of the key detergent additives could be substituted by other chemicals, instruments are introduced to limit or prohibit P-containing detergents production and consumption of both urban and rural residents. As detergent-P contributes to a considerable percentage of the total P emission from households, a source-control strategy, advocated by Chinese governments including the local administration at Dianchi Basin, is expected to benefit the eutrophication control. The P-detergents bans are regarded as one of the most important P-based measures, and are thus included in our agenda for environmental policy evaluation.

In addition, this study ignores crop farming for several reasons. First, the evaluation of environmental policies targeted at this source is a time consuming process. This could be particular the case for analyzing agri-environmental policies, because a large number of agronomic factors, involving farmland use, planting styles and traditions, landscape, crops varieties, cultivated soils and fertilization application, determine agricultural runoff in a certain way and hence should be taken into consideration (Cestti et al. 2003; Lee and Jones-Lee 2001). Probably, an in-depth evaluation of a large amount of the current Chinese agri-environmental policies would lead to an overlook of other regulatory domains. Second, the regulations on agricultural runoff and livestock emission belong to the same category of environmental regulation that focuses on productive process. Hence, selecting one of them would not undermine the overall quality of the evaluation. Third, compared to the long-term agri-environmental practices, the intensive livestock regulation has to a large extent been absent in China until the start of this century. Undoubtedly, and one could argue that it is more urgent to test this new regulation on intensive livestock husbandry. Fourth, there exist many sophisticated discussions on Chinese agri-environmental policies, although they are not formalized, standardized and brought together into an evaluation methodology (e.g. Li et al. 1997; Lindert 1999; WB 1999; Zeng 1999; OECD 2002; USDA 2002; Quan and Yan 2002; Rozelle and Sumner 2003; Zhang 2004; Zhang et al. 2004).

CHAPTER 6

Evaluation of P-related Environmental Regulation in China

6.1 Introduction

In this chapter, we will respond to our second research question – to what extent do existing policies alleviate and remedy the increasing P pollution of freshwater in China. Following the evaluation agenda based on the analysis of physical characteristics of P flows, we will concentrate on the current Chinese environmental regulation of the P flows associated with the selected sectors of production, consumption and waste treatment. Both the national and local environmental policies related to P pollution by intensive animal husbandry, control of P emissions via urban human sources and P detergents ban, will be evaluated by applying the EEA evaluation framework (EEA 2001).

Focusing on the research issue of eutrophication control in China, the requirement for domestic clean water comprises a substantial need of the society. Regarding the societal metabolism of P and its dominant influences on eutrophication in China as discussed elsewhere, this societal need results in policies that aim to ecologize P flows to improve and maintain surface water quality. The criterion of policy's relevance then can be evaluated by analyzing policy objectives whether they aim at problem solving. To reduce P loads to surface water, besides other remedial motivation (e.g. dilution of P concentration in lakes, excavation of lake sediments), is a common goal of the three environmental policies. As these policies focus on the P flows that are characterized by different physical profiles and social and economic attributes, their concrete objectives differ from reducing P use at source, through enhancing P recycling, to removing P at the end of pipe. While all these policies are relevant for curbing eutrophication, their contributions to the ecological restructuring of societal P metabolism may significantly vary. This is also taken into consideration for evaluating the relevance of these environmental policies.

The costs arise during the whole process of policy design and implementation. As the costs of policy design are barely recorded in Chinese current environmental statistics and scientific studies, only those related to policy implementation are taken into account. Besides the inputs of administrative and human resources, the capital expenditures on urban sewage infrastructure, as requested by a centralized wastewater treatment strategy, will be given a focused analysis.

The ultimate environmental impact on water quality, induced by a shift of human behavior or environmental performance of target groups, is difficult to be illustrated for two reasons. First, while an expected or unexpected change of the eutrophication state is a hybrid result of many policy measures, it is difficult to track the contribution of each policy via current environmental monitoring data. Second, a causal linkage between a reduction of P load and its induced environmental effect on surface water remains unclear. As eutrophication is caused by many meteorological, hydrological and geographical factors as well as nutrients enrichment, a reduction of P load cannot often lead to a direct environmental consequence measured by eutrophication indicators (concentrations of algae, chlorophyll and P in surface waterbody). Some even argued that the load reduction cannot become a sufficient indicator for improvement of water quality (see section 6.4.1). Thus, it may not be possible to quantitatively evaluate the ultimate environmental impact of these policies on the environment. In this case, an evaluation on immediate outputs and outcomes, as proposed by the EEA framework, is regarded as a rough proxy for evaluating the impacts on environment. This study will then apply this alternative method.

By applying this evaluation framework, we attempt to address questions of relevance, cost-effectiveness, effectiveness and welfare in relation to policy design and implementation. The evaluation of national environmental policies relies on statistical data and literatures. The fieldworks at Dianchi Basin enable us to use quantitative and qualitative evidences to evaluate how successful local governments are in respond to national legislation and the varieties of policy measures between national and local regimes.

6.2 Regulation of P Emission from Intensive Livestock

Since the last two decades Chinese animal husbandry has been rapidly industrialized, as characterized by the structural shift from an enormous number of small and dispersed family farms to a limited number of intensified and concentrated feedlots. As discussed in previous chapters, this substantial transformation and the parallel growth of livestock production has considerably aggravated ecological deterioration of surface freshwater. Animal wastes generated by industrialized livestock production are rather difficult to handle appropriate and in turn are more likely to threaten the environment. In contrast, conventional family breeding has maintained a close relationship with farmland by manure application and has imposed less environmental impacts. Based on the PHOSFLOW model, animal husbandry was responsible for 36% of the total P load on waterbodies in China in 2000. Intensive feedlots contributed for 74% to this. By contrast, the results of the PHOSFAD model showed that intensive livestock production impose relative minor environmental impacts on Dianchi Lake. This was due to less intensive livestock production and the prevalence of manure reuse, instead of successful environmental regulation. In this section, we will focus on the environmental policies with respect to the national and local intensive livestock production, respectively.

Because the first nationwide regulatory framework in China was established in 2001 and was implemented in 2003, we will separate our discussion into three independent parts. First, following a brief overview of the development of animal husbandry and governmental supports, especially with respect to the "Project Vegetable Basket" in China, the environmental performance of intensive feedlots will be given close attention. This can be regarded as the state-of-the-art before the implementation of the national regulatory framework. In the second part, we will concentrate on the newly implemented regulatory framework for animal husbandry. Because no available data can be linked to the costs and actual environmental effects of this regulation and its implementation, we will emphasize on the policy objectives and potential outcomes and impacts in order to answer the EEA proposed evaluative questions of relevance and effectiveness, from an *ex ante* perspective. Finally, we will evaluate the environmental regulation in the case of Dianchi Basin, in parallel with the implementation of the national policy framework. The local manure network, responsible for the large amount of P recycling, will be analyzed as a special issue in the last section.

6.2.1 Economic and environmental profiles of animal husbandry in China

Livestock and poultry husbandry has played an increasingly important role in China's agricultural sector. With an annual growth rate of 16% since the 1980s (calculated at current prices), output value of livestock production amounted to 739 billion Yuan RMB and accounted for 30% of agricultural gross value in 2000. Yields of meat, eggs and milk have increased annually by 9%, 13% and 13% in absolute quantity during the last two decades, respectively (see Figure 3.4). This growth has also significantly contributed to the development of the rural economy, by providing employment for 80 million rural labors and contributing to 15% – even 30% in some areas – of the net income of rural households in 2000.

The rapid development of animal husbandry in China has followed a persistent governmental promotion. Aiming at accelerating a structural change of agricultural production and enhancing farmers' income, China's government has introduced a series of policies in order to stimulate the development of animal husbandry industry in the last two decades. The "Project Vegetable Basket" has been regarded as the most influential action (Li, Shan and Xu 2002; Jin and Wang 2002). This project was introduced by the Ministry of Agriculture (MOA) in 1988, with the overriding objective to develop the production and distribution of agricultural and livestock products in large and medium sized cities across China. By 2000, a number of measures have been implemented under this project (CG, DESE and ERM 2004). The main contents include: 1) establishing a project Chief Executive in each city, to guide implementation and ensure progress; 2) intensifying the production bases for vegetables, meat, eggs, milk and fisheries in each location, to increase the volume and variety of food supplies; 3) strengthening the infrastructures in relation to breeding, feed, disease prevention, processing, refrigeration and storage; 4) accelerating the extension of the latest agricultural research and technology to the farms; 5) enhancing production levels and self-sufficiency in target cities for a basic range of "vegetable basket" commodities; and 6) developing a range of economic incentives at provincial and municipal levels to support intensive livestock breeding, including preferential land, loan and tax.

With these strong political, administrative, financial and technical aids, the implementation of "Project Vegetable Basket" has rapidly and greatly stimulated the structural change in China's animal production. According to the national investigation conducted by the State Environmental Protection Administration (SEPA)[34], the output of hogs, meat chicken (broilers) and egg chicken (layers) produced by intensive feedlots and farms accounted for 23%, 48% and 44% of national total in heads in 1999, respectively (SEPA 2002). Because of the regional varieties in demand for livestock products as well as the differentiation of overall economic development, geographical distribution of these intensive livestock factories is uneven across the country. 52% and 41% of them were located in coastal areas in East China and areas with high population density in Middle China, respectively. Only 7% can be found in West China in 1999. At the provincial level, Beijing, Shanghai and Guangdong were the three leading regions, measured by the percentage of intensive livestock production to the national total output of animal husbandry sector. And Guangdong, Shandong, Henan, Hebei, Hunan, Liaoning, and Jilin were together responsible for 50% of the national output of intensive feedlots in 1999.

Table 6.1 Environmental performance of intensive feedlots in China (unit: %)

Items		Feedlots						
		Hog	Meat chicken	Egg chicken	Beef cattle	Dairy cow	Duck	Goose
Compliance of environ. regulation	Environ. permit (EP)	8.9	2.0	2.3	7.0	13.0	1.1	3.2
	Environ. impact assessment (EIA)	6.4	1.2	1.5	4.3	9.1	0.7	2.8
Equipment of environ. facilities	Separation of wastes and wastewater	22.6	20.9	13.3	19.8	23.9	25.8	25.0
	Waste disposal	11.9	2.0	3.2	7.2	10.1	5.3	2.4
	Wastewater treatment	16.9	1.4	1.8	6.4	10.8	5.2	4.4

Sources: cited from SEPA (2002).

It is also apparent that the governmental promotion for animal husbandry mainly focuses on economic development and environmental measures have been ignored to a large degree (see further discussion in next section). Not surprisingly, the environmental performances of the intensive feedlots and farms are weakened. First, a large amount of intensive feedlots did not comply with the national fundamental environmental regulations that have confronted industrial factories.[35] As shown in Table 6.1, less than 10% of all feedlots, except for dairy

[34] The investigation is the largest and latest national census after the establishment of the People's Republic of China. It officially provides fundamental information on the nationwide development and environmental performance of large scale animal husbandry, covering 32564 large scale feedlots in 23 provinces (including autonomous regions and municipalities directly under the Central Government). This investigation was conducted between November 2000 – March 2001 and the main results ware published in 2002 in Chinese. All data refers to the year of 1999. Intensive feedlots or farms were defined as those that hold 200 and more pigs, 40 and more dairy cows, 80 and more beef cattle, 2000 and more chickens, 1000 and more ducks, or 1000 and more gooses.

[35] A broad constellation of policies and regulations has been developed since the "First National Conference for Environmental Protection" in 1973, a start-point of China's efforts on environmental protection over thirty years till up present. Among others, eight approaches, entitled as China's basic regulatory instruments, have initialized the national

cow farms, had environmental permit issued by local EPBs or had carried out environmental impact assessment in the early stage of construction, based on the results of the latest national investigation (SEPA 2002). Environmental facilities, which co-determine environmental performance of these intensive livestock factories, were failed to meet basic requirements for effective pollution control. Roughly, more than 80% of these feedlots lacked necessary environmental infrastructures and more than 70% of them were even not equipped yet with facilities to separate solid wastes from wastewater, as shown in Table 6.1. Similar evidence, based on observations at the local level, is found in a number of studies. According to the World Bank (2001), for instance, only 10% of 94 intensive feedlots were levied by pollution fees, and 60% of 66 intensive feedlots had not implemented any pollution control measure.

Table 6.2 Amount of intensive livestock per unit available farmland in China (unit: heads/ha)

Region	Hogs		Dairy cows		Egg chickens	
	Range	Average	Range	Average	Range	Average
South China	600 – 2143	1071	30 – 214	94	25000 – 28125	28125
North China	150 – 183	160	3 – 7	5	4327 – 7031	5921
National average	167 – 366	227	7 – 79	10	7759 – 20455	15000
Desired carrying capacity	10 – 13		3 – 4		129 – 172	

Sources: own calculation based on SEPA (2002).

Notes: carrying capacity refers to the animal amount per unit cultivated area, which will not lead to undesired ecological deterioration with respect to the environmental qualities of soil, ground water, air and surface water. The figures of desired carrying capacity are equivalent to 30 – 40 kg P/ha, and are calculated based on related empirical studies in China (Shen, Wang and Yuan 1994; Lu, Shen and Wang 1994) and regulatory directives of developed countries and regions (OCED 1989; Hotte, van der Vlies and Hafkamp 1995; Jiang et al. 1995; De Clercq et al. 2001; Baldock, Dwyer and Vinas 2002). To convert nutrient regulation into limitation of animal heads, the indicators of breeding period and P emission load of animals are applied. The breeding periods are 199 days, 365 days and 210 days, and the P emission loads are 1.7 kg/head·yr, 10.07 kg/head·yr and 0.135 kg/head·yr for pigs, dairy cows and egg chicken, respectively.

Second, a large number of intensive feedlots appeared in suburbs and in rural areas, as a direct result of the implementation of the “Project Vegetable Basket”. However, without deliberate location planning, most of them were inclined to be placed as close as possible to large and medium sized cities, in order to reduce costs of marketing and transportation. About 80% of the intensive feedlots concentrated around large cities, including Beijing, Shanghai, Tianjin, Guangzhou, Hangzhou (SEPA 2002). Moreover, some of these intensive livestock feedlots, originally built up in suburbs or rural areas, were included in the cities following urban expansion. Due to China’s scarcity in land resources and existing land tenure, the distribution of intensive feedlots consequently has magnified undesired environmental damage by

fundamental framework of environmental regulation. These approaches involve: environmental responsibility system; quantitative examination of the comprehensive improvement of urban environments; centralized pollution control; mandatory pollution control; pollution discharge permits; environmental impact assessment; three simultaneity (san tongshi); and pollution levy. For sophisticated discussions on China’s fundamental framework of environmental regulation, see Vermeer (1998); Ma and Ortolana (2000); Zhang (2002); and Zhang and Chen (2003).

enhancing the nutrient gap between animal husbandry and crop farming. There were hardly any farmland close to these intensive feedlots and thus the potential for nutrient recycling, by which the huge amount of animal waste can be applied as manure, was undermined. Divided by the area of available farmland around intensive feedlots, the average breeding densities of hogs, dairy cows and egg chicken in China dramatically exceeded the desired carrying capacities of neighboring lands, as indicated in Table 6.2. It is also apparent that animal husbandry was far more intensive in South China, measured by the available farmland, than that in the North, while both regions bore too many animals.

Third, the geographical concentration of the intensified animal production raised another severe ecological issue. Located too close to residential communities and surface freshwater, the intensive breeding feedlots were more likely to lead to direct and severe damages to human health and environment. It has been argued that the minimum distance between intensive feedlots and sensitive objects (environment and residential community) should be 2 km – 6 km depending on animal species and animal heads, in order to achieve a sound reduction of environmental impacts, (Li 2002; Li, Shan and Xu 2002; Jin and Wang 2002; CG, DESE and ERM 2004). However, according to the results of SEPA's investigation, 8% – 10% of the intensive feedlots were sited less than 50 meters away from sources of local drinking water; and 25% – 40% of them fell within a 150-meter range from residential communities or water supply sources (SEPA 2002).

Based on the discussions above, it can be concluded that in general the uncontrolled P emission from intensive animal husbandry in China has escalated parallel to the gradual growth in total animal heads and the rapid shift in breeding structure. This in turn leads to an increase of environmental deterioration, particularly in the localities where economic activities and population are more intensive. The impacts on water eutrophication with respect to the environmental performance of intensive livestock feedlots are characterized in two ways. On the one hand, internal environmental management of the industrialized breeding factories is generally poor due to the weak compliance with the existing governmental regulations. This results in direct release of P load to surrounding ambient and residential communities. On the other hand, the risk of P leaching from the intensive feedlots dramatically increased and affected water quality in line with the process of spatial concentration of animals that disturbed the nutrient balance between livestock and crop operations in local areas. In particular, the conflict between the intensive breeding activities and local environment has been increasingly reported in a number of empirical studies at Beijing (Zhang 1996), Shanghai (Xu 2001; Jiang and Gu 2002), Guangzhou (Ding 2000), Jiangsu Province (Wu 2001), and the Yangtze Delta (Liu et al. 2002).

6.2.2 The national regulatory framework: an *ex ante* evaluation

As one of the most intensive regions employ in livestock production in the world (Smil 2000; Gerber et al. 2005), China has been in long-term absence of effective environmental regulation on animal husbandry. Neither the Law of the People's Republic of China on

Prevention and Control of Water Pollution[36] nor the Environmental Protection Law of the People's Republic of China[37], which are leading legislative documents in relation with environmental regulation, refers to the concept of animal waste and articulates specific and feasible regulatory measures on it. The lack of legitimation for pursuing sound environmental performance of intensive livestock production in the national juridical system is partially responsible for a immaturity of administrative regulation by related governmental agencies, involving MOA responsible for overall affairs of the development of animal husbandry industry, and SEPA. It is notified that administrative jurisdiction of livestock-related environmental regulation has not been specifically committed to MOA or SEPA until the 1998 Administrative Reform. This means that environmental concerns on intensive livestock production were hardly institutionally embedded, which in turn affected the inputs and outputs of intensive livestock environmental regulation severely. Although MOA and SEPA proposed some instructions recently, involving the "Circular on Accelerating Construction of Energy and Environmental Engineering in the Ninth Five Years" (reference no. *nonghuannengfa*[1996]1) issued by MOA[38] and "Instructions on Strengthening Rural Ecological Environmental Protection by SEPA" (reference no. *huanfa*[1999]247) promulgated by SEPA[39], articulation and stipulation of administrative jurisdictions and responsibilities, resources mobilization and allocation, economic incentives and public participation still remain extremely poor. Consequently, environmental achievement has been limited till the late 1990s, as discussed in the former section. At local level, likewise, environmental concerns for intensive livestock production were out of sight of local governments. Only Shanghai Municipality recently constituted a livestock-related regulatory framework, consisting of "Implementation Scheme for Integrated Pollution Control for Animal Waste in 1999-2002 in Shanghai" and "Technical Plan for Integrated Pollution Control and Utilization of Animal Waste in Shanghai (1999-2005)". In summary, environmental objectives and policy measures for intensive feedlots have not been integrated into national and local legislations and administrations and this omission of effective regulatory measures has not been significantly changed yet until the year of 2001.

In 2001 SEPA initialized a national regulation framework through three innovative documents on environmental administration, technical guidance and discharge standard, in line with the transfer of administrative responsibilities. As abstracted in Table 6.3, this new policy has

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[36] Adopted on May 11, 1984, and amended on May 15, 1996.

[37] Adopted at the 11th Meeting of the Standing Committee of the Seventh National People's Congress on December 26, 1989, promulgated by Order No. 22 of the President of the People's Republic of China on December 26, 1989, and effective on the date of promulgation.

[38] The MOA's Circular (1996) required local agricultural departments must enforce those intensive feedlots and farms (holding animal stocks over one hundred heads of cattle, one thousand heads of hogs, or ten thousands heads of poultries) without environmental facilities to undertake necessary control measures and improve their environmental performance during the Ninth Five Years; New feedlots and farms must carry out the three simultaneous practice, i.e. installations for the prevention and control of pollution at a construction project must be designed, built and commissioned together with the principal part of the project, and their environmental expenditures must exceed 10% of total investment.

[39] The SEPA's Instructions (1999) required local EPBs should supervise and promote the intensive feedlots and farms (in certain scale) to carry out the three simultaneity measure seriously, and local EPBs must enforce those intensive feedlots and farms located in national key environmental areas or around large and medium cities to build up the environmental facilities for wastewater treatment or waste utilization before the end of 2002.

adopted and institutionalized a number of environmental measures, mainly including the general ecological principle of pollution control, the requirement for more effective participation of local competent administrations, the enforcement of some key approaches employed in the Chinese fundamental framework of environmental regulation, the intervention in the spatial distribution of intensive animal husbandry, and the specification of environmental technologies and standards. Compared to the previously weak institutions, the newly developed framework has partially filled up the policy vacuum and significantly strengthened the overall capacity of environmental regulation and administrations towards intensive livestock production in China. Moreover, it offers a relatively comprehensive and feasible framework to local environmental agencies for the first time. It then largely increases the potentials for regulatory improvement and innovation at the local level.

Although ecological rationality is prioritized to serve as the regulatory principle, apparently the SEPA's policy is oriented to an end-of-pipe abatement of pollution. It stipulates maximum pollutant concentration in discharged wastewater, instead of focusing on integrated nutrient-based management on entire process of livestock production. Compared to its specific elaboration on the enforcement of technologies and infrastructures of waste storage and treatment as well as on the discharge standards, the stipulations on limitation and prevention of P emission from intensive livestock remain vague.[40] First, while the regulation generally proclaims that new constructed intensive feedlots should limit their size according to the assimilation capacity of local farmland to achieve a nutrient balance between crop farming and animal breeding, it is not addressed and detailed in the specific requirements, i.e. on which distance owners are required to seek for available farmland and how to let farmers accept manure application, compulsorily or voluntarily. More importantly, SEPA's policy has not imposed capacity limits on existing intensive feedlots, which concentrate in the areas with high population density and limited available farmland (see Table 6.2). The spatial restrictions to livestock are insufficient, because only the forbidden regions for intensive livestock production are specified but not where and to what extent intensive livestock should be restricted. Second, although SEPA's regulation has proclaimed that intensive feedlots should adopt appropriate nutrient ingredients and application of microbial and enzyme additives is encouraged in order to reduce nutrient emission at source, there is no specific clause to articulate what kind of ingredients can be regarded as environmentally sound and what decrease in nutrients emission should be achieved by improved feeding activities. Third, manure reuse is generally emphasized as the overriding approach for waste disposal, but is not enforced. The policy identified the necessary facilities for manure storage and stipulated that

[40] By contrast, large areas with enormous P surpluses in Western Europe, including the Netherlands, northwestern Germany, Brittany in French, the Italian Po valley, and the northeastern United States have established various integrated environmental regulatory framework (USDA and USEPA 1999; Delgado, Rosegrant and Steinfeld 1999; Baldock, Dwyer and Vinas 2002; USEPA 2002; EEA 2003a, 2003b). The principle underlying strategy is to minimize potential environmental impacts of intensive livestock production by promoting and maintaining the nutrient integration of livestock production and cropland or pastureland based on regional nutrient balance (GAO 1995; Jiang et al. 1995; Steinfeld, de Hann and Blackburn 1998; De Clercq et al. 2001; Ribaudo et al. 2003; Westerman and Bicudo 2005). The concept of ecological capacity or carrying capacity of land, thereby, has become a vital applied indicator for stipulating the maximum of animal heads per hectare land or per each farm to ensure environmentally safe spreading of manure (Steinfeld et al. 1998; Gollehon, Caswell and Ribaudo 2001).

the volume of manure should not exceed the annual nutrient demand by crops. However, the weak stipulations on available farmland for nutrient demand calculation and on the responsibilities and obligations between farmers and feedlot owners make a feasible and economic manure application scheme questionable. Therefore, this puzzle of SEPA's enactments is not likely to facilitate P recycling from livestock to farmland.

Table 6.3 Outline of the SEPA's regulation framework for intensive animal husbandry

Item	Description	Main contents
Management Methods for the Pollution from Livestock and Poultry Breeding	Official code: SEPA Decree No.9 Issue date: May 8, 2001 Implementation date: May 8, 2001	Recycling, reuse and reduction of animal waste are highly prioritized as the principle for pollution control Pollution investigation and impact assessment should be integrated into the environmental protection plan of country and above level governments EIA, EP, TS, PDRR, PDL and PF must be carried out [a] Intensive livestock production is forbidden in specified areas
Technical Standard of Preventing Pollution for Livestock and Poultry Breeding	Official code: HJ/T81-2001 Issue date: Dec 19, 2001 Implementation date: April 1, 2002	Prescription of basic technical requirements, covering: location selection, interior layout, cleaning techniques, waste storage and treatment, manure reuse, feeds and feeding management, disposal of dead animals, and monitoring
Discharge Standard of Pollutants for Livestock and Poultry Breeding	Official code: GB18596-2001 Issue date: Dec 28, 2001 Implementation date: Jan 1, 2003 [b]	Stipulation of daily maximum allowable concentration of water and odor pollutants, maximum allowable discharge quantity of wastewater and hygiene requirement of waste based on classification of intensive feedlots [c]

Notes: [a] EIA – environmental impact assessment; EP – environmental permits; TS – three simultaneity; PDRR – pollution discharge reporting and registration; PDF – pollutants discharge fee; PF – pollution fine.

[b] The standard went into effect for all Class I feedlots and those Class II feedlots located in key environmental protection cities, basins and *severe* pollution areas on Jan 1, 2003. At the county and above level, EPBs can set specific implementation dates for other Class II feedlots, which should be before 1 July 2004.

[c] Intensive feedlots are categorized into two classes according to animal heads in breeding. Class II feedlots refer to 500 – 3,000 heads pigs, 100 – 200 heads adult dairy cows, 200 – 400 heads beef cattle, 15,000 – 100,000 heads egg chickens, or 30,000 – 200,000 heads meat chickens. The feedlots in larger scales are then entitled as belonging to the Class I.

With respect to the rapid increase in P emission by intensive livestock production, the relevance of the policy framework is further weakened due to its limited target group. The regulation is confined itself to large scale feedlots, as described in Table 6.3. However, the definition of "intensive" is questionable, because only a small part of industrialized livestock factories have been taken into account. A significant part of medium and small scale feedlots escape the proposed jurisdiction, while they contribute with a large quota to the national output of livestock production.[41] Accordingly, this policy could become less relevant for

[41] In addition, the enormous number of family farms, responsible for 70% of national output of livestock production, has not been involved in the SEPA's regulation framework yet. Compared to the concentrated and industrialized animal husbandry, the dispersive and traditional breeding activities accounted for a relative small amount of pollution emission, based on the PHOSFLOW and PHOSFAD models. Because of the diversities of production structure and breeding style across the

areas with low livestock intensity, but high environmental sensitivity, e.g. the Dianchi Basin as discussed in the next section.

SEPA's regulation has a phased implementation schedule as presented in Table 6.3. Based on the policy enactments, only some of the intensive feedlots located in environmental sensitive or severely polluted areas are eligible to SEPA's regulation before the end of 2003. Others remained out of sight until the middle of 2004. Moreover, the ambiguous definition of "severe" pollution areas further provided potential possibilities for local governments to lag the implementation schedule, as many claim that their administrative areas have not suffered severe pollution.

The objectives of SEPA's regulation as discussed above have justified a weak policy output of proposed measures, which principally focuses on enhancement of waste treatment and disposal and is less effective in promoting source control and P recycling.[42] Besides the discharge standard (GB18596-2001), the six key measures, involving environmental impact assessment, environmental permits, three simultaneity, pollution discharge reporting and registration, pollutants discharge fee and pollution fine, are derived from the national fundamental framework of environmental regulation. They have been developed based on China's experience on industrial pollution control. These approaches, however, emphasize active administrative intervention, while market-based incentives and public participation are basically ignored. At the same time, the pollutant discharge standard is directed to a concentration control strategy, but not serving the limitation of P emission in total quantity based on regional nutrient balance. All these conventional and fragmented measures will have limited influences on the reduction of P at source and sound P recycling to farmland.

The environmental effects in regard to both outcome and impact of SEPA's regulation could be further impaired by an inherently limited administrative jurisdiction. The SEPA's strategy has been developed from an independent environmental perspective. It hardly integrates with the existing administration on livestock production, which is overwhelmingly focused on the agricultural economic growth and rural development, e.g. the "Project Vegetable Basket". While in the Chinese bureaucratic system the agricultural department is exclusively committed to the development of livestock production, the active participation of MOA and its local affiliations and the development of an efficient co-operative relationship between

large-territory country, however, family farms could impose dominant influence on local environment. Therefore, it has been argued that the SEPA's new regulation should also include appropriate measures associated with family farms (e.g. see discussion in CG, DESE and ERM 2004).

[42] Various kinds of measures, including legislation, financial mechanisms, taxation and administrative instruments, have been widely applied in advanced countries and regions. The Dutch government, for instance, has developed a holistic regulatory framework since the mid 1980s, including the production rights, manure disposal contracts, mineral accounting system and technical guidance for manure storage and application (De Haan and Van der Zee 1994; Hotte, van der Vlies and Hafkamp 1995; Neeteson 2000; Erisman et al. 2001). The production rights is to enforce tradable quotas based on the land capacities of individual farms. The manure disposal contracts aims to ensure that the amount of manure produced would not exceed the amount that can be disposed of under certain standards, and the mineral accounting system is to bring a balance between the nutrient entering and leaving at farm level. Accordingly, Dutch farmers are required to calculate the nutrient input and output of their farms during overall operation process. Any change in nutrient caused by cultivation, animal breeding, fertilizer application, processing and distribution should be monitored and further recorded in nutrient balance sheets which are to be checked periodically and occasionally. If the accounting results show that nutrient output exceeds input, farmers then will be levied by relative high tax and bear additional cost for transportation and assimilation of those manure excess.

SEPA and MOA are vital for a successful ecologization of intensive livestock production. However, the desired integrated strategy remains unclear in the current policy framework. Not surprisingly, the ideology, policy and strategy of continuously accelerating countrywide intensive livestock production, rather than promoting sustainable development or even simple pollution control, still overridingly occupies key pivots of the most important MOA's documents, such as the "Tenth National Five-Year Plan for Agriculture and Rural Economy" (2001) and the "Instructions on accelerating the development of national livestock production" (developed by MOA and transmitted by the State Council on Oct 4, 2001, reference no. *guobanfa*[2001]76).

At the same time, the new policy framework has ignored building capacity of environmental agencies themselves, in particularly at the local level. This could largely influence policy implementation. Environmental authorities have traditionally focused on issues of industrial pollution control and urban environmental management. The long-term absence of knowledge, expertise and professionals on intensive agriculture has not been improved since environmental authorities were committed to livestock-related environmental regulation by the Administrative Reform in 1998 (Li 2002; Li, Shan and Xu 2002).[43] Therefore, the development of institutional capacity is urgently needed for effective implementation of SEPA's regulation. However, disappointingly the new legal framework has paid more attention to technical aspects than to underlying institutions. The lack of specific fiscal funds and exclusive personnel authorization will significantly hamper the input of administrative resources, and thus undermine the obtaining of desired effects on target group and surface waterbody. This will be further illustrated by the case study on Dianchi Basin.

In summary, while the SEPA's policy – as a first comprehensive framework – built up a milestone in the regulation of intensive animal husbandry in China, there is still a major need for future improvement. As the policy objective is substantially oriented to an abatement of P, the relevance in evaluative term of the EEA framework is largely undermined by the shortages in its proposals of regional nutrient balance, environmentally sound feeding and feasible manure reuse, and by the limitation of target group and the bargained implementation schedule. Since the policy relies on a conventional regulatory approaches and an end-of-pipe technological regime – instead of pursuing integrated nutrient management strategies which have been broadly applied in most livestock intensive areas in Europe and the United States – the potential environmental effects on livestock-related P flows are expected to be limited. Furthermore, the implementation of SEPA's regulation could suffer from a low-level administrative priority and a weak institutional capacity.

Additionally, probably with acknowledgement of these substantial weaknesses, the State Council in response has instructed SEPA to develop a national ordinance on the basis of the Decree in 2001, as one of the priorities under the national legislative plan for 2004. So far,

[43] There might be more at stake. The Ninth National People's Congress (NPC) turned out to provoke a radical reform of government administration. When the dust had settled, the number of ministry-level bodies had been reduced from 40 to 29, and 50% of the governmental position had been listed for removal from the governmental payroll within three years. See details in Jahiel (1998). The reform seems more influential on local administrative capacity of livestock regulation, e.g. in Dianchi Basin as discussed in next section.

SEPA has completed the draft as well as its supporting documents, which is scheduled for submission to the State Council for review in late 2004. Having highest priority in administrations, this ordinance could offer great opportunities to integrate individual departmental interests and thus foster an effective mechanism for cross-departmental and integrated decision making. This could considerably boost the implementation of the SEPA's policy framework and facilitate better achievements overall.

6.2.3 P regulation on animal husbandry in Dianchi Basin

Animal husbandry is relatively less industrialized in Dianchi Basin compared to livestock intensive areas in China. The amount of intensive hogs, cattle, sheep and goats and poultries accounted for 17%, 0.2%, 7% and 49% of the total in 2000, respectively, based on the definition of "intensive" proposed by Kunming Environmental Protection Bureau (KEPB) as described in Table 6.4. According to the PHOSFAD model, the intensive livestock production was 'only' responsible for 28 tons P leaking to the lake in 2000, which mainly attributed to the huge amount of manure application.

The relative slight stress on the environment, however, does not mean a mature local environmental regulation on intensive livestock. It is surprising that none of the existing governmental documents, including the most predominant administrative law "Ordinance for Conservation of Dianchi Lake" and the recent environmental action "The Tenth Five Year Plan for Water Pollution Prevention and Control of Dianchi Lake"[44], has specified environmental requirements for intensive livestock production in Dianchi Basin. The absence of legislation and administration on intensive livestock regulation implies that local regulatory practices to a large extent rely on the individual opinion and preference of local officials, before the promulgation of the national policy. In fact, KEPB and its affiliations at county level decided which intensive feedlots are crucial polluters that need to be subject to regulation, mainly based on general knowledge and personal experiences. Moreover, due to the lack of feasible measures and capable faculties, local EPBs have imitated their experiences on industrial pollution control to intensive livestock management, but have not been engaged in pursuing possible institutional and instrumental innovations. Only a few measures, involving environmental impact assessment, three simultaneity, environmental permits and pollutants discharge fee, have been adopted by local EPBs on intensive livestock feedlots.

Consequently, local EPBs have not promoted an environmentally sound achievement. First, only 4.6% of the intensive feedlots applied for a local environmental permit and nearly none of them carried out environmental impact assessments or installed basic facilities for separating animal dung and urine at source (Wang 2004). Second, most feedlots discharged wastewater without any treatment and only few intensive feedlots were equipped with elementary facilities, such as sandy septic tanks. Third, a major part of the intensive feedlots

[44] The Ordinance was approved by the Standing Committee of Yunnan People's Congress (SCYPC) on 25 Mar 1988 and amended by SCYPC on 21 Jan 2002. The 10th five-year environmental plan was promoted by KEPB in 2001 and was formally approved by SEPA on 12 May 2003 (official document number: huanfa[2003]84).

was located around Dianchi Lake and some were even as near as 2 – 3 km to the lake (KIES 2002; Li and Lu 2004). Compared to the discussion at the national level (see Table 6.1), the environmental performance of intensive livestock feedlots in Dianchi Basin was even worse. This then illustrates that the local environmental regulation on intensive livestock, without specific formulation on environmental policies and measures, has not resulted in a reliable environmental outcome and impact.

Table 6.4 Numbers of intensive livestock and feedlots in Dianchi Basin in 2000

District / Classification	Cattle		Dairy cows	Hogs		Sheep and goats		Meat chickens		Egg chickens
	Stock	Output	Stock	Stock	Output	Stock	Output	Stock	Output	Stock
Chenggong	0	0	118	11385	14724	1986	404	115106	351621	355782
Guandu	0	28	178	9778	12861	2154	1046	1362690	1315240	867700
Jinning	99	172	852	10495	22122	628	1446	67750	173445	32110
Xishan	46	77	331	10365	13934	1364	605	60000	215000	217056
Songming	139	355	0	20	120	3064	1139	51000	51000	22700
Total	284	632	1479	42043	63761	9196	4640	1656546	2106305	1495348
Class I	0		0	0		0		0		1
Class II	0		1	13		0		2		27
< Class II	58		149	182		191		156		294
Total	58		150	195		191		158		322

Sources: KIES (2002).

Notes: Because of the long-term lack of national regulation and uneven development of animal husbandry across the country, the definition of intensive feedlots varies regionally. Based on the interpretation by KIES, all those feedlots which produced 5 heads and above cattle, 20 tons and above milk, 20 heads and above hogs, 10 heads and above sheep or goats, 2000 heads and above meat chickens, or held 500 heads and above egg chickens were taken into consideration for this investigation.

It could be risky to place hopeful expectations on local implementation of the national policy framework. Although SEPA's policy officially offers a promising formulation for intensive livestock regulation, the limited target group of intensive feedlots due to its broad definition leads to the scarce applicability for the case of Dianchi Basin. Measured by SEPA's standard, only one egg chicken feedlot met the regulatory standard of scale for Class I, and 6.7% of the hoggeries, 0.7% of the cow farms, 8.7% of the egg-chicken henneries, and 1.3% of the meat-chicken henneries reached that for Class II, as shown in Table 6.4. This implies that most livestock feedlots are small and medium sized in scale and continue to be out of control, if no feasible local standard is developed.

The underlying limitations in administrative capacity of local EPBs that are responsible for the current weakness of intensive livestock regulation, could further undermine expected environmental effects of the implementation of SEPA's policy. While the extension and aggravation of overall rural pollution impose more challenges to local EPBs, in particular to the capacity of environmental monitoring and management of environmental agencies at county level, the overall administrations have not been correspondingly strengthened. The

Jinning EPB, for instance, faces rapid growth of dairy cows breeding.[45] It saw its authorized quota of staff as well as its annual fiscal budget being reduced by one third during the 1998 Administrative Reform. Due to the shortage in administrative resources, the related regulation, thereafter, has retrogressed to one single instrument, i.e. charging on intensive feedlots by pollution fees. The other instruments were basically not applied. But, because the investment and economic profit of animal husbandry are in general lower than those of industrial enterprises, the amount of charged fees by local EPB were comparatively low and thus could hardly generate sufficient incentives to the regulator. As a consequence, the local EPB has to balance their limited inputs and expected outputs among the management on industrial factories, town and village enterprises (TVEs), agricultural runoff and livestock waste. The livestock-related regulation is barely given high priority by the local EPB and the environmental outcomes and impacts of the relevant policy measures are thus poor. There is a great need to strengthen and upgrade local capacity building, aiming to implement SEPA's policy. In addition, the development of national policy has to a certain degree promoted active local participation in intensive livestock regulation. As a direct reaction to the national regulation framework, the first comprehensive investigation on the intensive animal husbandry in Dianchi Basin was carried out in 2002 by Kunming Institute of Environmental Science (KIES), an affiliated agency of KEPB. This survey, in parallel with the first national investigation, initialized a state-of-the-art of local intensive livestock production and thus prepared for future policy implementation.

Table 6.5 Manure trading in Dianchi Basin in 2000

Market	Trade in quantity (tons/day)	Trade in value (thousands Yuan RMB/year)
Longmen Town	250 – 300	6388 – 9855
Taipingguan Village	140 – 160	3577 – 5256
Luoyang Village	15	383 – 493
Total	405 – 475	10348 – 15604

Sources: own calculation based on KIES (2002) and Li and Lu (2004).

As discussed elsewhere, the prevalent manure application was driven by the cultivation of flowers and vegetables in Chenggong County. It has bridged intensive livestock husbandry and farmland and thus has considerably alleviated P load on Dianchi Lake.[46] Since the late 1980s, local demand in organic fertilizer has rapidly increased because of the continuous deterioration of soil quality caused by intensive use of chemical fertilizers and could not maintain the production of economic crops (Wang 2002; Yan, Ma and Wang 2002). Without

[45] The Kunming government has promoted Jinning County as the production basis for dairy cows breeding since 1993. The amount of cows accordingly leapfrogged to 4027 heads in 2000, increasing nearly six times by 1993. One half cows were concentrated fed in forms of rural cooperatives, accounting for 40% of total number of intensive cows in Kunming municipality. In addition, one feedlot in designed scale of 1000 heads is the biggest one in Yunnan Province (JAD 2001).

[46] Dianchi Basin, particularly Chenggong County, is famous for flower cultivation. A governmental invested project for construction of the largest special market in Asia for flower trading was scheduled to be completed in Chonggong County in 2004. This suggests that the intensive production of flowers as well as and the demand in manure could further rise.

governmental intervention on manure use, flower growers started to seek and buy manure in nearby counties and villages, as local human excreta and animal dung could no longer meet the increasing consumption. While manure became a tradable and profitable commodity in Dianchi Basin, several hundreds native farmers, together with some outlanders, abandoned traditional agricultural production and turned to engage in manure-related business, including collection, transportation and distribution. Subsequently, three manure markets emerged at Chenggong County and trading networks covering the whole Dianchi Basin emerged in the end 1990s. Based on local studies (KIES 2002; Li and Lu 2004), the trading volume of manure added up to 405 – 475 tons/day, accounting for 15 – 18% of daily manure generation in Dianchi Basin and exceeding manure emitted from intensive feedlots. The turnover in value was estimated to reach 10.3 – 15.6 million Yuan RMB in 2000, as shown in Table 6.5. The manure network has greatly stimulated feedlot owners to pursue additional economic benefit and thus has objectively facilitated the reuse of animal waste: animal dung was separately collected before it mixed with urine and wastewater, and then sold to manure traders at an average price of 20 – 40 Yuan RMB per ton.

Lacking necessary governmental intervention, however, the exclusive manure network could not endogenously contribute to an environmentally-sound cycle of P. First, the massive manure use has been primarily forced by economic benefits of both flower growers and feedlot owners, instead of environmental concerns. This means that manure application could wither away following a decline in the flower cultivation. In fact, as the soil productivity in Chonggong County has gradually decreased, some of the farmland can hardly produce sufficient economic output to compensate overall inputs (Wang 2002). Consequently, since the late 1990s increasing amount of specialized flower growers have moved to farmland in other districts and counties, or even abandoned farming and shifted to jobs in urban areas. This in turn imposed significant influence on local application of manure. Second, the quantity of reused manure fluctuates, because the demand varies across different seasons. In fallow period and winter season, the decreased soil assimilation of manure reduces the manure quantity in trade. Accordingly, animal dung was discretionarily dumped in these off-seasons, due to the lack of necessary environmental facilities at intensive feedlots. This subsequently contributed to an increasing stress on water quality of Dianchi Lake. Third, the huge manure transportation by trucks and tractors does not entail any precautionary measures to avoid road leakage. All three trading markets lack any environmental infrastructure. The inevitable leakages of P through manure conveying, storage and loading affect the environment and the lake, as irrigation and drainage canals and ditches surrounding these markets finally discharge in the lake. In short, it is apparent that the current spontaneous manure network in Dianchi Basin cannot be regarded as a sophisticated scheme of manure reuse with respect to its environmental impacts, although it partially contributes to limit P enrichment in the lake. Therefore, there is a need for a governmental intervention through policies and legislations to create flexible and reliable nutrient relationships between crop farmers and animal breeders, aiming at ecologizing P flows by promoting sustainable nutrient recycling.

This study reveals a puzzle of intensive livestock management in Dianchi Basin. On the one hand, local regulation has been long-term absent and actual environmental effects of existing local measures have been largely weak due to the limited local administrative capacity. The national new policy framework can hardly be applied in Dianchi Basin, on the other hand, because of the irrelevance with respect to its broad definition of the 'intensive' feedlots. There is an urgent need to develop and formulate a set of locally feasible and effective policy instruments, combining national policies and local configurations.

6.3 Centralized Control for P in Urban Domestic Wastewater

The centralized wastewater treatment system emerged in the industrializing world in the nineteenth century, and was primarily implemented to enhance urban hygiene. The issue of P control has not been paid close attention until the spreading of eutrophication occurred worldwide in the 1960s, due to the gradual fertilization of surface waterbody by nutrients. With the continuing improvement of environmental technologies, P generated from human sources and detergents use is increasingly removed from wastewater at treatment plants (Morse, Lester and Perry, 1993; Stauffer 1998; Steen 1998; Moss 2000; OECD 2003). This giant infrastructure of municipal sewage networks and wastewater treatment plants has drastically reshaped the P cycle within modern economies, resulting in a series of economic and environmental consequences.

In this section, we will evaluate the Chinese centralized wastewater treatment strategy, by focusing on the EEA proposed evaluative elements and questions in relation with the developing sewage systems at both the national and local level. In addition, we will restrict our analysis to urban areas, because implementation of similar centralized strategies for rural wastewater control has not been scheduled on the Chinese governmental policy agenda yet.

6.3.1 Urban wastewater treatment and P control in China

China has experienced rapid urbanization progress since the Economic Reform in 1978, particularly during the last decade. There were in total 459 million people living in 662 cities and 17341 towns in 2000, accounting for 36.2% of national population and 1.5 times to that in 1985 (NBS 2003). Comparatively, the urban wastewater infrastructures have been considerably improved, particularly since 1995. By the end of 2000, the length of sewer pipelines extended to 142 thousands km, annually increasing by 7.4 thousands km since 1985, with 64.8% of all urban areas connected to the sewage systems in 1995 (NBS 2003; NECW 2003). In parallel, the total number of WWTPs reached 427 in 2000, among which the amount of primary and secondary treatment plants accounted for 145 and 282, respectively, increasing by 6.6 and 9.7 times from 1985, as shown in Figure 6.1 (Du, Du and Qian 2000; NECW 2003).[47]

[47] The first purification factory of urban wastewater was built up in 1984. The construction of Jizhuangzi plant, the first urban wastewater treatment plant in China with designed treatment capacity of 260 thousand tons per day, was began in 1982 and completed on 28 April 1984 in Tianjin Municipality in Northern China (NECW 2003).

Aiming at accelerating the development of sewage system to meet the increasing need in pollution control, the centralized treatment approach was upgraded as a national environmental strategy in 1991. As indicated by two official documents, i.e. the "Stipulations for Accelerating Urban Centralized Wastewater Treatment Infrastructures" (reference no. *chengjian*[1991]594) issued by Ministry of Construction (MOC) on 23 August 1991 and the "Measures for Implementation of Current Policies for Urban Wastewater Management" (reference no. *chengjian*[1991]840) issued by MOC and SEPA on 13 Dec 1991, the centralized sewage system has become one of the key urban infrastructures that was granted governmental supports and encouragement (Liu 1998).

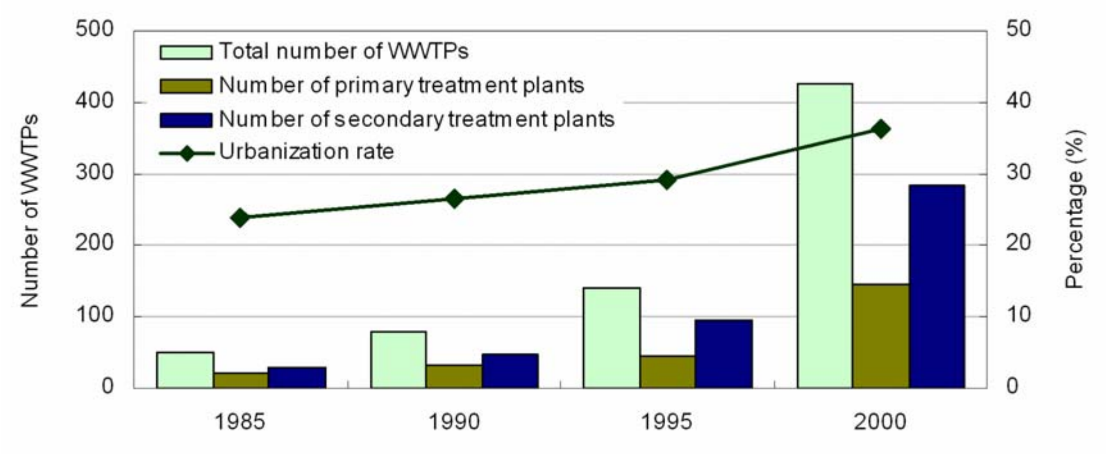

Figure 6.1 The development of urban wastewater treatment in China

Sources: based on Du, Du and Qian (2000); MOC (2001); NBS (2003); NECW (2003).

While governmental expectations on the contribution of a centralized strategy to alleviate the continuous deterioration of water environment have been high, specific goals for the promotion of urban wastewater control have not been formulated until 1996. In "The National Ninth Five-Year Plan for Environmental Protection and the Long-Term Targets for the Year 2010", SEPA proposed the first national goal: the treatment rate of urban domestic wastewater should reach 25% by 2000. Long-term goals on the treatment rate were addressed subsequently. By 2005, the rate should be on average 45% as requested by SEPA's "The Tenth Five-Year National Plan for Environmental Protection" (issued on 30 Dec 2001, reference no. *huanfa*[2001]210). And the rate at cities with over 500 thousands inhabitants should exceed 60%, as stipulated by "Circular on Urban Water Supply, Water Saving and Water Pollution Control by the State Council" (issued on 7 November 2000, reference no. *guofa*[2000]36). By the year 2010, the treatment rate should exceed 50%, 60% and 70% in towns, cities and environmentally key cities[48], respectively, as presented in "Technological Policy for Urban

[48] In 1989 SEPA started to quantitatively examine the comprehensive improvement of urban environments, one of eight key measures of China's fundamental environmental regulation system, on 32 cities. Included were 3 municipalities directly under the central government (Beijing, Tianjin and Shanghai), 26 capital cities at provincial level (excluding Lhasa), and 3 environmental sensitive cities (Dalian, Suzhou and Guilin). These 32 cities, entitled as environmental key cities under the national environmental examination, were supplemented by 5 cities in 1992 (cities specially designated in the state plan), 9

Wastewater Treatment and Pollution Prevention and Control" (reference no. *chengjian*[2000]124) issued by MOC.

Setting limits on pollutants concentration in effluent is the most important regulatory measure with respect to technology application and environmental performance of urban sewage systems. In 1996 the "Integrated Wastewater Discharge Standard" (GB 8978-1996) was introduced to restrict maximum allowable concentrations of sixty-nine pollutants in discharged wastewater for all pollution sources, excluding some special manufacturing sectors which should comply with corresponding industrial standards respectively. As municipal WWTPs were involved as potential polluters in this standard, WWTP operators should undertake necessary actions to limit the P concentration in effluent of treated wastewater less than 0.5 mg P/L or 1.0 mg P/L, depending on the environmental functions of the receiving waterbodies as specified by the Environment Quality Standard for Surface Water (GB 3838-1988). This restriction was replaced afterwards by another standard, i.e. "Discharge Standard of Pollutants for Municipal Wastewater Treatment Plant" (GB 18918-2002) issued by SEPA on 24 Dec 2002 and implemented on 1 July 2003. Compared to its former counterpart, the new standard further elaborated pollutant restrictions based on a classification of WWTPs that takes the varieties of treated wastewater reuse, construction schedule and locality into consideration. As presented in Table 6.6, however, the concentration limits on P effluence become less stringent, especially for existing WWTPs. In comparison with the P standards of Urban Wastewater Treatment Directive (91/271/EEC) (Blöch 2001; Farmer 1999) and of Clean Water Act in United States (Litke 1999), China's new standard is rather lax (see Table 6.7). Regarding the differences in the environmental state of surface water between China and these developed regions, the substantial decreasing stringency in P limitation suggests an unwanted decline in the stimulation and enforcement of technological applications. Since most towns basically lack sewage systems, the lower limits could significantly hamper the further application of the centralized P control strategy in China.

On the other hand, the new standard represents the realistic governmental reflection on the low technological level of sewage treatment systems in China. Based on the PHOSFLOW model, daily ingestion quantity of P nutrient of Chinese urban residents accounted for 1402 mg P/capita·day in 2000. Taking the household detergents consumption (CNCIC 2002) and the total quantity of urban domestic wastewater (SEPA 2001) into account, the P concentration in urban wastewater will be on average 10.6 mg P/L at the national level, regardless of various industrial sources of P emission. Accordingly, the shift in P standards from 1.0 mg P/L to 3.0 mg P/L, for instance, means that the governmental goal in P removal efficiency decreases from 92% to 76%. Whereas the P-removal rate of primary, secondary and tertiary treatment methods can reach, depending on various techniques and equipments (Morse, Lester and Perry 1993; Brett 1997), 5% – 10%, 20% – 40% and 70% – 95%, respectively, the new standards on P effluence suggests a more rational and realistic expectation of the environmental performance of China's WWTPs. Combining our estimates

cities in 1996 (coastal economic open cities and special economic cities) and one in 2002 (Lhasa). Therefore, the total number of environmental key cities sum up to 47 by 2003.

on P removal efficiency and the governmental proposals of treatment rates, the national objectives for P removal from urban households by WWTPs can be roughly interpreted as 23%, 34% and 45% in 2000, 2005 and 2010, respectively, assuming that the national treatment rate of urban domestic wastewater is on average 60% in 2010 and that P concentration in influence is kept constant.

Table 6.6 National discharge standards on WWTPs (unit: mg P/L)

Discharge Standard		Standard I	Standard II	Standard III
Integrated wastewater discharge standard (GB 8978-1996) [a]		0.5	1.0	-
Discharge standard of pollutants for municipal WWTPs (GB 18918-2002) [b]	Constructed before 31 Dec 2005	A: 1.0 B: 1.5	3.0	5.0
	Start construction after 1 Jan 2006	A: 0.5 B: 1.0	3.0	5.0

Notes: [a] Standard I and II are relevant for the WWTPs discharge treated wastewater into Class III, and Class IV and V surface waterbody, respectively. For details on the classification of environmental functions of surface water, see Chapter 3.

[b] Standard I-A should be complied in case of wastewater reuse. The applicability of Standard I-B and Standard II are same with Standard I and Standard II of GB 8978-1996. The town-level WWTPs, except for those located in key watersheds or water resource conservation zones and those adopted secondary or tertiary treatment techniques, should implement Standard III.

The centralized P abatement can be very expensive due to vast investments on construction and operation of the centralized sewage system. Along with technological enhancement, costs of primary, secondary and tertiary treatment systems dramatically increase correspondingly.[49] This implies that more stringent requirement for P control will result in a rising input of financial resources on the centralized technosphere.[50] Although the total expenditure on urban sewage system in China reached 15 billion RMB Yuan in 2000 (NECW 2003), an increase by 15.6 times compared to 1990 (Anonymous 2004), it is apparent that more investments are needed, given the relative immature technological development and the poor quality of the domestic water environment. Based on the "Code for Design of Building Water Supply and Drainage" (GB 15-88) and "Norm for Budgetary Estimate on Construction Engineering in Beijing" (1996), Liu and Chen (2000) calculated the entire costs for urban sewage system by a numerical approach. The total construction expenditures averaged 1.86 billion RMB Yuan in 1996 prices for a city with one million residents. Household sanitation,

[49] While the costs of construction and operation increase in parallel with update of treatment levels, they can significantly vary among different countries. For rich discussions on this issue in case of China, see Wang, Wang and Li (1992); Wu (1992); Zhang and Qian (1998); Du, Du and Qian (2000); Chu et al. (2004); Liu, Mol and Chen (2005). On the other hand, it is difficult to distinguish the exclusive cost of P removal from the total expenditure on construction and operation of the centralized sewage system, because various pollutants can be partially eliminated at the same time. Morse and his colleagues (1993) presented a rough estimate that the costs of chemicals purchase and biological reactors operation for P control could annually reach 600 million euros in EU countries, averaging 1.8 euro per European per year. Besides this study, however, no more arguments are available so far.

[50] Total 68.5 billion euros, for instance, is requested from the European Countries during 1993 – 2005 to meet the Directive 91/271 requirements (Farmer 2001), while the United States has financed 200 billion and 154 billion dollars on, respectively, the construction and operation of WWTPs between 1972 and 1996 (Litke 1999).

sewer networks and WWTPs cost 0.52, 1.02 and 0.32 billion RMB Yuan, respectively. Accordingly, the national demand for infrastructure construction could rise up to 500 – 600 billion RMB Yuan, with an annual operational cost of 70 – 80 billion RMB Yuan (Liu, Mol and Chen, *forthcoming*). Compared to the actual investment, a sharp increase in capital inputs is desired. However, it is argued that the thorough renovation of urban wastewater infrastructure cannot be done overnight in China, due to several vital weaknesses. In short, the lack of sufficient and efficient financing incentives and market instruments, the weak fiscal capacity of local governments, the limited innovation through R&D institutions and the lack of mechanism for commercializing technical R&D results, and the increasing but still limited domestic supply of environmental equipment significantly undermine the future development of the centralized wastewater control systems, and thus the efficiency in controlling P from households in China (Liu, Mol and Chen, *forthcoming*).

The actual state of China's urban wastewater infrastructure cannot efficiently remedy the continuous deterioration of the water environment. First of all, only 34.2% of urban domestic wastewater was subject to treatment in 2000 (MOC 2001), which means that the majority of P was directly released into surface waters. While no advanced technologies (tertiary) were applied (Wang and Lou 2004), primary and secondary plants treated 32% and 68% of the wastewater in 2000 (MOC 2001), respectively. Given the limited results in nutrient abatement of primary and secondary methods, P to a large extent leaked into surface water without being effectively removed at the WWTPs. According to the accounts of the PHOSFLOW model, the national P removal efficiency accounted for 24% and only 8% of total P from urban households was abated in 2000. Compared to the national objectives at the same year, while the treatment rate exceeded SEPA's requirement by 9 per cent, the low P removal efficiency resulted in a discount of 15 per cent.

Second, WWTPs were placed in 232 cities, basically of large and medium scales, while most small cities and almost all towns had not launched construction projects in 2000 (NECW 2003). Third, most environmentally sensitive areas have not established complete urban sewage systems. For instance, the national program of water pollution prevention and control in the key watersheds of "Three Rivers and Three Lakes"[51] requested that 581 municipal WWTPs should be constructed before 2005. However, only 90 plants were built, 170 plants were under construction and the others were still not launched by Oct 15, 2003 (SEPA 2003). Fourth, the actual operation of the WWTPs was surprisingly weak. This can be conclude from the latest national environmental enforcement action, during which all WWTPs and industrial polluters were involved (SEPA 2004). The results show that 275 WWTPs were in irregular operation by 30 June 2004, accounting for 51.7%. Among these, 43 WWTPs (8.1%) were basically not running, 121 WWTPs (22.7%) operated at less than 70% of their designed capacity because they lacked associated sewage networks, and 111 WWTPs (20.9%) did not meet the new discharge standard over 152 days in average. More importantly, 60% of these standard violators were related to P overloading in discharged wastewater.

[51] Three rivers refer Huai River, Hai River and Liao River and three lakes include Taihu Lake, Chaohu Lake and Dianchi Lake. All of them are regarded as national key areas for water environmental protection. These six watersheds were upgraded as the national key environmental protection areas on the Forth National Environmental Protection Conference in 1996.

Compared to that in advanced countries, the China's achievement is rather limited. Under the EU 1991 Directive, for instance, 84% of European households were served by sewage systems, with most of the population connected to advanced treatment plants and 45% of them connected to tertiary plants, as shown in Table 6.7. Consequently, on average 56 – 62% of P from European households was removed in 1998 (Farmer 2001). As indicated by the EEA (2003a; 2003b), the levels of wastewater treatment in EU countries, particularly in Western Europe, have significantly improved since the 1970s, and the eutrophication of European lakes, reflected as P concentration, is generally decreasing, which strongly suggests the success of policies such as the urban wastewater treatment directive in reducing pollution of rivers. Therefore, it can be concluded that the environmental effectiveness of urban centralized P abatement in China remarkably lags behind Europe, although it has been considerably improved over last decades.

Table 6.7 Comparison of economic and technical profiles of centralized sewage systems

Country/ Region	mg P/L		Total costs of construction (C) & operation (O)	Outcomes	Environmental effects
	Influx	Efflux			
EU	7-34	1-2 [a]	C + O: 69 billion € (1993-2005)	Connected pop.: 84% (1998) Tech. level: sec. 55%; tert. 45% (1998)	P-removal rate: 56-62% (1998)
US	11	0.5-1.5	C: 200 billion US$ O: 154 billion US$ (1972-1996)	WWTP: 16,000 (1996) Connected pop.: 70% (1996) Tech. level: sec. 75%; tert. 25% (1996)	P-removal rate: 23-37% (1996)
China	10.6 [b]	0.5-5	C: 500-600 billion ¥ O: 70-80 billion ¥ (2000-)	WWTP: 427 (2000) Service areas: 65% (1995) Tech. level: prim. 68%; sec. 32% (2000)	w.w. treat. rate: 34% (2000) P-removal rate: 24% [b] (2000)

Notes: [a] The EU Directive (Directive 91/271) specified standards and related deadlines for wastewater collection and treatment, based on classification of environmental sensitive areas and WWTP scales in terms of population equivalent (1 unit p.e. = 60 g BOD/day). In addition, some countries stipulated rather strict discharge standards, e.g. Denmark (0.3 mg P/L) and Sweden (0.3-0.5 mg P/L).

[b] Own calculation based on the PHOSFLOW (3.0) model.

The development of the urban sewage infrastructures forces China to confront negative environmental impacts. As the centralized control strategy just removes 'pollutants' into sewage sludge rather than promotes a recovery and recycling of resources, including P, it does not really solve the problem (Beck 1997; Chen and Beck 1997; CEEP 1998; USEPA 1999; Stenger 2000; Hahn 2001; Brunner 2001; Suh and Rousseaux 2002; Kvarnström et al. 2003; Bengtsson and Tillman 2004).[52] The central focus is how to appropriately and efficiently

[52] Both advanced chemical and biological methods for P removal can generate a huge amount of sludge besides primary and secondary sludge. In general, biological methods produce 3.4 kg sludge by removing 1.0 kg P from wastewater. Sludge generation of chemical methods can be even higher, i.e. 5 – 7 kg sludge/kg P-removed, because of intensive input of chemical precipitators (Morse, Lester and Perry 1993; Brett 1997). More importantly, the additional costs for P control caused by sludge treatment, including concentration and dehydration, are also difficult to be quantified because of differences in applied techniques, operation and management.

dispose sludge, so that it does not lead to a second pollution.[53] While no available technologies for stable recovery and recycling of P are likely to be successfully commercialized within a short term (CEEP 1998; Driver 1998; Piekema and Roeleveld 2001; SCOPE 2001), applying sewage sludge to agricultural land remains the presently dominant approach (Bowker and Stensel 1990; EEA 1997; USEPA 1999). Average 50% of P in sewage sludge, for instance, was subject to recycling over the 1990s among EU countries (EEA 1997).

Traditionally, agricultural reuse of urban human excreta has been overwhelming in China and 90% in quantity was still applied to farmland in 1980 (Chen and Tang 1998). However, the emergence and subsequent development of sewage systems contributed to and continuously enlarged the nutrient gap between urban and rural areas in China. Some have argued that sewage sludge has become less acceptable for farmers, because of the gradual increase in individual consciousness on the potential ecological risks on the one hand, and the universal lack of necessary infrastructure for sewage sludge refinement and manure composting on the other (Bian 2000; Chen 2002). This is further illustrated by local manure application in the case of Dianchi Basin. The other approaches of sludge disposal, as previously discussed, are also difficult to adopt. Less than 25% of total WWTPs in China are equipped with sludge stabilization facilities, basic preliminary installations employed in the whole technological chain of sludge disposal (Wang and Mo 2001). More recently, SEPA (2004) proclaimed that the WWTPs' capacity of sewage sludge disposal was severely limited as the related facilities were still insufficient by the year 2003. As the application of centralized control strategy in China has substantially re-directed the urban human P flows from agricultural lands into natural soils by dumping or land-filling, negative environmental impacts arise in parallel. Although these P immobilized on natural lands slightly affect surface water, this P transferring positions itself against to the underlying proposal for ecologizing P flows by the substantial social needs for cleaning water.

Although the relevant data are limited, particularly in relation with sludge disposal, the analysis above shows that China, as a developing country in pursuit of the centralized P control strategy, has not achieved a substantial success in regulating the outflow of P from urban households. While the policy objectives of national P abatement in general are relevant compared to societal needs, relying on the international experiences and the state of domestic water environment, our analysis reveals that the policy effectiveness was reduced by 65% in 2000. A significant improvement of the centralized P abatement, for instance, towards that of

[53] The possible options include agricultural reuse, ocean dumping, land-filling, incineration, digesting for industrial materials, and decomposing for biogas generation. Every solution has its own disadvantages: ocean dumping was prohibited in many countries due to significant influences on ocean ecology; land-filling has become less economic in parallel with the gradual increase in land rent as well as sludge volume; incineration can remarkably reduce the volume of sludge but the relative high investment and potential air pollution undermine wide application; the technologies of industrial utilization and biogas generation emerged in the 1970s but are still hardly commercialized. A number of barriers also challenge agricultural application, the currently dominant approach for sludge disposal. They include the potential technological barriers and investment gap of sludge refinement and transportation, the temporal inconsistence between sludge generation and application, the acceptance of public and especially farmers, and particularly the high risk in public health. For a sophisticated discussion on sludge disposal, see Brett et al. (1997); CEEP (1998); Bohm (2001); Hahn (2001); Le Moux and Gazzo (2001); Kvarnström et al. (2003); Bengtsson and Tillman (2004); Lundin et al. (2004).

the European countries, is likely to be a long-term perspective, because the urban wastewater infrastructures are universally underdeveloped. Measured by the huge investment, the limited environmental effect becomes even less cost-effective. More importantly, the costly and rigid technological regime can hardly facilitate P recovery and recycling that are underlying propositions of an ecologically rational eutrophication policy. This is then responsible for a decrease of the Chinese social welfare in our evaluative term.

Because of the intrinsic weaknesses of the centralized sewage system in sustainable water use and nutrient recovery and recycling, some alternative strategies have been argued. Among these, decentralized source-separated strategies have received increasing attention since the mid 1990s (Larsen and Gujer 1996; Beck 1997; Esrey et al. 1998; Otterpohl, Albold and Oldenburg 1999; Berndtsson and Hyvonen 2002; Wilsenach and van Loosdrecht 2003; IWA and GTZ 2004).[54] This decentralized and downsized sanitation concept, focused on ecologically sustainable and economically feasible closed-loop systems rather than on expensive end-of-pipe technologies, advances a new philosophy. It departs from the one-way flow of excreta from terrestrial to aquatic environments, as introduced by the conventional flush-and-discharge sewage system. The holistic alternative separates nutrients and domestic used water at source and handles both components individually based on material flows approaches. As such, it avoids the disadvantages of conventional wastewater solutions and enables and facilitates nutrient recycling. Although the reinvention and transition of urban wastewater systems poses a major challenge, it does provide a promising prospect for future P recovery and recycling in an ecological and economic efficient way.[55]

The source-separated strategy can make sense for China's future promotion of controlling P from urban households. The underdevelopment and the low degree of penetration of the centralized sewage system make a shift of the conventional technological regime in China much easier. Up till now, this revolutionary transition of urban wastewater infrastructures has not become notable objects of study within Chinese academic schools and governmental agencies.

6.3.2 Urban centralized P control in Dianchi Basin

An urban sewage system has been developed since 1990 in Kunming City, one of the environmentally key cities, aiming to alleviate urban pollution stress on Dianchi Lake. As indicated by the local ordinance of "Regulation on Urban Sewage in Kunming Municipality"[56], implementation of a centralized strategy is officially upgraded as the basic principle of urban sewage management.

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[54] Some demonstration projects were even carried out in European countries, especially Sweden (Fittschen and Niemczynowicz 1997; Hanaus, Hellstrom and Johansson 1997; Skjelhaugen 1999; Czemiel 2000; VERNA and HSB 2000).

[55] Detailed studies are essential as a first step, *inter alia*, of technological, organizational, economic and social aspects. In addition, the involvement of multi-stakeholders, such as residents, building owners, farmers, politicians, officials and other interested parties from the start seems essential (Larsen et al. 2001; Pahl-Wostl et al. 2003). All these problems cannot be solved overnight, as it requires nothing less than a paradigmatic change of a large socio-technical system.

[56] This ordinance was adopted by the Eleventh Kunming Standing Committee of National People's Congress on 24 November 2001 and was approved by the Ninth Yunnan Standing Committee of National People's Congress on 21 January 2002.

Compared to the achievement at the national level, the development of urban sewage infrastructures in Kunming city is more successful (SEPA 2003). By the end of 2000 four WWTPs were employed for the treatment of 61.8% urban domestic wastewater of Kunming city, nearly double of the national average (Guo and Ma 2000). Combining designed capacities with actual treatment quantities of these WWTPs, the daily operations were much better than elsewhere in China, which in turn provided a stable environmental achievement. With respect to P removal, the adaptation of activated sludge methods involving oxidization channel, A^2/O approaches and sequence bulk reactors (SBR), advances nutrients abatement as well (KDAB 2004a). According to in-situ monitoring data of efflux quality and as shown in Table 6.8, the P removal efficiency was on average 74.1%, 50 per cent higher than the national figure (Ma et al. 2003). With these data, 45.8% of P generated from Kunming households was removed in 2000. According to the PHOSFAD model, it accounted for 394 tons P transferred into sewage sludge, represented by the flow S_{10} as shown in Figure 5.2.

Table 6.8 Urban wastewater treatment and P removal in Dianchi Basin in 2000

WWTP	Designed capacity (kt/day)	Treatment quantity (kt/day)	Cost (million RMB ¥)	Date for built up	P concentration		P removal efficiency (%)	Sludge quantity (ton)
					Influx (mg P/L)	Efflux (mg P/L)		
No.1	55	52	33.0	1990	3.4	0.3	91.2	227.4
No.2	100	101	138.8	1995	3.1	0.5	83.9	406.4
No.3	150	145	189.0	1997	2.8	1.3	53.6	518.7
No.4	60	74	60.0	1997	3.1	0.5	83.9	214.6
Total/ Average	365	372	420.8		3.0	0.8	74.1	1367.1

Sources: based on Guo and Ma (2000); Ma et al. (2003); KDAB (2004a).

The application of centralized sewage technologies has placed high expectations by both central and local governments. In "The Ninth Five-Year Plan for Water Environment Protection and Control in Dianchi Basin and the Long-Term Targets for the Year 2010" approved by the State Council in 1998, the treatment rate of urban wastewater was requested to reach 80% in 2000. However, it was subsequently criticized as an unrealistic objective (Zhang 2001; Qiang 2002; He and Zhao 2003). "The Tenth Five Year Plan for Water Pollution Prevention and Control of Dianchi Lake", for instance, acknowledged that the objective was in general over-designed, because it overlooked the difficulty in alleviating lake eutrophication, and the lack of integrated measures, financial input and administrative institutions. Therefore, the 80% objective for urban wastewater treatment was related to the year 2005. Furthermore, all new and expanded WWTPs were required to adopt secondary or enhanced secondary treatment technologies, aiming at better performance in nutrient abatement. With respect to the current technological profile as discussed earlier, these requirements are appropriate to maintain P removal efficiency at a relatively stable level. But meanwhile it is difficult to provide sufficient stimulations for further improvement of P abatement. It is also notable that the local government has not localized the national discharge standard (GB 18918-2002) by putting more stringent limits on P effluent. Measured by Standard I-B as presented in Table 6.6, all four WWTPs in Kunming city complied with the national regulation (< 1.5 mg P/L), although the P effluent of No.3 WWTPs was relatively high. However, as Ma and his colleagues argued (2003), controlling P effluent concentration within 0.5 – 1.0 mg P/L would be the appropriate standard with respect to nutrients enrichment in Dianchi Lake. In addition, no local formulation conditioned sewage infrastructures in towns in the Dianchi watershed area. This means that the local town-level sewage management should obey related national requirements for treatment rate of domestic wastewater, e.g. *chengjian*[2000]124, which are rather lax as discussed earlier. Whereas these towns were responsible for only 5.5% of total urban population in Dianchi Basin in 2000 (KSB 2001), the local sewage regulation focusing on Kunming city does not substantially undermine the desired environmental effect on controlling P from urban households of the whole basin. Therefore, the policy's relevance and effectiveness remain high.

The relative maturity of urban wastewater infrastructures in Kunming city was mainly attributed to persistent governmental investments over the last decades. In particular, it benefited from strong fiscal support from the central government on pollution control of national environmental key areas. The investments on local urban wastewater infrastructures accounted for 420.8 million RMB Yuan, as shown in Table 6.8. As the total environmental expenditure on Dianchi Lake reached 3.09 billion RMB Yuan between 1990 and 1998 (Wang 1998), the cost of Kunming sewage system accounted for 13.6% of the total financial input. It would be interesting to compare the investments percentages at the national and local levels. Together with the quantity of removed P by urban sewage systems, this could generate insights into the cost-effectiveness question. However, large data gaps make such a comparison in possible.

Table 6.9 Outline of the WB funded subprojects for urban sewage systems

Location of sewage systems	Design contents		Construction Schedule		Funds (million RMB ¥)
	Capacity (thousand tons/day)	Length of sewer pipelines (km)	Start	Complete	
Kunming	No.1 WWTP: 65	4.0	Unknown	Unknown	123.0
	No.5 WWTP: 75 No.6 WWTP: 50	58.4	May 1999	Nov 2001	229.0 187.0
Jinning	15 – 30 [a]	3.5	Oct 1999	Unknown	37.02
Chenggong	15 – 30 [a]	12.0	July 2000	Unknown	40.16
Total	220 – 250 [a]	77.9			616.18

Sources: KDAB (2004b); He and Zhi (2004).

Notes: [a] According to the World Bank program, the designed treatment capacities of the two town-level WWTPs will reach 15 kt/day by 2005 and further double by 2010.

The development of urban sewage systems in Dianchi Basin was also accelerated by an international cooperation program since 1997, i.e. Integrated Environmental Regulation

Program in Dianchi Basin, sponsored by World Bank (WB) with 1.8 billion RMB.[57] As shown in Table 6.9, the fund of 616 million RMB Yuan on urban sewage infrastructure, nearly one and a half times the previous expenditures on it, has to a large extent complemented the universal weakness of governments' financing capacity, and thus has stimulated the centralized sewage infrastructures in Dianchi Basin. Together the new sewage infrastructures, the total treatment capacity will increase to 585 and 615 thousands tons per day by 2005 and 2010, respectively, accounting for 80% of urban domestic wastewater of Dianchi Basin (He and Zhi 2004). Moreover, all these new WWTPs adopt secondary or enhanced secondary treatment techniques, as the governmental regulation requests. This will definitely benefit the local P abatement: the two two-level WWTPs adopt ordinary activated sludge methods; another two relative large sized WWTPs located at Kunming city apply nutrient-removal enhanced A^2/O techniques. Accordingly, it was argued that the P abatement efficiency could maintain at least 70% (He and Zhi 2004) and even increase to 82% (He and Zhao 2003). As SEPA (2003) stated that No.5 WWTP was scheduled for final checking and No.6 WWTP was in a pilot-operating process by 15 Oct 2003, it can be expected that the new WWTPs could begin ordinary operation in 2004. By the year 2005, therefore, 56.0% - 65.6% of P from Kunming urban households could be removed. In addition, as a large part of outflows of the two town-level WWTPs was planned to be used by the agricultural sector (He and Zhao 2003), the quantity of P released to surface water from town households could reduce remarkably.

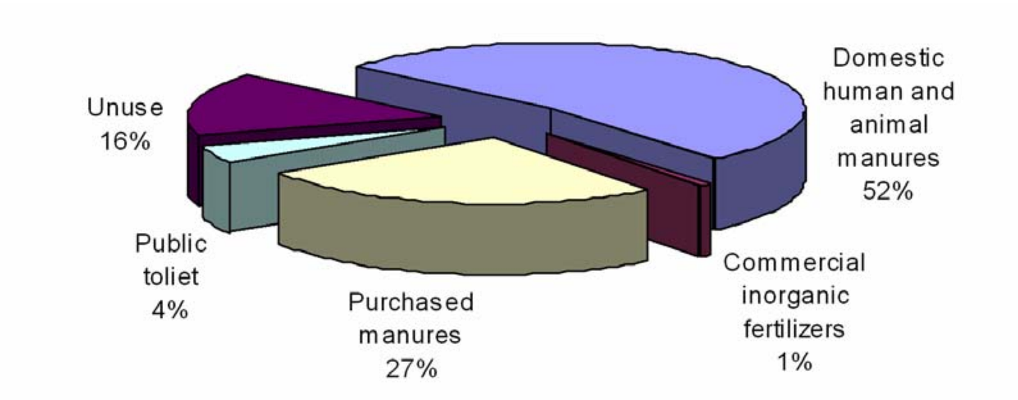

Figure 6.2 Main sources of manure application in Dianchi Basin

Sources: based on field investigation in 2002.

It is apparent that the relative advanced wastewater treatment generates more sewage sludge in Dianchi Basin. By 2000, all four WWTPs equipped sludge dehydration facilities, and sewage sludge was basically subject to land-filling (He and Zhao 2003; KDAB 2004a). At the same time, however, the local centralized sewage system has completely destroyed the P

[57] The World Bank project includes seven sub-programs: 1) construction of urban infrastructures in Kunming city; 2) enlargement of water supply for Kunming city; 3) collection, disposal and treatment of industrial and hospital wastes and urban domestic wastes; 4) construction of sewage system and treatment facility for Chenggong County and Jinning County; 5) industrial pollution control of Kunming Chemical Fertilizer Plant and Kunyang Phosphate Fertilizer Plant; 6) improvement of rural environment; 7) environmental monitoring and capacity building. For details see Xu (1996); Wang (1998); Zhang (2001).

recycle from urban households to rural farmland, which in turn significantly undermines the possibility for ecological redirection of P flows. The locally pervasive manure reuse as discussed earlier shows no acceptability of sewage sludge by individual farmers, because they are afraid that the market of flowers and vegetables could stagnate due to the potentially high ecological risks. According to our field investigation in 2002, while manure application dominated local farming practices, none of the 460 individual farmers used sewage sludge on their farmlands.

It can be concluded that the overall profile of local centralized sewage infrastructure is impressive with better performances compared to the national average. In parallel with the eutrophication of Dianchi Lake, local government proposed rather stringent goals in term of wastewater treatment rate. This then enables a stricter and thus relevant objective for P abatement. With respect to the environmental outcomes in 2000, the effectiveness of local policy is reliable and also could be promising for the year 2005. The central government funds and international financial aids contribute to intensive expenditures on local sewage systems. While these contributed to local P abatement, it also makes a low cost-effectiveness of the local centralized strategy visible. The empirical study also reveals that an ecologically rational sludge disposal, i.e. agricultural application, has not been practiced in Dianchi Basin. This, in parallel with the discussion at the national level, is responsible for an undesired decline of the social welfare.

6.4 The Ban of P Containing Detergents

Limitation and prohibition of the consumption of P, in form of STPP as an additive to detergents, is a specific measure aiming at decreasing P release at source and thus alleviating P enrichment in surface waterbodies. This P-based strategy has been adopted widely, especially in EU countries and North America. However, since the early 1990s, discussions have emerged on the actual environmental effects of this policy measure. The international experiences and arguments are important and China could learn from it to develop rational policy decisions from a holistic perspective.

In this section, we will start with a brief introduction on STPP use and bans in a global context. The emphasis will then be on questions of environmental effects and negative impacts of this measure. Second, since P ban is not a Chinese common policy at the national level, we will evaluate this policy via a number of domestic empirical studies. Third, analysis of policy implementation in Dianchi Basin, where P-containing detergents ban was put into practice in 1998, will not suffer from the data gap at the national level.

6.4.1 STPP use in a global context: background of the P ban

STPP (sodium tripolyphosphate, [$Na_5P_3O_{10}$]) is the most widely used washing builder and, in conjunction with surfactants, allows detergents to perform efficiently in all washing conditions. It has been argued that no other single chemical can offer all, or even most, of the different functions of STPP.[58] The industrialized production of STPP began in 1943 and reached the largest global production in the 1960s. The maximum quantity of STPP production in the US, for instance, accounted for 220 mkt in 1967 and was responsible for 10% of total domestically consumed phosphate rocks (Cao 1997a). Afterwards, global production has gradually declined mainly due to bans of phosphate-containing detergents in developed countries. In the late 1990s, it decreased to one half of the peak amount, with an average consumed quantity of 8 – 9 kg/cap·yr in the US (Litke 1999).

The functions of P, in the form of STPP as the detergent additive, can be replaced theoretically by other chemicals. With the spreading of eutrophication worldwide since the 1960s, a number of developed countries launched campaigns to restrict production and consumption of STPP, aiming at reducing the household P load on freshwaters. This kind of source-control measures, i.e. limiting or banning P contained in detergents, was successful in these countries (refer to Table 6.10). In the early 1990s, for instance, consumption of P-free detergents accounted for 45%, 97% and 100% in US, Japan and European countries (Litke 1999; Morse 2000; Wang, Wei and Wang 2003).

The environmental impacts of P-free detergents have been under discussion since the middle 1980s. The central question was to what extent P detergents ban measures lead to an improvement of eutrophication by reducing household P load. Hoffman & Bishop (1994) discovered a comparable reduction of P concentration in domestic wastewater before and after the implementation of the ban. Lee and his colleague (1995) subsequently criticized that load reduction can not be a correct indicator for the improvement of water quality. After analyzing more than 400 monitoring data derived from nearly 40 lakes in US and European countries, they argued that a public perceivable change in eutrophication related water quality requires a reduction of about 20% to 25% algal available P load to the waterbody (Lee and Jones 1986). This suggested that a P-ban measure would not benefit environmental improvement, if the percentage of P released by households via detergents consumption of the total P load from various sources is lower than one fourth. Notably, this statistical percentage often exceeds the actual figure (cf. Table 6.10). Today, most scholars have acknowledged that limiting or banning household consumption of P-containing detergents would hardly lead to a significant environmental effect. As the UN claims, P-free detergents use contributes to a limited degree to the reduction of P emission at source and thus could pose a marginal impact on freshwater environment (Li 2000).

P detergents ban brings up negative impacts on twofold. First, none of the substituting chemicals, such as sodium carbonates, sodium silicates (zeolites A) and sodium nitrilotriacetate, can work as well as STPP additive. According to a broad experimental test on washing effectiveness of detergents conducted in Austria, Spain, Belgium, French and

[58] Several important functions entailed by STPP include: 1) counteract the effects of calcium and magnesium salts present in hard water and in soils, thus allowing surfactants to function correctly; 2) prevent deposit of calcium and magnesium incrustations on fabrics and on the washing machine's heating elements; 3) break up large particles of dirt into smaller ones and keep them in suspension in wash water, i.e. anti-redeposition function; 4) stabilize detergents' physical properties; 5) facilitate dissolving of detergents; 6) re-dissolve calcium and magnesium compounds present in washing machine from previous washes.

England, washing performances of most P-containing detergents were clearly better than P-free detergents (CEEP 1998b). Second, a number of academic studies show that all these substitutes have worse impacts on the environment and human health that STPP. Based on LCA approaches, for instance, it was concluded that STPP has the least impacts on environment and human health, both from an environmental and an economic perspective (Morse et al. 1993; Milson and Jones 1994). In particular, 17 European experts involved in a Delphi method came to the consensus that P detergents ban had no advantages compared to conventional wastewater treatment and tertiary treatment is a more reliable solution for control P from households (Cao 1997b). In parallel with these reflections and studies on P-containing detergents use, some developed countries have re-promoted the production and consumption of STPP since the 1990s (Anonymous 1995).

6.4.2 Analysis of P detergents ban in China: An overview

The use of STPP has rapidly increased with economic development and population growth in China sine the 1990s. The production of various detergents and STPP reached 3582 mkt and 682 mkt in 2000, respectively, with increase rates of 39% and 163% compared to 1990. While the national production ranked second in the world in 2000, per capita consumption was far less than the global average. In terms of various products of household detergents, the quantity of per capita consumption was on average 3 kg/per·yr in China, much lower than 27 kg/per·yr in US, 25 kg/per·yr in Germany, 10 kg/per·yr in Japan, and 7 kg/per·yr for the whole world in the late 1990s (Zhang 1999).

China has been involved in research and development of STPP substitutes over the last two decades. In the mid 1970s, the former Ministry of Light Industry (MLI)[59] promoted industrial institutes to engage in inventing possible substitutes. This led to the establishment of the first product line for zeolites with a designed capacity of 20 thousands tons in 1992 (Liu 1999). Subsequently, China stipulated a series of standards for the production of P-related detergents in the 1990s. The “Standard for Zeolites 4A in Non-Phosphate Laundary Powders” (QB1767-93) and “Standard for Zeolites 4A in Detergents” (QB1768-93) were promulgated on 1 July 1993 by MLI. Correspondingly, SEPA provided the first technical requirement of environmental-labeled products by “Non-Phosphate Detergents for Fabrics” (HJBZ008-95) on 28 Dec 1995. Combining the first two standards, the central government issued the national standard “Non-Phosphate Laundry Powders” (QB13171-96) in the name of China National Light Industry Associations (CNLIA) in 1996. The standard fostered the emergence of a national technical standard, i.e. “Laundry Powders” (GB/T13171-1997), promulgated by the State Technical Supervision Bureau (STSB)[60] on 28 May 1997. Both of them became effective on 1 January 1998. It can be noted that, without specific formulations on detergents consumption, all these standards are orientated to technical guidelines from a production

[59] The Ministry of Light Industry was retracted in 1993 in parallel with China’s administrative adjustment promoted by the Centre and some authorities were switched to the new established China National Light Industry Associations (CNLIA).

[60] In 1998, STSB was changed its name into State Bureau of Quality and Technical Supervision (SBQTS). It was afterwards retracted in 2001 and was combined into a new established governmental bureau – i.e. State General Administration of the People’s Republic of China for Quality Supervision and Inspection and Quarantine (AQSIQ) – with the State Administration for Entry-Exit Inspection and Quarantine (under the General Administration of Customs).

perspective. Moreover, none of these regulations related to a compulsory technological switch from STPP to other substitutes on the production side. Therefore, these regulations cannot be regarded as national P detergents ban policies.

The first local ordinance in relation with distribution and consumption of P-containing detergents came into being in Taihu Basin of Jiangsu Province in Oct 1996.[61] A number of municipalities, involving Wuxi, Kunming, Shenzheng, Dalian and Shanghai, subsequently advanced more or less similar regulations (Jiang 1999). In general, those measures focused on laundry powder [62] consumption in households, but lacked necessary outlines on commercial-used synthetic detergents.

Since the late 1990s, many Chinese scholars have been involved in a controversy on whether the central government should promote these P detergents ban campaigns and to what extent they could contribute to the desired improvement of lake eutrophication. Liu and colleagues (1998) claimed these measures should be given a high priority in all population intensive regions to P load for households. Jiang (1999) followed up with a comparative analysis, which concluded that P detergents ban should be strengthened instead of relying on costly sewage systems. In contrast, other domestic experts were rather conservative or even negative regarding P detergents bans according to local empirical evidences. Based on two-year monitoring data following a controlled experiment in Taihu Lake, Shu and colleagues (1999) and Huang (2001) suggested that the P detergents ban should not be regarded a predominant measure due to the limited environmental effects on the lake. The studies on Xihu Lake in Zhejiang province and urban waterbodies in Shanghai justified this argument as well (Qian 2000; Song et al. 2003).

To illuminate the environmental effects of P detergent bans, we now focus on Taihu Basin based on literatures (Shu et al. 1999; Huang et al. 2000; Huang et al. 2001) and monthly reports on water quality from Oct 1998 to Jan 2001, provided by China National Environmental Monitoring Center (CNEMC). The results show that the local ban program led to 16 – 25% decline of P concentration in domestic wastewater during Jan 1999 – Dec 2000. However, monitored P concentrations in inflow streams and the lake did not follow this reduction: 9 inflow values decreased, and 8 values increased and 4 values remained relatively unchanged in 1999. In 2000, the corresponding numbers were 6, 11 and 3 respectively. Regarding water quality of Taihu Lake, average P concentrations and the eutrophication index decreased by 16% and 3% in 1999, but rebounded by 19% and 7% in next year. Apparently, a puzzle arises due to the inconsistency between the reduction of P detergent use and actual environmental effects. As detergents use is not the unique source of P load, the monitoring results illustrate that local P detergent ban programs can hardly lead to expected environmental improvement independently, although it does contribute to an abatement of P at source. This is consistent with global experiences as shown in Table 6.10. Therefore, it

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[61] The "Ordinance for Water Pollution Prevention and Control in Taihu Basin of Jiangsu Province" was approved by Jiangsu Standing Committee of National People's Congress on 14 June and was carried out on 1 Oct 1996. It stipulates ban and limitation of P-containing detergents in Class I and II, and Class III conservation areas, respectively, from 1 Jan 1999.

[62] STPP content in ingredients accounts for 17%, converting to a P content of 4.25%.

suggests that detergent P ban measures can only be regarded as one of the components of an integrated policy framework, instead of a panacea for curing eutrophication.

Table 6.10 P detergents use and environmental effects of P ban programs

Country/ region	P use (g P/pers·y)	P in ww (%) [a]	P load (%) [b]	Environmental effects	
				Outcomes	Impacts
US (1960s)	1000-1500	70	33	STTP consumption: 1 Mt in the 1970s decreased to 560-600 Tt Efflux of WWTPs in target areas: 35-45% lower	No significant improvement; but some non-targeted areas made progress
Japan (1970s)	-	38-64	14-24	non-STPP detergents: 97% of total production in 1988	Unclear
EU (1970s)	-	65	14-19	Emission: decreased to 2 g P/p·d Load: decreased to 2-3% of total	No obvious influence on water
Taihu (1998)	187	27	16	P in wastewater: decreased by 16-25%	No considerable improvement
Dianchi (2000)	166	31	22	Unknown	Unknown

Sources: based on Cao (1997b); Litke (1999); Morse (2000); Wang, Wei and Wang (2003).

Notes: [a] it refers to the contribution of detergent use to P concentration in domestic wastewater; [b] it refers to the contribution of detergent use to total P load.

In parallel with international studies, it has been argued that P detergents ban could introduce unexpected social and economic impacts to China. First, because of the difficulty for STPP suppliers to change products, declining demand of STPP imposes serious consequences on them.[63] In 2000, China had over 30 STPP suppliers with 30 thousands employees, responsible for 682 thousand tons and over 2 billion RMB Yuan output in product and value terms, respectively (Li and Xie 1999). Although in 2000 41% of STPP was exported, it is clear that a proposed national campaign of P detergents ban could lead to significant losses of social and economic welfare.

Second, additional high costs are requested to produce STPP substitutes. China's production quantity of zeolite 4A, one of the most promising substitutes, reached 40 thousands tons in 2000 (Shu et al. 1999). This production could only substitute 10% of STTP used in various domestic laundry powders, i.e. 1710 thousands tons (CNLIA 2001). Besides potential demands in machinery and infrastructure, a technical change is required due to adoption of P-free ingredients. Based on the differences in ingredients cost as presented in Table 6.11, a switch from STPP to zeolite 4A could lead to additional investments of 4 billion RMB Yuan.

[63] Pyrogenation methods and extraction methods are regarded as main techniques employed for production of STPP. Applying the former techniques, apatites [$Ca_2F(PO_4)_3$] is deoxidized into phosphor element under 1500-1600 Celsius degree and is in turn processed into phosphoric acid which is basic material for manufacture of STPP and other phosphates. The latter is to process apatites with sulfuric acid under certain conditions. About 90% STPP suppliers in China adopted pyrogenation method because of higher quality of phosphoric acid and less wastes generation. In comparison with extraction methods, however, its higher productive cost and lack of purification equipments considerably undermine possibilities for shifting to production of fertilizers or food additives. For a sophisticated discussion on production of phosphate industries, see Jiang (1999).

Third, higher price but worse performance of P-free detergents is unlikely to attract most consumers and thus limits market share if under full competition. The results of Chinese detergents washing performance tests are consistent with the international ones. One of these domestic tests on 20 brands of P-free laundry powders showed that only one brand achieved the washing quality requested by the national standard QB13171-96 (Zhang 1999). Moreover, the performance of P-free detergents is even worse in northern areas of China where the cadmium concentration in groundwater is higher than in the South. As the additional production cost could theoretically lead to an increase of 23% in market price as shown in Table 6.11, P-free detergents could hardly compete with STPP products.

Table 6.11 Ingredients and prices of laundry powders in China (units: %, Yuan RMB/ton)

	Surface activator	STPP	Sodium carbonate	Soluble glass	Zeolite 4A	Organic acids	Fibrins	Brightener	Sodium sulfate	Total Cost
P-containing	15.0	15.0	5.0	5.0	0.0	0.0	1.0	0.1	56.0	2782
P-free	18.0	0.0	10.0	5.0	20.0	3.0	1.0	0.1	40.0	3430
Price	9000	5000	1600	2000	2800	20000	10000	10000	700	

Sources: Components of laundry powders cited from the national standard (GB/T13171-96); prices of raw materials are cited from Shu et al. (1999).

Forth, implementation of P detergents ban requests effective market supervision, management and control, which is a challenge in transitional China. A number of small-scaled enterprises engaged in production of P-free detergents have been equipped with low level technologies and illegitimate prescriptions. Some of them adopted sodium carbonate or other chemicals (e.g. APE) as substitutes of zeolite 4A. According to some, these substitutes to a certain degree decrease production cost and improve washing performance, but meanwhile impose more negative impacts on environment and human health (Shu et al. 1999).

It is difficult to conclude our policy evaluation at the national level, because the P detergents ban has not been updated as a common policy in China. However, the overview of the local empirical studies facilitates general conclusions, although the cost-effectiveness of these measures remains unknown. Undoubtedly, the common objective of this kind of policies, i.e. banning P use at source and thus reducing P emission is relevant for alleviating eutrophication. The problems occur when taking actual environmental effects and negative societal impacts into consideration. Apparently, none of these current local P detergents ban programs resulted in a significant improvement of surface water quality in China, while the environmental outcomes of varied considerably. Therefore, the effectiveness of these measures could call questionable within China. At the same time, it can be argued that the social and economic impacts, involving STPP suppliers, the additional cost of substitute production, the low acceptability of P-free detergents due to their poor washing performance and the challenge of market supervision and enforcement, put pressure of P detergent ban politics. Together with the discovery of the more negative environmental impacts of substitutes as discussed earlier, the overall impacts of the P detergents ban lower the social welfare.

6.4.3 Implementation of P detergents ban in Dianchi Basin

P detergents ban has been promoted by the local government as a preferential strategy for an effective P control at source. Regarding the date of policy implementation, Dianchi Basin is the first watershed under such kind of regulation in China. On May 26, 1998, Kunming local government promulgated an official document, i.e. the "Announcement for Prohibition of Distribution and Limitation of Consumption of P-containing Detergents in Dianchi Basin", and proclaimed it was to be carried out from Oct 1 in the same year, three-months earlier than the implementing date of Taihu Basin.[64] The Kunming Dianchi Administration Bureau (KDAB) was authorized as the governmental agency responsible for mobilizing and organizing the implementation of the ordinance in practice.

KDAB has developed three measures in relation with regulation for P detergents ban: 1) unannounced inspection on production, distribution, marketing and consumption, together with all relevant governmental agencies; 2) ordinary market supervision implemented by Kunming Administration Bureau of Industry and Commerce (KABIC); and 3) public propagation and education relying on local media and sub-district offices. Among others, unannounced inspections – without informing time and place to involved actors in advance – have been advocated as the most important measure.

The measure of unannounced inspection was undertaken in the form of joint-actions. Based on the KDAB (2000), a close insight into one of these joint-actions can be given. On 17 March 2000, a large unannounced inspection, covering producers, marketing and commercial consumers, was carried out in Kunming city. First, five main enterprises were requested to report their recent productions and to provide production records to inspectors. The results showed that one enterprise rectified prescription of laundry powders and stopped the use of P chemicals from the day the promulgation was announced. Two enterprises continued producing P-containing laundry powders, but didn't distribute them within Dianchi Basin. The remaining two were newly established enterprises, exclusively producing P-free laundry powders, after the announcement came into being. Second, while no evidence from ten large-scaled emporiums in Kunming city indicated offences against the announcement, P-containing laundry powders were still prevalent on wholesale markets and accounted for some 750 tons, responsible for half of total quantity of various laundry powders. Most wholesalers defended themselves with the fact that there was no prohibition on sales to outsides of Dianchi Basin. Third, 15 main commercial consumers, involving hotels, restaurants and cleaning corporations, were subject to inspection and monitoring. Among others, only one actor used P-containing laundry powders. In parallel, similar inspection programs were undertaken in rural areas within the jurisdiction of Kunming Municipality in the following week. In total 2312 households – some of them engaged in small retail business – were subject to unannounced inspections, and 2% of them were subject to confiscation and charges for their uses of P-containing laundry powders, as shown in Table 6.12. In parallel, a one-week propagation program was carried out in Kunming city, coordinated with public

[64] Before this ordinance, Kunming government published a proclamation "Announcement for an Enlargement of P-free Detergents Use" on 10 Oct 1996, and called upon for an active participation of producers and consumers.

media and sub-district offices. The main activities included broadcasting the announcement on television, consulting and publicizing on-street and circulating posters and materials.

It is apparent that the intensive input of human and administrative resources during enforcement comprised considerable implementation costs. As a coordinating authority on Dianchi Lake, KDAB mobilized a number of local administrations, involving Kunming Committee of Economy and Trade, Kunming Financial Committee, Kunming Technical Supervision Bureau, Kunming Environmental Protection Bureau, Kunming Administration Bureau of Town and Village Enterprises, Kunming Administration Bureau of Industry and Commerce, Kunming Bureau of Public Security and so forth, to engage in this first joint-action in 2000. As shown in Table 6.12, the unannounced inspection entailed at least 549 governmental personnel for an entire week, excluding a large number of volunteers. Together with expenditures on the propagation program during the same week, the total implementation costs are substantial.

Table 6.12 Results of the first unannounced inspection in 2000 on P-ban in Dianchi Basin

County/ district	Number of inspecting targets under check [a]	Number of inspecting targets in charge	Confiscated P laundry powders (kg)	Fines (RMB Yuan)	Personnel involved
Wuhua	1586	8	1016	4800	163
Panlong	70	37	367	1200	37
Guandu	31	n.a. [b]	234	n.a.	24
Chenggong	116	n.a.	300	n.a.	n.a.
Jinning	84	n.a.	325	n.a.	245
Xishan	225	n.a.	1569	600	70
Songming	200	1	32	n.a.	10
Total	2312	46	3843	6600	549

Sources: KDAB (2000).

Notes: [a] The inspecting targets include individual sellers and household consumers; [b] n.a. refers no data available.

Taking into account the expenditures, the measure of unannounced inspections can hardly become a regular measure. Because local municipal governments did not provide special administrative funds on implementing the P ban policy, KDAB had to limit the frequency of joint-actions to 2 – 3 times annually. Furthermore, KDAB requested costs sharing with other governmental departments, relying on its higher jurisdiction within the administration of conservation of Dianchi Lake.[65] However, these costs were unexpected expenditures and thus were seen as undesired by participating governmental departments. Since the same is true for

[65] According to the "Ordinance for Conservation of Dianchi Lake", the fundamental responsibility of KDAB was stipulated as to bring all control measures for Dianchi Lake into effect by coordinating related governmental departments and organizations.

all these unannounced inspections, therefore, this administrative weakness will not produce significant changes in the behavior of involved actors as well as the market.

The same situation came up with the second measure. Without continuous budget support from KDAB and other agencies, ordinary market supervision, as desired by KDAB, was difficult to be integrated into the daily functions of KABIC. Wholesale markets, within the jurisdiction of KABIC, continued with selling huge amounts of P-containing laundry powders. Although the wholesalers claimed that these products were prepared for outside markets, it is very likely that these are distributed within the basin, in particular the rural areas as discussed below. Besides the lack of administrative funds, the reasons for the weakness of market supervision are twofold. First, it is important to keep in mind that markets follow consumer preferences. According to our field surveys in both 2000 and 2002 (see Chapter 2.5), a considerable amount of individuals – particularly rural residents – insisted in using P-containing laundry powders, because of their low prices and better performance in bubbling and cleaning. This market demand resulted in market supply and stimulated a number of wholesalers and individual businessmen in selling P- detergents. Second, an open market follows principally price signals and less administrative intervention. P-containing laundry powders entered circulation within Dianchi Basin via various trading channels. Therefore, the large amount of P-containing laundry powders were found on the wholesale market and a lot of rural grocers never sold P-free laundry powders, although most local producers under inspection stopped detergent production with P chemical additives.

The emergence of a large number of P-free laundry powders further enlarged difficulties in market supervision. According to KDAB (2000), there were over 50 brands labeled as P-free on the local market, of which about 40 came up after the promulgation of the announcement. However, quality certification and verification as well as labeling significantly lagged behind market development. According to sampling tests, many P-free laundry powders contained rather high levels of STPP and did not match the national quality standard. This evidence further justifies our doubt on the effectiveness of current market supervision in Dianchi Basin.

The third measure, i.e. public propagation, aimed at a wide acknowledgement and acceptance of P-free laundry powders among urban and rural residents. The main actions included broadcasting the announcement and education programs on television and radio, on-street dissemination of booklets and posters, and posting banners at public places. This measure closely related to the joint inspection, as they were implemented normally within the same period. As discussed above, however, a large amount of P-containing laundry powders still occupied local markets, while consumers' preferences directly stimulated and fostered market suppliers. This suggests a limited actual effect of public propagation. The findings of a local study further support this.[66] As shown in Table 6.13, 20.7% of the urban residents and 31.4% of rural residents had never heard about the announcement, and 43.0% of rural residences even did not acquire any knowledge on P-free laundry powders at all. The amount of residents

[66] This survey was conducted during 16 – 28 July 20001. The number of relevant samples was 980, covering all sub-districts around Dianchi Lake.

who knew the announcement but still insisted in their traditional washing behavior, reached remarkable numbers in both urban (12.7%) and rural areas (27.6%).

Table 6.13 Outcomes of the measure of propagation and education in Dianchi Basin

Individuals	Knowledge of the announcement (%)		Consumption of P-containing laundry powders while knowing the announcement (%)	
	Unknown	Known	Use	Un-use
Urban residences	20.7	79.3	12.7	87.3
Rural residences	31.4	68.6	27.6	72.4
average	26.1	73.9	19.2	80.8

Sources: based on Wang (2001).

To summarize, the local P detergent ban measures have not produced a reliable environmental effectiveness with a relative low inputs of administrative and human resources. It is can be concluded by the remarks of an anonymous official at KDAB:

> *"There is nearly no sale of P-containing laundry powders in Kunming city according to the results of our several inspective actions. However, it is really hard to estimate the amount on sale on the markets of a number of towns and villages around Dianchi Lake, as well as some underground markets… Lots of rural residents still prefer P-containing laundry powders because of relative low prices and better performance of decontamination… Actually, we have spent a lot of labors and money on each inspection, but we haven't succeeded with a reliable and sound achievement so far."*

6.5 Discussions and Conclusions

In this chapter, we have applied the EEA evaluation framework for selected policies on P flows associated with intensive livestock production, centralized sewage system and P detergents ban. Although the data gaps do not enable us to answer all proposed questions by the normative framework, the evaluation provides an insight in the successes and futures of China's policy responses to the three vital clusters of P flows. The evaluation results are summarized in Table 6.14.

SEPA's policy framework establishes a basic regulatory regime and forms an answer to the policy vacuum for environmental management of intensive livestock production in China. According to the *ex ante* evaluation, however, it is fair to say there exist some opportunities for future improvement of this national policy. The policy relevance is week due to its shortages in pursuing an integrated nutrient management that redirects all related P flows, particularly the inflow and recycling flow. The limited jurisdiction that SEPA has on the agricultural sector and the delayed schedule could further worsen the relevance of SEPA's policy across the country. The defects in policy priority and institutional capacity on this topic throughout the administrative system in China will significantly restrict the needed inputs of various kinds of administrative resources and the formulation and implementation of effective measures. The inputs, in terms of financial funds and professional staff, were still limited, in particular at the local level. The proposed key regulatory measures were limited to conventional approaches and end-of-pipe technical regimes. All these weaknesses in relation with policy design and implementation undermine the goal of a shift of environmental performance of intensive feedlots. Therefore, the effectiveness of SEPA's policy can be expected to be limited, a partial success in reducing the concentration of P from intensive livestock. The cost-effectiveness of the proposed livestock regulation is acceptable, as the inputs were rather low, especially at the local level, and the outputs and the outcomes too. The environmental outcomes and impacts can be improved if local EPBs successfully shift their focus and resources to intensive feedlots. While the national policy imposes barely influences on recovery and recycling of P, negative environmental impacts caused by disposal of the removed P at intensive feedlots can hardly lead to an increase of social welfare.

Table 6.14 Summary of evaluation results of P-related policies in China

P-related Policies	Relevance	Cost -effectiveness	Effectiveness	Welfare
Intensive livestock regulation	★☆☆ [a, b]	★★☆ [b]	★☆☆ [a, b]	★☆☆ [a, b]
Centralized WWT	★★☆ [a, b]	★☆☆ [a, b]	★☆☆ [a]/★★☆ [b]	★☆☆ [a, b]
P detergents ban	★★☆ [a]/★★★ [b]	★☆☆ [b]	★★☆ [a, b]	★★☆ [a, b]

Notes: ★★★ – sufficient; ★★☆ – moderate; ★☆☆ – insufficient. [a, b] The evaluation results are based on the evidences at the national level (*a*) and the local level (*b*).

The centralized sewage system is the dominant but not the only strategy for controlling P flows from urban households. While alternative source-separated control is receiving increasing attentions, urban sewage systems are still promising for both P removal and recovery, due to the continuing improvement of technologies and equipment. Its development in China as a whole, measured by both treatment rate of domestic wastewater and P-removal efficiency, lags behind that in advanced countries. Therefore, the national objectives of wastewater treatment rate and thus of P abatement are basically adequate, since construction and improvement of the vast infrastructure system cannot be done overnight. This justifies the policy's relevance of the centralized sewage strategy advocated by Chinese government. In Dianchi Basin, the more stringent local objectives, compared to the national ones, are relevant in relation to the local water quality. The centralized wastewater treatment strategy suffers from the huge investment demand. As a successful market-based financing of urban sewage systems is largely limited by some inherent barriers, the governmental fiscal expenditures remain insufficient to produce an effective environmental outcomes compared to the national objectives of P abatement. The development of the urban sewage system in Dianchi Basin is relatively successful and produces a more effective environmental outcome, due to central governmental subsidies and international funds. Combining the actual environmental outcomes with the huge investment, however, this centralized sewage system reduces P load to surface water with a dramatically low cost-effectiveness at the both national and local levels. Last but not least, sludge disposal, a vital element of the centralized sewage strategy, greatly undermines the overall welfare of P control at both national and local levels.

The third P control approach, i.e. P detergents ban, is the only measure in our analysis that aims at source control instead of end-of-pipe technologies. The policy's relevance comes up due to the ecological rationality that the strict objective of banning P consumption will lead to a reduction of P load, which meets the social needs for clean water. The actual environmental effects of these policies, with respect to the reduction of P in domestic wastewater and in surface water, however, are less effective compared to the common objective of P detergents bans. Regarding the local policy measures in Dianchi Basin, the environmental outcomes are limited but in principal moderate, while the results considerably vary among urban and rural areas. Due to large administrative inputs during enforcement, local P detergents ban measures become less cost-effective, although the expenditure from fiscal funds and human resources were annual variation in Dianchi Basin. As many negative social, economic and environmental impacts are introduced by STPP substitutes, it can be concluded that P detergents ban could lead to a remarkable – but not substantial – reduction of social welfare.

It is notified that none of three policy regimes achieves an overall success. Only the policy relevance of P detergents ban can be regarded as some kind of satisfied. More importantly, it is apparent that P, to a large extent, is still treated as the key pollutant responsible for eutrophication instead of a recoverable nutrient and mineral material. The substantial lack of systematical reflection on the entire P cycle, therefore, hardly generates sufficient incentives to contribute to a successful eutrophication control in China towards both ecological effectiveness and economic efficiency.

CHAPTER 7

Ecologizing Societal P Metabolism: Conclusions

7.1 Introduction

Phosphorous (P), intensively extracted from the natural sink in the lithosphere and processed through various production-consumption cycles, ultimately reaches the waterbody by different pathways to become a main cause of eutrophication, one of the key water problems in China. The P flows within the Chinese economy are characterized by complicated physical interconnections in high intensities among a number of production and consumption sectors. For this reason, regulating and controlling P flows in order to prevent eutrophication is a significant challenge in China. This challenge has justified the scientific rationality of this study: in dealing with eutrophication, this study explores the fundamental physical processes and flows of a society's P metabolism. Based on the explorations of the P flows in China's economy, solutions can be suggested and assessed.

This chapter starts by formulating the main findings in terms of the physical characteristics of P metabolism at the national and the local levels as well as the potential strategies for improving P throughput (section 7.2). In section 7.3, the Chinese national and local environmental policies that regulate P flows are evaluated, especially those associated with intensive livestock production, urban wastewater treatment and P-detergent consumption. Section 7.4 reflects on the methodology of this study and its wider applicability in China. The last section is dedicated to some final remarks on ecologizing P flows in a long-term perspective.

7.2 Physical Characteristics of P Throughput

7.2.1 The physical structure of P throughputs

In this study static substance flow analysis (SFA) models for P have been developed by quantitatively balancing and estimating unknown but critical flows in order to depict how a nation or a region extracts P from nature, metabolizes it via various economic activities, and finally excretes P as waste into the environment. The analysis of the P metabolic structure in this study shows a clear form: the societal P metabolisms, both within the national and local economies, have developed into a once-through system where P is mobilized in large quantities from nonrenewable rocks and subsequently leaks into either the soil (natural or cultivated) or surface water.

By constructing the National Static Phosphorus Flow Analysis Model of China (abbr. PHOSFLOW), the Chinese physical profile of P throughput was analyzed (cf. Figure 3.1 and 5.1). From an overall perspective on physical accounting, a set of aggregated flows can be calculated in relation to both the economy and environmental impacts (see Figure 5.3). On the input side, the direct material input (DMI) indicating the quantity of P inflow to the national economy was 5984 metric kilotons (mkt) P in the year 2000. Domestic extraction (DE) accounted for 82% of DMI. Together with the domestic hidden flow (DHF), which takes place during mining processes, the total material requirement (TMR) grew to 8159 mkt P. On the output side, the domestic processed output (DPO) into the domestic environment from various productive and consumptive activities added up to 5271 mkt P, of which 1075 mkt P entered the surface water. Import and export of commodities contributed to 1099 mkt P and 713 mkt P respectively for the Chinese economy and resulted in a net P import of 385 mkt P.

Each sector of the entire P production and consumption chain involved in the Chinese economy was independently discussed (see Figure 5.5). The chemical industries processed 4457 mkt P ores in 2000, which led to relatively slight impacts on surface water, but to significant environmental stress on natural land. Combining domestic products and import, the agricultural application of chemical fertilizers added up to 4348 mkt P, accounting for 73% of DMI. While the crop uptake of 3691 mkt P was harvested, the rest of the total inorganic and organic P application was either retained in cultivated soils, i.e. 2084 mkt P, or leached into surrounding waters, i.e. 259 mkt P. Large-scale intensive and small-scale family animal breeding consumed 1214 mkt P and 1067 mkt P from various sources, accounting for 20% and 18% of DMI, respectively. In addition to livestock production, animal husbandry as a whole was also responsible for recycling 1044 mkt P onto agricultural land, dumping 717 mkt P and discharging 387 mkt P into surface waterbodies. In particular, the intensive feedlots and farms contributed 74%, 72% and 33% to these P flows, respectively. A total of 1502 mkt P, consisting of 1401 mkt P in foods and 101 mkt P in detergents, was consumed by Chinese households, accounting for 25% of the national DMI. The disposal of human excreta and kitchen garbage significantly varied although a majority of the P generated from urban households ended up in landfills and that from rural households was mostly recycled onto farmlands. A small portion of the P contained in detergents used by urban residents were removed by the sewage systems and the rest, i.e. 98 mkt P, together with all those P consumed in rural areas, entered surface water.

In order to acquire comprehensive knowledge of the physical profile of P flows in China, the SFA approach was also applied at a regional level. Following the same analytical steps, the Regional Static Phosphorus Flow Analysis Model of the Dianchi Watershed (abbr. PHOSFAD) presents the fundamental physical characteristics of the local P throughput in both aggregate and sectoral terms for the year 2000 (cf. Figure 4.5 and 5.2). With respect to the aggregate structure of P throughput (see Figure 5.4), the DMI in this local economy reached 1025 mkt, which was dominated by a DE of 1022 mkt P. The DHF competed with local extraction by a huge magnitude of 954 mkt P, accounting for 48% of the TMR of P. Regarding the output side, 25% of the DMI comprised the local DPO by 253 mkt P and the rest was exported mainly in forms of raw ores and various chemical products. Only a small part of DPO, i.e. 1181 metric tons (mt) P, ultimately accumulated in the lake and then caused the eutrophication.

In terms of sectoral P throughput (see Figure 5.5), the local chemical industries were responsible for roughly 20% of the total domestic production of P chemicals. As most of these products were exported, the local consumption of the produced fertilizers and detergents, i.e. 4589 mt P and 370 mt P, respectively, accounted for only a small portion of the DMI. The application of chemical fertilizers and various manures contributed 1387 mt P to agricultural products and also resulted in 351 mt P leaching to Dianchi Lake and 5120 mt P depositing in the cultivated soils. The intensive and dispersive livestock consumed 552 mt P and 1927 mt P by various feeds, respectively, and then produced 484 mt P and 1534 mt P manures for recycling. The local sewage system collected 533 mt P in urban domestic wastewater, of which 377 mt P and 156 mt P were caused by food and detergents consumption, respectively. The not-removed P in a quantity of 138 mt, together with the uncollected urban wastewater and the widely scattered sources of rural households, contributed to the eutrophication of Dianchi Lake.

The normalization of these modified SFA models facilitates an understanding of the similarities and differences of the physical characteristics of P throughput between the national and the local economies. First, the local P throughput in Dianchi Basin was basically export-oriented but relied on domestic reserves. The local export of P accounted for 75% of DMI, 63% than that of the nation. Second, the local P fluxes were more intensive compared to those at the national level. This is indicated by the local DMI per capita which accounted for 460 kg P/capita, almost one hundred times the intensity of national gross input. Third, the import and export of agricultural goods was, in general, balanced at both the national and local levels, except for the crop trading in Dianchi Basin which led to a net import of P. Fourth, the increasingly widespread cultivation of vegetables and flowers in Dianchi Basin raised the P fertilizer application to 105 kg P/ha, three times the national average. This thus led to an increase in agricultural runoff of P. Fifth, the large-scale feedlots and farms in Dianchi Basin are comparatively underdeveloped. Their negative environmental impacts were rather limited since a major part of their animal waste was recycled to the farmlands. Sixth, the physical configurations of P flows associated with urban and rural residents were mostly similar between the national and the local level, the exception being the urban sewage system, which is more efficient in terms of P removal in Dianchi Basin.

The modification of the static SFA models, achieved by applying a set of data processing principles and new data series (see Chapter 5.1), demonstrates a gradual increase in the knowledge of P throughput at the national and the local levels, respectively. Arguably, the modified P fluxes as discussed above, compared to the previous accounts, are more similar to their counterparts in the natural cycle, although the data-induced and structure-induced changes in P loads to waterbodies by various sources are uncomfortably significant (see Table 5.1 and 5.2).

In analyzing the ecological restructuring of China, that is, examining the environmental effectiveness and economic efficiency of critical P flows throughout time, it was found that China has become more efficient in the use of P materials over the last two decades measured per unit of national economic growth. But the absolute quantities of these P flows have increased, which is of course especially relevant for the environment.

7.2.2 Structural change of societal P throughput: an outlook

The SFA models provide not only a systematic reflection on the physical characteristics of the current P throughput but also an analytical tool to discover the potential alternatives for solving the P-related ecological problems involving reserve depletion, ecological deterioration and eutrophication. One of the common starting points is to close the societal P cycle. A close-loop national P throughput is unrealistic, however, under the increasing trend of globalization and international trade in P ore export and P fertilizer import. This is also true for the local economy of Dianchi Basin due to the common regional exchanges of goods. Therefore, efforts to minimize the P inputs and outputs should be directed at improving the overall and sectoral material efficiency of P throughput and at increasing re-use and recycling. Taking eutrophication control into consideration, some general suggestions are discussed on the P flows in conjunction with the structural adjustment of agricultural production, improvement of material productivity, enhancement of manure application, greening of the human diet and the decentralization of the urban sewage system. It is fair to say that the preliminary discussion remains at a strategic level and thus demands further in-depth studies.

First, some adjustments to the overall metabolic structure in relation to agricultural production and commodities trading are desired. The basic idea is that an increase in the import of agricultural products would equally decrease the intensity of domestic production and thus the productive P load to the water environment. In other words, the domestic or local production can be ‘replaced’ by the import of corresponding products, although P is a non-substitutable nutrient. While this is not possible for the whole country, it could be advisable for Dianchi Basin where the cultivation of vegetables and flowers is to a large extent export-oriented. This justifies reducing local production if the economic losses could be bearable. For the same reason, it could be also interesting to balance the economic losses and ecological benefits caused by a decrease in intensive livestock. Moreover, it is important to be aware that a decline in livestock can also contribute to a reduction of feed grain supply and hence of P input to and runoff from farmlands.

Second, the material productivities of P can be improved, since they are pervasively low in both China and the case study region. The biological limits on nutrient absorptivity by crops, varying largely among different plant species, naturally restrict the P productivity of the farming sector. However, the currently high inputs of P fertilizers can certainly be lowered by relying on a variety of well-tested best management practices. An enhancement of P utilization is also expected for animal husbandry. Besides an overall improvement of feedlots and feed management, there is another option. The fact that a bulk of P (60-70%) in most cereal and leguminous grains is organically bound in phytic-acid and hence almost indigestible for monogastric animals (pigs and chickens) necessitates the use of P additives but results in large P losses in excreted manure. A more efficient feeding relies on addition of phytase and enhanced utilization of phytate, which could thus substantially reduce the P excretion.

Third, the issue of recycling animal and human manure needs more attention. The manure contains almost half of the DMI of P but only one- half of this P is reused for agricultural production at the national level. Although this is not a common case in Dianchi Basin, a significant nutrient gap of P between urban residents and farmlands is still visible. A successful manure reuse scheme would face a number of inherent challenges: the bulkiness and uneven distribution of the manure and the prohibitive cost of its application beyond a limited geographical radius. Since no available technologies are likely to be successfully commercialized in a short period of time, agricultural reuse remains the most promising way to recycle the huge amount of excreted P. Thus, it is vitally important to improve manure reuse in both the national and the local economies in order to reduce the overall P input and P losses.

Fourth, there is the possibility for greening the human diet. The nutritional status of Chinese people, measured by the daily ingestion of 1402 mg P/capita and 1470 mg P/capita for urban and rural residents, respectively, has exceeded the recommended daily reference ingestions (DRIs, cf. 1000 mg P/capita per day). Whereas animal food supplied 30% and 14% of the total P ingestion of urban and rural residents, respectively, a slight decrease in the consumption of livestock products would not lead to a large compromise of the current nutritional status. This could further decrease the need for P fertilizers without any investment because roughly 30% of all P fertilizers are used for producing feed grains in China. Consequently, the consumption-driven structural change in P throughput would considerably reduce the P input and P emissions from crop farming and animal husbandry as well as from human sources. However, the social acceptability and equality, the economic feasibility, and more importantly, the scientific evidence for the potential impacts of diet changes on nutrients balance and human health remain unclear.

Fifth, decentralization of the collective system of urban wastewater treatment could provide a new approach for China and especially for Dianchi Basin. It facilitates the recycling of P from human sources by separating excreta and urine at the source and consequently reduces the enormous investment in new central municipal infrastructure. Again, there exist many social, economic and institutional barriers that need to be removed.

7.3 Regulating the Societal P Metabolism

As the physical profiles of the societal P throughput are a result of human activities engaged in the production, consumption and treatment of P, the ecological restructuring of P flows can be achieved only by shifting the underlying socio-economic processes and dynamics. Hence, it is vitally important to connect the P flows with environmental regulations introduced to intervene in social practices and human behaviors. The SFA models enable the identification of environmentally critical P flows that should be the central object of concern for environmental regulation (see Chapter 5.7). The results have shown that the P flows associated with intensive livestock, the construction of urban centralized sewage and P-detergents are the appropriate candidates for this concern and that these issues also have the attention of environmental authorities in China.

In order to test how and to what extent the involved environmental policies have successfully (re)directed these P flows to generate environmental benefits for eutrophication control, an analytical framework proposed by the European Environment Agency (EEA) in 2001 for conducting environmental policy evaluation is applied. This method facilitates a systematic investigation into both the policy design in terms of the relevance of policy objectives and the policy implementation with regard to evaluating issues of economic efficiency, environmental effectiveness and overall social impact (see Table 6.14).

The increasing environmental impacts caused by the substantial transformation of Chinese animal husbandry were not a concern of the central government until this century. The nationwide regulatory framework, consisting of three documents on environmental administration, technical guidance and discharge standards, was initialized by SEPA in 2001. While this promoted the first important step in the regulation of intensive livestock husbandry, there are still some *ex ante* weaknesses, in particular the appropriateness of the policy's objectives and the potential environmental effects. The policy's objectives are largely unconcerned with a regional P balance, environmentally sound feeding and feasible manure reuse. Although these three options are vitally important for reducing environmental impacts, the regulation framework pays no attention to them. With respect to the enormous P emissions by the intensive livestock husbandry, the policy's relevance could also be compromised by its limited target group (only large-scale feedlots) and the agreed-on implementation schedule. Since the measures are basically dependent on a conventional regulatory approach and an end-of-pipe technical regime aims to reduce the P effluent from the livestock factories, the potential effects on both the behavior of intensive feedlots and on the environment could be limited, in particular for reducing P input and enhancing P recycling. The policy's effectiveness could further suffer from a low-level administrative priority, the traditional conflict between the development of the rural economy and the environmental regulation of intensive livestock in China, and a weak institutional capacity.

The intensive livestock regulation in Dianchi Basin faces a puzzle. Local regulation has been absent for quite some time, and the actual effects of existing local measures have been largely weakened by the limitation of the local administrative capacity, especially the capacity of environmental monitoring and managing environmental agencies at the county level. Meanwhile, the new national policy framework can hardly be applied in Dianchi Basin because of the limited target group of intensive feedlots. In particular, the spontaneous manure network in Dianchi Basin, driven more by the pursuit of local economics and than by environmental benefits, is responsible for a huge amount of the P recycling loop and thus contributes to limiting P leaking to the lake. However, some inherent weaknesses in the environmental effects of animal manure application challenge an integrated P management.

Unfortunately, this exclusive manure network has not been taken into consideration by either the national policy framework or the current local livestock regulation.

The centralized P control strategy, parallel to the construction of urban sewage infrastructures, has been advocated by Chinese government. The combination of P removal efficiencies of different treatment technologies and the government- proposed wastewater treatment rates enables the national P control objectives to reduce by 23%, 34% and 45% of the total domestic P in 2000, 2005 and 2010, respectively. These policy objectives are, in general, relevant to the urgent need for curbing eutrophication. But they are also somewhat ambitious with respect to the underdevelopment of urban sewage systems and the low level application of treatment technologies in China. A more realistic expectation is presented in the new national standard stipulated in 2002. This current regulatory measure to limit P concentrations in the effluent of WWTPs is less stringent compared to its predecessor, especially for existing WWTPs. Taking the actual environmental outcomes of 2000 into account, the policy effectiveness in terms of the P abatement rate is discounted by 65%. If measured by the demand on investment in the urban sewage infrastructures, the limited achievements of the environmental outcomes by the centralized P control approach becomes even less cost-effective. More importantly, this costly and rigid P control strategy by centralized wastewater treatment can hardly contribute to a real reduction of the overall environmental impacts of P, as recovery and recycling of P in the form of bulk sewage sludge is difficult.

The urban centralized P control in Dianchi Basin has shifted to a modern mode, as indicated by a higher wastewater treatment rate and more advanced technology. The local government initially proposed a more stringent objective compared to its national counterpart, i.e. an 80% urban wastewater treatment rate in 2000. Compared with the actual environmental outcomes in that year, the policy is 18% less effective than the expectation. Acknowledging the actual achievement, a more realistic objective was subsequently introduced by the local government who proposed the same rate for 2005. The total expenditures in the sewage infrastructure in Dianchi Basin, mainly financed by the central government and the World Bank, lead to a low cost-effectiveness of the centralized P control approach. While large financial aid could contribute to a promising environmental outcome in terms of P removal, it could also result in an increase in the P flow in the form of sludge. Since the agricultural reuse of the sewage sludge has not been practiced in Dianchi Basin, the social welfare measured by the overall environmental impact of the centralized P control remains questionable.

The third cluster of the environmental policy, i.e. the ban on P-detergents, has not become a common item on the political agenda for the whole country yet. The summary of various local empirical studies enables the evaluative questions to be answered at the national level although the costs of implementing the policy remain unknown. As the only strategy in our analysis aiming at source control, these policies prohibit the production and consumption of P-containing detergents in most local practices. Since the detergents containing P contributed roughly 10% to the total national P load to surface water in 2000, this strict objective is relevant for pursuing clean surface water. However, none of the local bans on P-detergents in China has produced a reliable environmental effect measured by the P concentration in surface waterbodies, while the P concentration in domestic wastewater varied considerably. While the effectiveness of these policies could be called questionable, the undesired decline of social welfare would occur at the national level because of a number of negative social and economic impacts. These impacts involve losses for STPP suppliers, additional costs of substitute production, the low acceptability of P-free detergents due to their poor washing performance, the challenge to market supervision and enforcement, and the negative environmental impacts of the STPP substitutes. A complete ban of P-detergents has been promoted by the local government in Dianchi Basin since 1998. The fact that the P contained in detergents contributed to 22% of the total P load in the lake in 2000 justifies the relevance of the policy's objective. Three measures, consisting of unannounced inspection, market supervision and public propagation, have been introduced to put the policy into practice. The actual outcomes are remarkable as local production has successfully shifted to P-free ingredients. But meanwhile the behavior of wholesalers and both urban and rural residents needs to be further improved. These outcomes, achieved by a large input of human and administrative resources, necessitate a further reduction in the considerable implementation costs. Compared to the initial objective, the environmental effects can be expected to be limited since no local evidence causally links the ban-induced reduction of P in domestic wastewater to the water quality of the lake.

Although data insufficiency limits a final evaluation, there are some general comments on the current environmental regulations in China from the perspective of societal P metabolism. The evaluated environmental policies demonstrate two main societal responses to P- caused eutrophication: an endeavor to prevent P from escaping to the waterbodies, and an attempt to reduce P use at the household source. However, end-of-pipe approaches that remove P from effluence to a place where P is less likely to harm the water environment do not necessarily result in reducing the P input or increasing P recycling, both of which are underlying propositions of an ecologically rational eutrophication policy. At the same time, the current and potential environmental effects of these approaches in terms of reducing P outflows are limited, due to a large number of economic, social and institutional barriers. The ban of P-detergent – a source-control strategy – could considerably reduce the P input in the economy. As reduced P consumption depends on introducing substitute materials, this policy could result in undesired negative impacts on the whole society and the environment. In this sense, the P control approach at source could become less promising. Furthermore, the ban, like the other two end-of-pipe strategies, relies mainly on government-oriented regulatory measures, while other measures, especially the market-based economic incentives, are to a large extent ignored.

7.4 The Methodology and Its Application: A Revisit

7.4.1 Advantages and limitations of the methods

This study has developed a comprehensive research approach for inquiring into the material metabolism of P in human society. The methodology starts with a quantitative analysis of the entire life cycle of societal P metabolism, identifies from that the critical P flows with crucial environmental consequences and relates them to an evaluation of the environmental policies directed at these P flows.

SFA approaches have been developed as an analytical tool to gain insight into the physical profile of substance metabolism in human society. By abstracting substances from their common forms in bulk material flows, SFA models contribute to a precise observation and analysis of the societal flows of a certain substance, which could generate innovative perspectives that often differ from a common- sense view or a result based on conventional environmental studies (cf. Chapter 2.2). The physical-based approaches attempt to connect environmental problems to their economic origins and thus to provide a better understanding of how and where the societal substance throughput has resulted in environmental problems. The potential solutions can then be identified, directly oriented at an improvement of the identified structural flaws of the substance throughput in the economy.

Some methodological flaws of the SFA approaches limit their role as a tool for decision-making. First, as SFA explores the whole economy from a physical point of view, some underlying information related to substance flows can be missed. Although economic distribution, social behavior and environmental regulation provide the necessary background information for constructing SFA models, SFA studies can hardly offer valuable feedback to (re)direct such factors as determining substance emission or distribution coefficients (e.g. Table 3.3). Second, an SFA model points to the economic origins and environmental termination of substance flows, but it fails to connect with an environmental quality model to reveal actual or potential environmental impacts. Third, SFA models – operating in various ways – do not include a normative assessment of the main findings by themselves. Although the aggregate structure and sectoral efficiency of P flows are distinguished in this study, the standardization of SFA models and their policy implications remains a challenge.

The applied environmental policy evaluation methodology forces a systematic collection, organization and analysis of information of the whole policy process. The evaluating process itself contributes to a comprehensive survey of the quality and quantity of current environmental data and thus facilitates the establishment of a logical and complete database in relation to environmental policy design and implementation. The normative evaluation framework also makes it possible to compare responses to similar environmental problems among countries (e.g. European countries) or among different regions within one country. Such comparative studies facilitate future improvements in policy design by taking empirical evidence and experiences from different socio-economic and regulatory backgrounds into account. This becomes significantly important when investigating and evaluating the environmental policies and regulations for China, the largest developing county in economic and ecological transition.

The applicability of the evaluation framework for evaluating cost-effectiveness could be questionable when dealing with substance-related environmental policies, for instance, regulating P flows. In general terms, P causes environmental problems in forms of bulk material. When disconnected from other materials and substances, P flows can be mathematically accounted for, which contributes to an in-depth understanding of the physical structure of P throughput. However, if we aim to measure the economic efficiency of P throughput or P policies, such disconnection may lead to rather vague results for a cost-effectiveness evaluation. Except for the ban on detergents containing P, P is not the exclusive target of the evaluated environmental policies. Consequently, it is difficult to break down the actual or potential costs of various pollutant control measures and directly relate them to P. The abatement of P generated from urban households, for instance, depends on the centralized sewage systems, which demand huge investments in wastewater infrastructures. P is removed or recycled together with N, another indispensable nutrient and causal factor for eutrophication. Accordingly, in most cases the cost-effectiveness of policy measures can be examined only by combining the expenditures on the control of various pollutants. This does not reflect the actual economic profile of P regulation, and conclusions on cost-effectiveness of P-abatement can not be drawn easily.

The application of a static SFA approach, followed by an environmental policy evaluation methodology, has further demonstrated some advantages because, when combined, each approach overcomes the other's flaws. Although the physical-based approach can analytically offer potential options for environmental improvement, it cannot evaluate the various options in terms of social, political and economic feasibility (cf. section 7.2.2 in this chapter). The applied policy evaluation methodology, while certainly not the final word, adds much to limit these shortcomings. Second, since the SFA approach is exclusively dedicated to P, the impacts of other related materials/substances that are introduced in the environment in reducing P use are often entirely ignored. The substitution materials following a P-detergents ban, for instance, are not introduced in the SFA model but are unlikely to be more acceptable because of their higher negative environmental impacts compared to STPP (see Chapter 6.4).

At the same time an environmental policy evaluation profits from the main findings of SFA models. First, since there are many potential and actual material- or substance-related environmental policies (in China but also elsewhere), it is nearly impossible to evaluate them all. A model-based assessment of the P load by different sources to the waterbody enables evaluators to focus on a few clusters of environmental policies related to the crucial P flows (see Chapter 5.7). This contributes to an in-depth and focused evaluation instead of a superficial policy analysis. Second, the understanding of physical interconnections of P among production, consumption and waste treatment sectors fosters a systematic perspective of the environmental policy evaluation. The steps to fulfill the social need for clean water can then be evaluated by focusing on ecologically 'rational' regulations, and not just end-of-pipe measures focusing on the last discharge of P. Accordingly, the evaluation can also look into the current environmental policies in terms of pollution prevention and waste recycling, which are vital for future policy (re)design and implementation.

The methodology followed in this study, however, has not been completed. It proved difficult to further or even fully integrate these two methods. Apparently, the methods are running separately to a large extent, with some logical linkages as discussed earlier. One would have liked, for instance, a stronger and valuable feedback of the policy evaluation into the SFA models. Investigations into the potential for stronger linkages of these two – and perhaps additional – methodologies is required.

7.4.2 Applicability of this methodology for China

The application of this methodology has successfully produced two P flow models and a normative evaluation of three clusters of P-related environmental policies, which can be understood as an innovation in contemporary research on P in China. The development of static SFA models has for the first time provided an insight into the material throughput in term of P in China and the local economy of Dianchi Basin. This provides a scientific basis for reinventing the eutrophication control framework at both the national and local levels. Implementing a systematic policy evaluation, which differs from program evaluation or project evaluation (e.g. environmental impacts assessment), which have been and are still predominantly applied in contemporary China, has also contributed to a better understanding and assessment of the national and local environmental policies on P. By introducing the substance flow analysis method and the policy evaluation method into China, this study offers a promising research paradigm and thus helps Chinese researchers to become more actively involved in the area of material metabolism, one of the fastest growing international scientific fields.

The data do matter both for the construction of the SFA models and for the implementation of environmental policy evaluation (see Chapter 5.1 and Chapter 6 elsewhere). Although data quantity and quality are commonly two of the main difficulties in yielding robust findings, the situation is even worse at the current national and the local databases in China. The insufficient and sometimes inconsistent statistics and data required large investments in time and labor in this research. Moreover, the accessibility of crucial information, especially on environmental performance as well as environmental quality, proved rather difficult, which in turn undermines the reliability of the SFA models as well as the environmental policy evaluation. Therefore, there is an urgent need to improve the data gap and data reliability at both the national and local levels to enable more feasible research schemes and more tangible and reliable outputs.

Undoubtedly, this study can be regarded only as a starting point for inquiries into the societal P metabolism in China. It became apparent that the application of this methodology is a time-consuming process, mainly because of the collection, organization, verification and assessment of the large amounts of environmental and socio-economic data from various sources (see Chapter 2.5, Chapter 5.1 and Chapter 6.1). Together with the existing data problems, this justifies suggesting a number of directions for subsequent studies. First, both the static SFA models, in particular the P flows associated with the subsystems of intensive livestock husbandry and dispersive family breeding in the PHOSFLOW model and those in commodities import and export in the PHOSFAD model, can be further modified and verified. But, with respect to current data conditions, it will prove to be rather difficult to apply dynamic SFA approaches for P flows in China, both at the national and local levels. Second, policy evaluation should be further elaborated based on more empirical evidence of the causal relationship between environmental effects and environmental regulations, especially the implementation of the national new policy framework for intensive livestock regulation. Third, some other crucial P flows for eutrophication, especially agricultural runoff, should be taken further into consideration in policy evaluation in order to test the various effects and impacts of existing agri-environmental regulations. Fourth, an *ex ante* evaluation of the model-based proposals for the expected structural change is strongly recommended. This could largely benefit the design of ‘rational’ policy scenarios, the introduction of new feasible policy measures and policy-induced reliable outcomes in practice.

Besides providing information on eutrophication, this study can generate interesting contributions to other environmental issues through its illumination of the physical characteristics of P flows in China. The issue of sustainable resource use, for instance, originating from a conflict between the gradually increasing P extraction and the finite non-renewable mineral reserve, is one of the pivots. While all options for minimizing P input discussed throughout this study contribute to preventing reserve depletion, more efforts to promote the material productivities of the mining and industrial sectors are desired. With respect to the huge amount of P accumulation in natural soils indicated by the DHF and industrial waste, re-mining from waste P tailings and P gypsums could become potential measures. Furthermore, some concrete findings of this study could help to support the arguments of grain security and agricultural structural adjustment in China.

The comprehensive methodology applied in this study can prove valuable to investigate the societal metabolisms of other substances and materials within the whole economy or individual societal sectors in China. In doing so, a few methodological revisions are needed, such as redefining a system boundary of physical flows, specifying the appropriate type of model, choosing the most suitable form of interpreting results and, most importantly, devising ways to collect and process the data. This study also suggests that this methodology can be successfully operationalized on different spatial scales, from a national to a local level. The latter application, causally bridging the local economic activities with environmental consequences, is more suitable for solving environmental problems on-site. This is vitally important in dealing with the puzzle that occurs in national environmental actions regarding the “Three Rivers and Three Lakes”, the key environmental protection areas in China.

Finally, this study can actively engage in the ongoing environmental campaign to establish research as a priority in the idea of a circular economy in China. The concept of the circular economy, advocated by Chinese governments, suggests an ideal mode of sustainable development. Similar to the main propositions of industrial ecology, it emphasizes an environmental optimization of material metabolism within the booming economy via ‘reduce’, ‘reuse’ and ‘recycle’ strategies (3R principles). As the Chinese practice of the circular economy to a large extent stagnates at an early stage of theoretical arguments, this study could appropriately position itself in – in both methodological and practical terms – a development of reliable analytical tools and a reinvention of feasible environmental policies in China.

7.5 Epilogue: Towards a Gradual Restructuring of P Flows

Only a small amount of the enormous gross turnover of P terminates in water but it then dominates undesirable eutrophication. As P leakage takes place in all involved societal sectors, it could be extremely difficult as well as expensive to intercept all P effluents that potentially harm the surface water by conventional end-of-pipe approaches. Instead of continuously trying to limit the growth of P, therefore, there is a great need to reconstruct the physical structure of P flows, in particular by redirecting the crucial P flows with highly negative environmental impacts. The ecological restructuring of the current one-through mode of societal P metabolism is thus desired, leading to a structural shift in the societal production and consumption of P flows. The ecologically rational switch can contribute to a substantial decline of P outflows by minimizing P input and maximizing P recycling. This could be more promising for curbing water eutrophication.

However, ecologizing societal P flows can by no means be achieved simply by copying the natural mode of the P cycle. In that sense, industrial ecology is a mis-learning term. A reconfiguration of material flows should take all those related social, economic, technological and cultural configurations into consideration. Among other things, economic feasibility, ecological capacity and social acceptability should be given a central place, something that a policy evaluation methodology helps to do. When ecological restructuring significantly challenges the existing environmental regulations on P flows in China at both the national and local levels, there is an urgent need to (re)invent and (re)formulate smart environmental policies and measures, taking the geographical variety of P flows and all underlying socio-economic barriers into consideration. Since ecologizing the P flows only succeeds when measures are institutionalized into the economy and society as a whole, this process will most likely be a gradual one rather than a radical revolution, which, by the way, fits into the current dominant Chinese ideology.

References

Adriaanse, A., S. Bringezu, A. Hammond, *et al.* 1997. Resource Flows: The Material Basis of Industrial Economies. Washington, D.C.: World Resources Institute.

Amann, C., W. Bruckner, M. Fischer-Kowalski, *et al.* 2002. Material flow accounting in Amazonia: A tool for sustainable development. *Social Ecology Working Paper 63*, IFF.

Anderberg, S. 1998. Industrial metabolism and the linkages between economics, ethics and the environment. *Ecological Economics* 24 311-320.

Andrews, C. J. 1999. Putting industrial ecology into place. *Journal of the American Planning Association* 65 (4): 364-375.

Anonymous. 1995. Sodium tripolyphophate - The green choice? *Phosphorus and Potassium* 23-26.

Anonymous. 2004. The development of urban water and wastewater infrastructures and related technological policies in China. *Information on China construction* (5). Available at http://www.jsbwater.com/200405 /20040530.htm.

Ayres, R. U. 1994. Industrial metabolism: theory and policy. In *Industrial Metabolism: Restructuring for Sustainable Development*, edited by R. U. Ayres and U. E. Simonis, p:3-20. Tokyo/New York/Paris: United Nations University Press.

Ayres, R. U. 2004. On the life cycle metaphor: where ecology and economics diverge. *Ecological Economics* 48 (4): 425-438.

Ayres, R. U., and A. V. Kneese. 1969. Production, consumption and externalities. *American Economic Review* 59 282-297.

Ayres, R. U., and L. W. Ayres. 1996. *Industrial Ecology: Towards Closing the Materials Cycle*. Cheltenham: Edward Elgar.

Ayres, R.U. 1992. Toxic heavy metals: material cycle optimization. *The National Academy of Sciences of the USA* 89(3): 815-820.

Ayres, R.U., and U.E. Simonis. 1994. *Industrial Metabolism: Restructuring for Sustainable Development*. Tokyo/New York/Paris: United Nations University Press.

Baldock, D., J. Dwyer, and J. M. S. Vinas. 2002. Environmental Integration and the CAP: A Report to the European Commission, DG Agriculture. London: Institute for European Environmental Policy (IEEP).

Bartelmus, P. 2002. Dematerialization and capital maintenance: Two sides of the sustainability coin. *Wuppertal Papers No.120 (ISSN 0949-5266).* Wuppertal, Germany: Wuppertal Institute.

Bartelmus, P., and A. Vesper. 2000. Green accounting and material flow analysis: Alternatives or complements? *Wuppertal Papers No.106 (ISSN 0949-5266).* Wuppertal, Germany: Wuppertal Institute.

Beck, M. B. 1987. Water quality modeling: A review of the analysis of uncertainty. *Water Resources Research* 23 (8): 1393-1442.

Beck, M. B. 1997. Applying systems analysis in managing the water environment: towards a new agenda. *Water Science and Technology* 36 (5): 1-17.

Bellamy, J. A., D. H. Walker, G. T. McDonald, *et al.* 2001. A systems approach to the evaluation of natural resource management initiatives. *Journal of Environmental Management* 63 407-423.

Bengtsson, M., and A.-M. Tillman. 2004. Actors and interpretations in an environmental controversy: the Swedish debate on sewage sludge use in agriculture. *Resources, Conservation and Recycling* 42 (1): 65-82.

Berkhout, F. 1999. Industrial metabolism: Concept and implications for statistics. *Eurostat Working Paper No. 2/1999/B/2*, European Commission.

Berndtsson, J. C., and I. Hyvonen. 2002. Are there sustainable alternatives to water-based sanitation system? Practical illustrations and policy issues. *Water Policy* (4): 515-530.

Bian, Y. S., ed. 2000. *Treatment and Recovery of Wastes in Ecological Agriculture*. Beijing: Chemical Industry Press. (in Chinese).

Blöch, H. 2001. EU policy on nutrients emissions: legislation and implementation. *Water Science and Technology* 44 (1): 1-6.

Bohm, R. 2001. Public health aspects connected with agricultural utilization of sewage sludge. Sewage Sludge Disposal: Sustainable and/or Reliable Solutions, p.45-59, *EWA/OWAV Workshop*, at Vienna. IWAG, TU-Wien.

Bouman, M., R. Heijungs, E. Van der Voet, *et al.* 2000. Material flows and economic models: an analytical comparison of SFA, LCA and partial equilibrium models. *Ecological Economics* 32 195-216.

Bowker, R. P. G., and H. D. Stensel. 1990. Phosphorus removal from wastewater. *Pollution Technology Review No.189*, Noyes Data Corporation, USA.

Brett, S., J. Guy, G. K. Morse, *et al.* 1997. Phosphorus Removal and Recovery Technologies. Brussels, Belguim: CEEP (Centre Europeen d'Etudes des Polyphosphates E.V.).

Bringezu, S. 1999. Industrial ecology and material flow analysis: Basic concepts, policy relevance and some case studies. *The international symposium "Industrial Ecology and Sustainability"*, at Technical University of Troyes, France.

Bringezu, S. 2000. History and overview of material flow analysis. *Special Session on Material Flow Accounting*, OECD Working Group on Environmental Information and Outlooks (WGEIO), p:13-23, at Paris.

Bringezu, S. 2001. Material flow analysis for the European Union and beyond: Implications for statistics and policy. *ISIE Conference: The Science and Culture of Industrial Ecology*, at Nordwijkerhood.

Bringezu, S. 2002. Towards sustainable resource management in the European Union. Wuppertal, Germany: Wuppertal Papers No.121 (ISSN 0949-5266), Wuppertal Institute.

Bringezu, S., and H. Schütz, 2001. *Total material requirement of the European Union*. Technical Report No. 26. Copenhagen: European Environmental Agency.

Brown, L. R. 2001. *Eco-Economy: Building an Economy for the Earth*. New York: W. W. Norton & Company.

Brown, L. R., and B. Halweil. 1998. China's water shortage could shake world food security. *World Watch* 11 (4): 10-21.

Brunner, P. H. 2001. Sewage sludge and regional materials management. Sewage Sludge Disposal: Sustainable and/or Reliable Solutions, p.33-44, *EWA/OWAV Workshop*, at Vienna. IWAG, TU-Wien.

Brunner, P. H., H. Daxbeck, and P. Baccini. 1994. Industrial metabolism at the regional and local level: A case-study on a Swiss region. In *Industrial Metabolism: Restructuring for Sustainable Development*, edited by R. U. Ayres and U. E. Simonis, p:163-193. Tokyo/New York/Paris: United Nations University Press.

Button, K., and P. Nijkamp. 1997. Environmental policy assessment and the usefulness of meta-analysis. *Socio-Economic Planning Science* 31 (3): 231-240.

Cai, C. F., S. W. Ding, Z. H. Shi, L. Huang, and G. Y. Zhang. 2001. Prediction on soil nutrients losses at typical small watershed of Three Gorges Area with GIS. *Journal of Soil and Water Conservation* 15(1): 9-12. (in Chinese).

Cao, Y. 1997a. Review of researches on environmental impacts of STPP (I): introduction to STPP. *Detergent and Cosmetics* (5): 29-31. (in Chinese).

Cao, Y. 1997b. Review of researches on environmental impacts of STPP (II): EU. *Detergent and Cosmetics* (5): 30-33. (in Chinese).

CEEP. 1997. *Phosphate*. Brussels, Belgium: Centre Europeen d'Etudes des Polyphosphates (the West European Phosphate Industry's Joint Research Association).

CEEP. 1998a. Phosphates: a sustainable future in recycling. Brussels, Belgium: Centre Europeen d'Etudes des Polyphosphates (the West European Phosphate Industry's Joint Research Association).

CEEP. 1998b. Press releases concerning phosphates and detergent wash performance. Brussels, Belgium:

Centre Europeen d'Etudes des Polyphosphates (the West European Phosphate Industry's Joint Research Association).

CEQ. 1997. The National Environmental Policy Act: A Study of Its Effectiveness after Twenty-five Years. Council on Environmental Quality, Executive Office of the President, USA.

CG, DESE, and ERM. 2004. Comprehensive Report on Key Problems and Underlying Causes of Rural NSP in China. *Study on Control and Management of Rural Non-Point Source Pollution, TA No.3891-PRC*. Mandaluyong (Philippines): Asian Development Bank.

Chang, J. I., C. H. Chen, and G. S. Y. Liu. 2000. The experience of waste minimization at a nitrocellulose manufacturing plant. *Resources, Conservation and Recycling* 30 333-351.

Chen, F. 2002. *Agricultural Ecology*. Beijing: China Agriculture University Press. (in Chinese).

Chen, F., ed. 2002. *Agricultural Ecology*. Beijing: China Agricultural University Press. (in Chinese).

Chen, J. F., and G. X. Tan, eds. 1989. *Production and Application of Phosphate*. Chengdu: Chengdu Science and Technology University Press. (in Chinese).

Chen, J. N., and T. Z. Zhang. 2003. General Plan for Ecological Economy and Urban Development of Guiyang Municipality in Southwestern China. Beijing: Department of Environmental Science and Engineering, Tsinghua University.

Chen, J., and M. B. Beck. 1997. Towards designing sustainable urban wastewater infrastructures: a screening analysis. *Water Science and Technology* 35 (9): 99-112.

Chen, L. K. 2000. Problems in the development of phosphate rock resources in Yunnan province and the suggestions. *IM&P* (5): 1-4. (in Chinese).

Chen, X. Q., and L. J. Qiao. 2001. A preliminary material input analysis of China. *Population and Environment* 23 (1): 117-126.

Chen, X., Z. Q. Wang, W. D. Yang, and X. J. Ye. 2000. Effects of sloping land use patterns on phosphorus loss in a micro-watershed of red soil area, Southern China. *Acta Ecological Sinica* 20(3): 374-377. (in Chinese).

Chen, Z. L., and Y. Z. Tang. 1998. Study on development and drawbacks of urban night-soil treatment and disposal system in China. *Environment and Hygiene Engineering* 6 (3): 125-131. (in Chinese).

Cheng, G. Q., Y. H. Zhou, J. M. Wang, *et al.* 1997. Estimates on supply and demand of feed in China. *Issues in Agricultural Economy* (5): 25-29. (in Chinese).

Chess, C. 2000. Evaluating environmental public participation: Methodological questions. *Journal of Environmental Planning and Management* 43 (6): 769-784.

Chu, J. Y., J. N. Chen, J. Zou, *et al.* 2004. Comparative study on resource allocation efficiency of urban wastewater treatment plants in China. *China Environmental Science* 24 (2): 242-246. (in Chinese).

Clark, L. L., E. D. Ingall, and R. Benner. 1998. Marine phosphorus is selectively remineralized. *Nature* 393 (6684): 426.

CNCIC, ed. 2002. *China Chemical Industry Yearbook 2001-2002*. Beijing: China National Chemical Information Center.

CNEMC. Monthly report on water quality of Taihu Lake. Oct. 1998 – Jan. 2001.

CNLIA, ed. 2001. *China Light Industry Yearbook 2001*. Beijing: Society of China Light Industry Yearbook.

Cook, E. 1971. The flow of energy in an industrial society. *Scientific American* 224 (3): 135-144.

Costanza, R., L. Wainger, C. Folke, *et al.* 1993. Modeling complex ecological economic systems: toward an evolutionary, dynamic understanding of people and nature. *BioScience* 43 (8): 545-555.

Côté, R. P., and E. Cohen-Rosenthal. 1998. Designing eco-industrial parks: a synthesis of some experiences. *Journal of Cleaner Production* 6 (3-4): 181-188.

Côté, R. P., and S. W. Peck. 2002. Eco-industrial networking: A typology with examples. *Partnering for Sustainability*, at Toronto, Ontario, April 8.

Cropper, M. L., and W. E. Oates. 1992. Environmental economics: A survey. *Journal of Economic Literature* XXX 675-740.

Cui, Y. T., ed. 2000. *Fertilizers and Ecological Environmental Protection*. Beijing: Chemical Industry Press. (in Chinese).

Czemiel, J. 2000. Phosphorus and nitrogen in sanitary systems in Kalmar. *Urban Water* (2): 63-69.

Daniels, P. L., and S. Moore. 2001. Approaches for quantifying the metabolism of physical economies - part I: methodological overview. *Journal of Industrial Ecology* 5 (4): 69-93.

Daniels, P.L. 2002. Approaches for quantifying the metabolism of physical economies: A comparative survey, part II: Review of individual approaches. *Journal of Industrial Ecology* 6(1): 65-88.

Daskalopoulos, E., O. Badr, and S. D. Probert. 1997. Economic and environmental evaluations of waste treatment and disposal technologies for municipal solid waste. *Applied Energy* 58 (4): 209-255.

De Clercq, P., A. C. Gertsis, G. Hofman, *et al.*, eds. 2001. *Nutrient Management Legislation in European Countries*. Wageningen, The Netherlands: Wageningen Press.

De Haan, F. A. M., and S. E. A. T. M. Van der Zee. 1994. Soil protection and intensive animal husbandry in the Netherlands. *Marine Pollution Bulletin* 29 (6-12): 439-443.

Decker, E. H., S. Elliott, F. A. Smith, *et al.* 2000. Energy and material flow through the urban ecosystem. *Annual Review of Energy and Environment* 25 685-740.

Delgado, C., M. Rosegrant, H. Steinfeld, *et al.* 1999. *Live Stock to 2020: The Next Food Revolution*. Washington, D.C.: International Food Policy Research Institute.

Deng, C., and P. G. Li. 2002. The application of substance flow analysis in environment. *Environmental Technology* (4): 45-49. (in Chinese).

DESE, KIES, and KWHIDI. 2003. Plan on Ecological Protection for Dianchi Lake Basin. Beijing: Department of Environmental Science and Engineering, Tsinghua University. (in Chinese).

Dieu, T. T. 2003. Greening Food Processing Industry in Vietnam: Putting Industrial Ecology to Work. *PhD thesis*, Environmental Policy Group, Department of Social Science, Wageningen University, Wageningen.

Dijk, J., H. Leneman, and M. van der Veen. 1996. The nutrient flow model for Dutch agriculture: A tool for environmental policy evaluation. *Journal of Environmental Management* 46 43-55.

Ding, J. H. 2000. The pollution of poultry and animal feces and the countermeasures in Guangzhou. *Research of Environmental Sciences* 13 (3): 57-59. (in Chinese).

Dokulil, M., W. Chen, and Q. Cai. 2000. Anthropogenic impacts to large lakes in China: the Tai Hu example. *Aquatic Ecosystem Health and Management* 3 81-94.

Dongye, M. X. 1996. Circulation and precipitation of phosphorus in oceanic basins. *Geology of Chemical Minerals* 18 (3): 191-195. (in Chinese).

Driver, J. 1998. Phosphates recovery for recycling from sewage and animal wastes. *Phosphorus and Potassium* 216 (Jul/Aug): 17-21.

Dryzek, J. S. 1987. *Rational ecology. Environment and political economy*. Oxford, UK: Basil Blackwell.

Du, P. F., W. T. Du, and Y. Qian. 2000. Primary and secondary municipal wastewater treatment cost comparison. *Journal of Tsinghua University (Sci & Tech)* 40 (4): 73-76. (in Chinese).

Duan, Y. H., and N. M. Zhang. 2003. Analysis on current status of rural area non-point pollution in Dianchi Lake Watershed. *Environmental Protection* (7): 28-30. (in Chinese).

Duchin, F. 1992. Industrial input-output analysis: implications for industrial ecology. *The National Academy of Sciences of the USA* 89(3): 851-855.

Eder, P., and M. Narodoslawsky. 1999. What environmental pressures are a region's industries responsible for? *Ecological Economics* 29 357-374.

EEA. 1997. Sludge Treatment and Disposal: Management Approaches and Experiences. *Environmental Issues Series No.7*. Copenhagen: European Environment Agency.

EEA. 1998. Europe's Environment: The Second Assessment. Copenhagen: European Environment Agency.

EEA. 2001. Reporting on environmental measures: Are we being effective? *Environmental Issue Report No.25*. Copenhagen: European Environment Agency.

EEA. 2002. Energy and environment in the European Union. *Environmental Issue Report No.31*. Copenhagen: European Environment Agency.

EEA. 2003a. Europe's water: an indicator-based assessment. *Topic report 1/2003*. Copenhagen: European Environment Agency.

EEA. 2003b. Europe's environment: the third assessment. *Environmental assessment report No.10*.

Copenhagen: European Environment Agency.

Elliott, E. D. 1997. Toward ecological law and policy. In *Thinking Ecologically: the next generation of environmental policy*, edited by M. R. Chertow and D. C. Esty, p:170-186. New Haven: Yale University Press.

Erisman, J. W., W. de Vries, H. Kros, *et al.* 2001. An outlook for a national integrated nitrogen policy. *Environmental Science & Policy* 4 (2-3): 87-95.

Erkman, S. 1999. *Industrial Ecology.* (Chinese version). Translated by X. Y. Xu. Beijing: Economic Daily Press.

ERM. 1999. The policy relevance of material flow accounts. *Eurostat Working Paper No. 2/1999/B/1*, European Commission.

Esrey, S. A., J. Gough, D. Rapaport, *et al.* 1998. Ecological Sanitation. Stockholm: Swedish International Development Cooperation Agency (SIDA).

European Parliament and the Council of the European Union. 2002. Decision No 1600/2002/EC of the European Parliament and of the Council of 22 July 2002 Laying Down the Sixth Community Environment Action Program. *Official Journal of the European Communities* L242 (45): 1-15.

Eurostat. 2000. Economy-wide Material Flow Accounts and Derived Indicators: A Methodological Guide. Luxembourg: Statistical Office of the European Communities.

Eurostat. 2001. Material use indicators for European Union, 1980-1997. *Eurostat Working Paper No. 2/2001/B/2*. Luxembourg: Office for Official Publications of the European Communities.

Eurostat. 2002. Material Use in the European Union 1980-2000: Indicators and Analysis. *KS-AO-02-005-EN-N*. Luxembourg: Office for Official Publications of the European Communities.

FAO. 2000. Fertilizer requirements in 2015 and 2030. Rome: Food and Agriculture Organization of the United Nations.

FAO. 2003a. Food Balance Sheet for China in 2000. *FAOSTAT Online Database*. Download on 12 December 2003.

FAO. 2003b. *FAOSTAT Online Database*. Download on 12 December 2003.

Farmer, A. M. 1999. Implementation of the 1991 EU Urban Waste Water Treatment Directive and its role in reducing phosphate discharges. London: Institute for European Environmental Policy.

Farmer, A. M. 2001. Reducing phosphate discharges: the role of the 1991 EC urban wastewater treatment directive. *Water Science and Technology* 44 (1): 41-48.

Femia, A., F. Hinterberger, and F. Luks. 2001. Ecological economic policy for sustainable development: Potentials and domains of intervention for delinking approaches. *Population and Environment* 23 (2): 157-174.

Fenton, M. D. 2001. Iron and steel recycling in the United States in 1998. Denver: *Open file report 01-224*, U.S. Geological Survey, U.S. Department of the Interior.

Fischer-Kowalski, M. 1994. Society's metabolism: on the childhood and adolescence of a rising conceptual star. In *The International Handbook of Environmental Sociology*, edited by M. Redclift and G. Woodgate, p:119-137. Cheltenham, UK: Edward Elgar.

Fittschen, I., and J. Niemczynowicz. 1997. Experiences with dry sanitation and greywater treatment in the ecovillage Toarp, Sweden. *Water Science and Technology* 35 (9): 161-170.

Fleig, A.-K. 2000. Eco-industrial parks: A strategy towards industrial ecology in developing and newly industrialized countries. For pilot project of Strengthening Environmental Capability in Developing Countries (ETC). *Working Papers ETC-11*. German: GTZ.

Fleig, A.-K. 2000. Eco-industrial parks: A strategy towards industrial ecology in developing and newly industrialized countries. German: GTZ.

Foster, J. B. 1999. Marx's theory of metabolic rift: Classical foundations for environmental sociology. *American Journal of Sociology* 105 (2): 366-405.

Frosch, R. A. 1992. Industrial ecology: a philosophical introduction. *The National Academy of Sciences of the USA*, p:800-803, at Washing D.C.

Frosch, R. A., and N. E. Gallopoulos. 1989. Strategies for manufacturing. *Scientific American* 261 (3): 144-152.

Fujie, K., and N. Goto. 2002. Materials flow analysis and modeling to establish a material flow network in regional area. *Sino-Japan Workshop on Circular Economy and Society*, at Beijing, Oct. 10-11.

Gao, C., and T. L. Zhang. 1999. Contribution of agricultural phosphorus losses to eutrophication of waters and its controlling strategies. *Journal of Lake Sciences* 11(4): 369-375. (in Chinese).

Gao, X. Z., W. Q. Ma, C. B. Ma, *et al.* 2002. Analysis on the current status of utilization of crop straw in China. *Jounral of Huazhong Agricultural University* 21 (3): 242-247. (in Chinese).

GAO. 1995. Animal agriculture: information on waste management and water quality issues. *GAO/RCED-95-200BR*. Washington, DC: Resources, Community, and Economic Development Division, Government Accounting Office.

Gerber, P., P. Chilonda, G. Franceschini, *et al.* 2005. Geographical determinants and environmental implications of livestock production intensification in Asia. *Bioresource Technology* 96 (2): 263-276.

Gollehon, N., M. Caswell, M. Ribaudo, *et al.* 2001. Confined Animal Production and Manure Nutrients. *Agriculture Information Bulletin No. 771*. Washington, DC: Resource Economics Division, Economic Research Service, U.S. Department of Agriculture.

Graedel, T. E. 2000. A structured approach to LCA improvement analysis. *Journal of Industrial Ecology* 3 (2&3): 85-93.

Graedel, T. E., and B. R. Allenby. 2004. *Industrial Ecology*. *Second ed*. Beijing: Tsinghua University Press. *Reprint edition*.

Guinee, J. B., J. C. J. M. van den Bergh, J. Boelens, *et al.* 1999. Evaluation of risks of metal flows and accumulation in economy and environment. *Ecological Economics* 30 47-65.

Guo, H. G., and P. J. Ma. 2000. Treatment framework and estimated investment of Dianchi Lake. *Yunnan Environmental Science* 19 (Supplement): 109-112. (in Chinese).

Haberl, H. 2001. The energetic metabolism of societies, part I: Accounting concepts. *Journal of Industrial Ecology* 5 (2): 11-33.

Haberl, H. 2002. The energetic metabolism of societies, part II: Empirical examples. *Journal of Industrial Ecology* 5 (2): 71-88.

Hahn, H. H. 2001. Agricultural use of sewage sludge - A controversial issue. Sewage Sludge Disposal: Sustainable and/or Reliable Solutions, p. 11-20, *EWA/OWAV Workshop*, at Vienna. IWAG, TU-Wien.

Han X. G., L. H. Li, and J. H. Huang. 1999. *An Introduction to Biogeochemistry*. Beijing: Higher Education Press; German: Springer-Verlag. (in Chinese).

Han, L. J., Q. J. Yan, X. Y. Liu, *et al.* 2002. Straw resources and their utilization in China. *Transactions of the CSAE* 18 (3): 87-91. (in Chinese).

Han, X. G., L. H. Li, and J. H. Huang, eds. 1999. *An Introduction to Biogeochemistry*. Beijing: Higher Education Press. (in Chinese).

Hanaus, J., D. Hellstrom, and E. Johansson. 1997. A study of a urine separation system in an ecological village in northern Sweden. *Water Science and Technology* 35 (9): 153-160.

Hannah, L., D. Lohse, C. Hutchinson, *et al.* 1994. A preliminary inventory of human disturbance of world ecosystems. *AMBIO* 23 246-250.

Hansen, E., and C. Lassen. 2003. Experience with the use of substance flow analysis in Denmark. *Journal of Industrial Ecology* 6 (2-3): 201-219.

Haygarth, P. 1997. Agriculture as a source of phosphorus transfer to water: sources and pathways. Brussels, Belgium: SCOPE (Scientific Committee on Phosphates in Europe).

He, L. P., and X. H. Zhao. 2003. Comprehensive treatment engineering measures of water pollution and its benefit analysis in Dianchi Lake Basin during the period of "the Ninth Five Year Plan". *Yunnan Environmental Science* 22 (3): 40-43. (in Chinese).

He, S. Z. 2002. Monitoring data on effects of phosphorus precipitation in septic tanks of selected residential areas in Kunming City. Kunming: KIES. July, 2002. *Personal communication*.

He, S. Z., and G. Q. Zhi. 2004. Analysis on efficiency of engineering system of water pollution control in Dianchi Lake Basin. *Yunnan Environmental Science* 23 (1): 17-18. (in Chinese).

Heeres, R. R., W. J. V. Vermeulen, and F. B. de Walle. 2004. Eco-industrial park initiatives in the USA and the Netherlands: first lessons. *Journal of Cleaner Production* 12 (8-10): 985-995.

Heijungs, R., and R. Kleijn. 2000. Numerical approaches towards life cycle interpretation: five examples. The Netherlands: *CML-SSP Working Paper 2000.001*, Centre of Environmental Science, Leiden University.

Heilig, G. K. 1999. *Can China Feed Itself? A System for Evaluation of Policy Options*. Laxenburg, Austria: IIASA. CD-ROM, Version 1.1.

Hekkert, M. P., L. A. J. Joosten, and E. Worrell. 2000. Analysis of the paper and wood flow in The Netherlands. *Resources, Conservation and Recycling* 30 29-48.

Hinterberger, F., and F. Schmidt-Bleek. 1999. Dematerialization, MIPS and Factor 10: Physical sustainability indicators as a social device. *Ecological Economics* 29 53-56.

Hinterberger, F., F. Luks, and F. Schmidt-Bleek. 1997. Material flows vs. 'natural capital' - What makes an economy sustainable? *Ecological Economics* 23 1-14.

Hinterberger, F., S. Giljum, and M. Hammer. 2003. Input-output analysis of material flows. Vienna, Austria: Sustainable Europe Research Institute.

Hoffman, F. A., and J. W. Bishop. 1994. Impacts of a phosphate detergent ban on concentrations of phosphorus in the James River, Virginia. *Water Resources* 28 1239-1240.

Hofstetter, P. 1998. *Perspectives in Life Cycle Impact Assessment: A Structured Approach to Combine Models of the Technosphere, Ecosphere, and Valuesphere*. Boston: Kluwer Academic.

Hogenboom, J., A. P. J. Mol, and G. Spaargaren. 2000. Dealing with environmental risks in reflexive modernity. In *Risk in the Modern Age: Social Theory, Science and Environmental Decision-Making*, edited by M. J. Cohen. Hampshire: Macmillan Press Ltd.

Holz, C. A. 2003. "Fast, Clear and Accurate": How reliable are Chinese output and economic growth statistics? *The China Quarterly* (173): 122-163.

Holz, C. A. 2004. Deconstructing China's GDP statistics. *China Economic Review* 15 (2): 164-202.

Hotte, M. H. H., J. van der Vlies, and W. A. Hafkamp. 1995. Dutch policies aimed at diminishing mineral surpluses from manure and fertilizer. In *Green Budget Reform: An International Casebook of Leading Practices*, edited by R. Gale, S. Barg and A. Gillies, p:136-150. London: Earchscan.

Huang, B. F. 1997. Estimates on the supply and demand of fodder in China in 2000. *China Grains Economy* 4: 10-13. (in Chinese).

Huang, W. Y., G. Gao, J. H. Shu, *et al.* 2001. Effect of banning phosphate in detergents on the eutrophication of Taihu Lake. *Rural Eco-Environment* 17 (4): 26-29. (in Chinese).

Huang, W. Y., J. H. Shu, and P. Z. Xu. 2000. The contribution of phosphorus in detergents from the catchment of YiLi to Lake Tai. *Transactions of Oceanology and Limnology* (4): 20-25. (in Chinese).

Huang, Y. T. 1999. Pollution conditions of Lake Dianchi and its comprehensive control countermeasures. *Environmental Pollution and Control* 21 (4): 28-31. (in Chinese).

Huang, Z. Q. 2001. Phosphate resources and its industrial development in Yunnan province. *Conservation and Utilization of Mineral Resources* (2): 17-21. (in Chinese).

Hubacek, K., and S. Giljum. 2003. Applying physical input-output analysis to estimate land appropriation (ecological footprints) of international trade activities. *Ecological Economics* 44 137-151.

IFA. 1999. Fertilizers Indicators. Paris: International Fertilizer Industry Association.

ILEC, and UNEP. 2003. World Lake Vision: A Call to Action. Kusatsu, Japan: International Lake Environment Committee Fundation and United Nations Environment Programme.

Isacsson, A., K. Jonsson, I. Linder, *et al.* 2000. Material flow accounts: DMI and DMC for Sweden 1987-1997. *Eurostat Working Paper No. 2/2000/B/2*. European Commission.

ISF. 2003. *A Database on China's Cultivated Areas and Phosphorus Content in Soils in 2000*. Beijing: The Chinese Academy of Agricultural Sciences (CAAS).

Isherwood, K. F. 2000. Mineral fertilizer use and the environment. Paris: International Fertilizer Industry Association.

IWA, and GTZ. 2004. Ecosan - Closing the Loop. The *2nd international symposium on ecological sanitation*, April 2003, at Lubeck, Germany. Deutsche Gesellschaft fur Technische

Zusammenarbeit (GTZ) GmbH.

IWAG. 2001. Sewage Sludge Disposal: Sustainable and/or Reliable Solutions. ÖWAV / EWA Workshop, 10-11 Sep, TU-Wien, Vienna.

JAD. 2001. Implementation scheme of the Tenth Five-Year Plan for dairy cow breeding. Kunming: Jinning Agricultural Department. (in Chinese).

Jahiel, A. R. 1998. The organization of environmental protection in China. *The China Quarterly* 156 (December): 757-787.

Janicke, M. 1995. Green industrial policy and the future of "dirty industries". Free University Berlin: Unpublished paper.

Jänicke, M. 2000. Ecological modernization: innovation and diffusion of policy and technology. *FFU-report 00-08*. Berlin: Forschungsstelle für Umweltpolitik (Environmental Policy Research Unit), Freie Universitit Berlin (Free University of Berlin).

Jänicke, M., and H. Weidner, eds. 1995. *Successful Environmental Policy - A Critical Evaluation of 24 Cases*. Berlin: Sigma.

Jänicke, M., M. Binder, and H. Mönch. 1997. 'Dirty industry': Patterns of change in industrial countries. *Environmental and Resource Economics* 9 467-491.

Jia, X. H., Y. F. Huang, and J. T. Xu, eds. 2002. *Manure Process and Utilization*. Beijing: Chemical Industry Press. (in Chinese).

Jiang, L. F., and J. X. Gu. 2002. Practice and enlightenment on the comprehensive harness of excrement from animals and poultries in Shanghai. *Ecology of Domestic Animal* 23 (1): 1-4. (in Chinese).

Jiang, L. F., S. Q. Zhou, Z. L. Yang, *et al.* 1995. Review of legislation for pollution control of animal waste in developed countries and regions. *Ecology of Domestic Animal* 16 (2): 44-49. (in Chinese).

Jiang, S. H. 1999. The present and prospects of phosphorus-free detergents in China. *Environmental Protection* (6): 36-38. (in Chinese).

Jiang, S. X. 1999. *Phosphoric Acid, Phosphorous Fertilizer and Compound Fertilizer*. Beijing: Chemical Industry Press. (in Chinese).

Jiang, S. X., ed. 1999. *Phosphoric Acid, Phosphorous Fertilizer and Compound Fertilizer*. Beijing: Chemical Industry Press. (in Chinese).

Jin X. C. 1995. *Lakes in China – Research of Their Environment, Vols. I and II*. Beijing: China Ocean Press,. (in Chinese).

Jin, D. X., and K. J. Wang. 2002. Pollution prevent and control countermeasures for livestock and poultry farms in scale. *Environmental Protection* (12): 18-20. (in Chinese).

Jin, J. Y., and S. Portch. 2001. Recent agricultural and fertilizer developments in China. *IFA Regional Conference for Asia and the Pacific*, 10-13 December, at Ha Noi, Viet Nam.

Jin, X. C. 1995. *Lakes in China - Research of Their Environment*. Vol. I & II. Beijing: China Ocean Press. (in Chinese).

Jin, X. C., ed. 2001. *Techniques for Control and Management of Lake Eutrophication*. Beijing: Chemical Industry Press. (in Chinese).

Jolly, J. H. 1992. Material flow of zinc in the United States 1850-1990. Denver: *Open file report/1992*, Bureau of Mines, U.S. Department of the Interior.

Joosten, L. A. J., M. P. Hekkert, and E. Worrell. 2000. Assessment of the plastic flows in The Netherlands using STREAMS. *Resources, Conservation and Recycling* 30 135-161.

Joosten, L. A. J., M. P. Hekkert, E. Worrell, *et al.* 1999. STREAMS: a new method for analysing material flows through society. *Resources, Conservation and Recycling* 27 249-266.

Joshi, S. 2000. Product environmental life-cycle assessment using input-output techniques. *Journal of Industrial Ecology* 3 (2&3): 95-120.

JSDLR. 2003. *Extraction and Utilization of Mineral Resources in China*. Nanjing: Jiangsu Department of Land and Resources. (in Chinese).

KAB, and KAA. 1999. Survey report on agricultural non-point source pollution and control measures of Dianchi Watershed. Kunming: Kunming Agricultural Bureau and Kunming Agricultural Academy. (in Chinese).

KAB. 2001. General introduction on production and export of economical crops in Kunming Municipality. Kunming: Kunming Agricultural Bureau. (in Chinese).

Karl, D. M. 2000. Aquatic ecology: Phosphorus, the staff of life. *Nature* 406 (6791): 31-33.

KDAB. 2000. Working Report on the First Shock Check on Ban and Limitation of P-Containing Laundry Powders in Dianchi Basin in 2000. *Interior document*. Kunming: Kunming Dianchi Administration Bureau. (in Chinese).

KDAB. 2004a. Operation State of Kunming Wastewater Treatment Plants in 2003. Kunming: Kunming Institute of Environmental Science. http://www.dianchi.gov.cn/news/news_detail.asp?id=16.

KDAB. 2004b. Introduction on World Bank's Integrated Environmental Regulation Program in Dianchi Basin. Kunming: Kunming Institute of Environmental Science. http://www.dianchi.-gov.cn/news/news_detail.asp?id=2.

Kelly, T. 1998. Crushed cement concrete substitution for construction aggregates - A materials flow analysis. Denver: *U.S. Geological Survey Circular 1177*, U.S. Department of the Interior.

KIES, and DESE. 2000. General survey on non-point source pollution of demonstration plot in Dianchi Watershed: Final report. Kunming: Kunming Institute of Environmental Science. (in Chinese).

KIES. 2002. Investigation Report on Livestock and Poultry Husbandry in Dianchi Watershed. Kunming: Kunming Institute of Environmental Science. (in Chinese).

Kincaid, J., and M. Overcash. 2001. Industrial ecosystem development at the metropolitan level. *Journal of Industrial Ecology* 5 (1): 117-126.

Kleijn, R. 2000. In=Out: the trivial central paradigm of MFA? *Journal of Industrial Ecology* 3 (2&3): 8-10.

Kleijn, R. 2001. Adding it all up: The sense and non-sense of bulk-MFA. *Journal of Industrial Ecology* 4 (2): 7-8.

Kleijn, R., A. Tukker, and E. van der Voet. 1997. Chlorine in the Netherlands - part I: An overview. *Journal of Industrial Ecology* 1 (1): 91-112.

Kleijn, R., R. Huele, and E. van der Voet. 2000. Dynamic substance flow analysis: the delaying mechanism of stocks, with the case of PVC in Sweden. *Ecological Economics* 32 241-254.

Konijn, P., S. de Boer, and J. van Dalen. 1997. Input-output analysis of material flows with application to iron, steel and zinc. *Structural Change and Economic Dynamics* 8 129-153.

Konikow, L. F., and J. D. Bredehoeft. 1992. Ground water models cannot be validated. *Advances in Water Resources* 15 (1): 75-83.

Korhonen, J., M. Wihersaari, and I. Savolainen. 2001. Industrial ecosystem in the Finnish forest industry: Using the material and energy flow model of a forest ecosystem in a forest industry system. *Ecological Economics* 39 145-161.

Kostick, D. S. 1996. The material flow concept of materials. *Nonrenewable Resources* 5 (4): 211-233.

Kramer, D. A. 2002. Magnesium recycling in the United States in 1998. Denver: *U.S. Geological Survey Circular 1196-E*, U.S. Department of the Interior.

Krotscheck, C., and M. Narodoslawsky. 1996. The sustainable process index: A new dimension in ecological evaluation. *Ecological Engineering* 6 241-258.

Krozer, J., and J. C. Vis. 1998. How to get LCA in the right direction? *Journal of Cleaner Production* 6 53-61.

KSB. 2001. *Kunming Statistical Yearbook 2000*. Beijing: China Statistics Press.

Kvarnström, E., C. Schönning, M. Carlsson-Reich, *et al.* 2003. Recycling of wastewater-derived phosphorus in Swedish agriculture - a proposal. *Water Science and Technology* 48 (1): 19-25.

Lai, S. Y. 2004. Study on Non-point Sources Pollution Investigation and Evaluation Method and Its Application. *MSc Thesis*. Department of Environmental Science and Engineering, Tsinghua University, Beijing. (in Chinese).

Larsen, T. A., and W. Gujer. 1996. Separate management of anthropogenic nutrient solutions (human urine). *Water Science and Technology* 34 (3-4): 87-94.

Larsen, T. A., I. Peters, A. Alder, R. Eggen, M. Maurer, and J. Muncke. 2001. Re-engineering the toilet for sustainable wastewater management. *Environmental Science & Technology* 35(9): 192A–197A.

Lee, G. F., and A. Jones-Lee. 1995. Comment on 'Impact of a phosphate detergent ban on concentrations of phosphorus in the James River, Virginia'. *Water Resources* 29 (5): 1425-1426.

Lee, G. F., and R. A. Jones. 1986. Detergent phosphate bans and eutrophication. *Environmental Science and Technology* 20 (4): 330-331.

Lenzen, M. 2001. Errors in conventional and input-output-based life-cycle inventories. *Journal of Industrial Ecology* 4 (4): 127-148.

Leontief, W. 1970. Environmental repercussions and the economic structure: An input-output approach. *Review of Economics and Statistics* 52 (1): 262-271.

Li, X. 2000. Phosphate-free doesn't mean environmental protection. *Detergent and Cosmetics* 23 (1): 228-229. (in Chinese).

Li, X. Y., C. S. Zuo, J. B. Tschirley, *et al.* 1997. Sustainable agriculture and rural development in China. Rome: Sustainable Development Department (SD), Food and Agriculture Organization of the United Nations.

Li, Y. 2002. Existing problems and prevention and control countermeasures of intensive livestock cultivation in China. *Shanghai Environmental Sciences* 21 (10): 597-599. (in Chinese).

Li, Y., Z. J. Shan, and D. H. Xu. 2002. Preliminary studies of management of livestock pollution in China. *Chinese Journal of Eco-Agriculture* 10 (2): 136-138. (in Chinese).

Li, Z. Q., and J. Xie. 1999. Environmental impact of phosphorus in detergent and approaches to solve it. *Sichuan Environment* 18 (3): 32-36. (in Chinese).

Li, Z. X., and Y. F. Lu. 2004. The investigations and assessment of dejection contamination of beasts and birds in scale cultivation in Dianchi drainage area. *Southwest China Journal of Agricultural Sciences* 17 (Supplement): 285-289. (in Chinese).

Lindert, P. H. 1999. The bad earth? China's soils and agricultural development since the 1930s. *Economic Development and Cultural Change* 47 (4): 701-736.

Litke, D. W. 1999. Review of phosphorus control measures in the United States and their effects on water quality. *Water-Resources Investigations Report 99-4007*. Denver: U.S. Geological Survey.

Liu, B. Z., G. L. Li, F. Q. Wu, and X. G. Zhao. 1995. The regular patterns of the loss of soil nutrients on Southern Loess Plateau. *Journal of Soil and Water Conservation* 9(2): 77-86. (in Chinese).

Liu, H. Z., L. J. Ren, and M. Hu. 1998. Situation and countermeasures of limiting phosphorus in China's lakes. *Environmental Protection* (8): 27-28. (in Chinese).

Liu, L. P. 2001. Analysis on eutrophication of Dianchi Lake and its control measures. *Yunnan Environmental Science* 20 (supplement): 25-27. (in Chinese).

Liu, P. F., Z. L. Chen, S. Y. Xu, *et al.* 2002. Waste loading and treatment strategies on the excreta of domestic animal in the Yangtze Delta. *Resources and Environment in the Yangtze Basin* 11 (5): 456-460. (in Chinese).

Liu, W. C. 1998. Economic policies for municipal sewage treatment technologies in China. *China Environmental Protection Industry* (6): 24-25. (in Chinese).

Liu, Y. 2004. Study on phosphorus societal metabolism and eutrophication control policy in China. *Ph.D. thesis*, Tsinghua University, Beijing, People's Republic of China.

Liu, Y., A. P. J. Mol, and J. Chen. 2005. The environmental industry in transitional China: Barriers and opportunities between state and market. *International Journal of Environment and Sustainable Development* 4 (5): *forthcoming*.

Liu, Y., A. P. J. Mol, and J. N. Chen. 2004. Material Flow and Ecological Restructuring in China: the Case of Phosphorus. *Journal of Industrial Ecology* 8 (3).

Liu, Y., and J. N. Chen. 2000. Estimate on the costs of urban sewage system and uncertainty analysis: a numerical approach. Beijing: Department of Environmental Science and Engineering, Tsinghua University. (in Chinese).

Liu, Y., J. Chen, and A. P. J. Mol. 2004. Evaluation of phosphorus flows in the Dianchi Watershed, southwest of China. *Population and Environment* 25(6): 637-656.

Llewellyn, T. O. 1994. Cadmium (material flow). Denver: *Information Circular 9380*, Bureau of Mines, U.S. Department of the Interior.

Loebenstein, J. R. 1994. The material flow of arsenic in the United States. Denver: *Information Circular 9382*, Bureau of Mines, U.S. Department of the Interior.

Lowe, E., and Y. Geng, eds. 2003. *Industrial Ecology and Ecological Industrial Parks*. Beijing: Chemical Industry Press. (in Chinese).

Lu, S. L. 1997. Pollution and control countermeasure in Dianchi Lake. *Chongqing Environmental Science* 19 (3): 1-4. (in Chinese).

Lu, S. L., G. X. Shen, and Y. G. Wang. 1994. Study on suitable application amount of livestock and poultry droppings in Shanghai area. *Acta Agricultural Shanghai* 10 (Supplement): 51-56. (in Chinese).

Lu, W. G. 1997. Estimates on grain consumption for feed in China. *China Grain Economics* (3): 38-40. (in Chinese).

Lu, Y. F., Z. X. Li, and B. K. Lei. 2003. Status and assessment of fertilizing the nitrogen fertilizer and phosphatic fertilizer to the field in Dianchi Watershed. *Yunnan Environmental Science* 22 (1): 34-37. (in Chinese).

Lundin, M., M. Olofsson, G. J. Pettersson, *et al.* 2004. Environmental and economic assessment of sewage sludge handling options. *Resources, Conservation and Recycling* 41 (4): 255-278.

Lundqvist, L. J. 1996. Environmental politics in the Nordic countries: policy, organization, and capacity. In *Governing the Environment: Politics, Policy and Organization in the Nordic Countries*, edited by P. M. Christiansen, p:13-27. Copenhagen: Nordic Council of Ministers.

Luo, F. Y. 1997. A discussion on the position and the development of the phosphate rock in Yunnan. *Yunnan Chemical Industry* (4): 8-12. (in Chinese).

Luo, L. 2000. Feed industry in China: A challenge to the next millennium. *Feed Industry* 21 (5): 5-6. (in Chinese).

Ma, P. S., H. L. Wang, L. H. Cheng, *et al.* 2003. Development of urban wastewater treatment in Kunming City. *China Water and Wastewater* 19 (4): 19-22. (in Chinese).

Ma, X., and L. Ortolano. 2000. *Environmental Regulation in China: Institutions, Enforcement, and Compliance*. Lanham: Rowman & Littlefield Publishers, Inc.

Mainstone, C. P., and W. Parr. 2002. Phosphorus in rivers -- ecology and management. *The Science of The Total Environment* 282-283 25-47.

Matos, G., and L. Wagner. 1998. Consumption of materials in the United States, 1900-1995. *Annual Review of Energy and Environment* 23 107-122.

Matson, P. A., W. J. Parton, A. G. Power, *et al.* 1997. Agricultural intensification and ecosystem properties. *Science* 277 504-509.

Matthews, E., C. Amann, S. Bringezu, *et al.* 2000. *The Weight of Nations: Material Outflows from Industrial Economies*. Washington, D.C.: World Resources Institute.

Matthews, H. S., and M. J. Small. 2001. Extending the boundaries of life-cycle assessment through environmental economic input-output models. *Journal of Industrial Ecology* 4 (3): 7-10.

Meng, T. Y. 2002. Eutrophication of Dianchi and its trend and treatment. *Yunnan Environmental Science* 21 (1): 35-38. (in Chinese).

Meng, Y. F. 1999. The development tendency analysis of nitrogen and phosphorus content in Waihai of Dianchi Lake. *Yunnan Environmental Science* 18 (4): 32-33. (in Chinese).

Mickwitz, P. 2000. Evaluating environmental policy instruments. *American Evaluation Society Conference*, at Waikiki, November 1st - 4th 2000.

Mickwitz, P. 2003. A framework for evaluating environmental policy instruments: context and key concepts. *Evaluation* 9 (4): 415-436.

MOC, ed. 2001. *Urban Construction Statistics Yearbook of China Cities*. Beijing: China Building Materials Industry Press.

MOH, ed. 2004. Health Yearbook of China 2003. Beijing: People's Medical Publishing House.

Mol, A. P. J. 1995. *The refinement of production. Ecological modernization theory and the chemical industry*. Utrecht, The Netherlands: International Books.

Mol, A.P.J. 2003. Political modernisation and environmental governance: between delinking and linking, *EUROPAEA, Journal of Europeanists* 8 (1-2):.169-185.

Moolenaar, S. W., E. J. M. Temminghoff, and F. A. M. D. Haan. 1998. Modeling dynamic copper balances for a contaminated sandy soil following land use change from agriculture to forestry.

Environmental Pollution 103 117-125.

Morse, G. K., J. N. Lester, and R. Perry. 1993. *The Economic and Environment Impact of Phosphorus Removal from Wasetwater in the European Community*. London: Centre Europeen D'Etudes des Polyphosphates E.V., Selper Publications.

Moss, B. 2000. Phosphorus, detergents and eutrophication control - A summary of experience in Europe and North America. *Detergent and Cosmetics* 23 (2, Supplement, July): 117-129.

NBS, ed. 2001. *China Statistical Yearbook 2001*. Beijing: China Statistics Press.

NBS, ed. 2003. *China Industry Economy Statistical Yearbook 2003*. Beijing: China Statistics Press.

NBS, ed. 2003. *China Statistical Yearbook 2003*. Beijing: China Statistics Press.

NECW. 2003. The state and planning of municipal sewage treatment in China. *China Environmental Protection Industry* (1): 32-35. National Engineering and Technology Research Center for Urban Water. (in Chinese).

Neeteson, J. J. 2000. Nitrogen and phosphorus management on Dutch dairy farms: legislation and strategies employed to meet the regulations. *Biology and Fertility of Soils* 30 (5/6): 566-572.

Nijkamp, P., and C. J. M. J. van den Bergh. 1997. New advances in economic modeling and evaluation of environmental issues. *European Journal of Operational Research* 99 180-196.

Niu, B. J. 2000. The impact on and strategy of China's fodder industry after entering WTO. *Journal of Huazhong Agricultural University* (*Social Sciences Edition*) 3: 15-17. (in Chinese).

Ober, J. A. 2002. Material flow of sulfur. Denver: *Open file report 02-298*, U.S. Geological Survey, U.S. Department of the Interior.

OECD. 1989. *Agricultural and Environmental Policies: Opportunities for Integration*. Paris: Organization for Economic Co-operation and Development.

OECD. 1994. Managing the Environment: The Role of Economic Instruments. Paris: Organization for Economic Cooperation and Development.

OECD. 2000. Special session on material flow accounting. Working Group on Environmental Information and Outlooks (WGEIO), at Paris.

OECD. 2002. Agricultural Policies in China after WTO accession. *Agricultural Policy Adjustments in China after WTO accession*, edited by W. Legg, X. D. Li, M. Patterson, S. Taylor and A. Trzeciak-Duval, at Beijing, 30-31 May, 2002.

OECD. 2003. *Environmental Performance Reviews: Water*. Paris: Organization for Economic Cooperation and Development.

Oreskes, N., K. Shrader-Frechette, and K. Belitz. 1994. Verification, Validation, and Confirmation of Numerical Models in the Earth Sciences. *Science* 263 (4, February): 641-646.

Otterpohl, R., A. Albold, and M. Oldenburg. 1999. Source control in urban sanitation and waste management: Ten systems with reuse of resources. *Water Science and Technology* 39(5): 153–160.

Pahl-Wostl, C., A. Schoborn, N. Willi, *et al.* 2003. Investigating consumer attitudes towards the new technology of urine separation. *Water Science and Technology* 48 (1): 57-65.

Palmer, M. 1998. Environmental regulation in the People's Republic of China: the face of domestic law. *The China Quarterly* 156 (December): 788-808.

Palumbi, S. R. 2001. Human as the world's greatest evolutionary force. *Science* 293 1786-1790.

Pastowski, A. 1997. Decoupling economic development and freight for reducing its negative impacts. *Wuppertal Papers No.79 (ISSN 0949-5266)*. Wuppertal, Germany: Wuppertal Institute.

Patton, C. V., and D. S. Sawicki. 2002. *Basic Methods of Policy Analysis and Planning*. Second ed. (Chinese version). Beijing: Huaxia Publishing House.

Patton, M. Q. 1980. *Qualitative Evaluation Methods*. London: Sage.

Peng, X. Y., and X. R. Hou. 2000. The theoretic and practical analysis of relationship between material input and income comparison between China and other Countries. *Chinese Soft Science (Zhongguo Ruankexue)* (9): 33-38. (in Chinese).

Picton, T., and P. L. Daniels. 1999. Ecological restructuring for sustainable development: evidence from the Australian economy. *Ecological Economics* 29 (3): 405-425.

Piekema, P., and P. Roeleveld. 2001. Recovery of phosphate. Sewage Sludge Disposal: Sustainable and/or

Reliable Solutions, p.89-102, *EWA/OWAV Workshop*, at Vienna. IWAG, TU-Wien.

Pope, L. M., C. R. Milligan, and D. P. Mau. 2002. Historical contributions of phosphorus from natural and agricultural sources and implications for stream water quality, Cheney Reservoir Watershed, South-Central Kansas. Denver: *Water-Resources Investigations Report 02–4021*, U.S. Department of the Interior and U.S. Geological Survey.

Posch, A. 2002. From 'industrial symbiosis' to 'sustainability networks'. The Environment and Sustainable Development in New Central Europe: Austria and Its Neighbors, at Minneapolis, USA.

Power, J. F. and R. I. Papendick. 1992. Organic sources of nutrients, in *Fertilizer Technology and Use* (3rd edition), Engelstad O. P. edited. Translated by Zhang Guo-Xin *et al.* Beijing: Chemical Industry Press.

Powers, C. W., and M. R. Chertow. 1997. Industrial ecology: overcoming policy fragmentation. In *Thinking Ecologically: The Next Generation of Environmental Policy*, edited by M. R. Chertow, and D. C. Esty, p:19-36. New Haven: Yale University Press.

Prosterman, R., T. Hanstad, and L. Ping. 1996. Can China Feed Itself? *Scientific American* November: 70-76.

Qian, T. M. 2000. Consideration of the changes of phosphorus loading in water before and after prohibiting (limiting) phosphorus in detergent. *Chongqing Environmental Science* 22 (5): 38-40. (in Chinese).

Qiang, J. H. 2002. Summarizing fro managements of pollution in water environment in Dianchi. *Yunnan Geographic Environment Research* 14 (1): 61-66. (in Chinese).

Qin, F. 1997. Discussion on market demand of feed in China. *Journal of Agrotechnical Economics* (4): 41-45. (in Chinese).

Quade, E. S. 1997. Predicting the consequences: Models and modeling. In *Handbook of Systems Analysis: Overview of Uses, Procedures, Applications, and Practice*, edited by H. J. Miser and E. S. Quade. Chichester: John Wiley&Sons.

Quan, W. M., and L. J. Yan. 2002. Effects of agricultural non-point source pollution on eutrophication of waterbody and its control measure. *Acta Ecological Sinica* 22 (3): 291-299. (in Chinese).

Rappaport, R. A. 1971. The flow of energy in an agricultural society. *Scientific American* 224 (3): 117-133.

Rawski, T. G., and R. W. Mead. 1998. On the trail of China's phantom farmers. *World Development* 26 (5): 767-781.

Ribaudo, M., N. Gollehon, M. Aillery, *et al.* 2003. Manure Management for Water Quality: Costs to Animal Feeding Operations of Applying Manure Nutrients to Land. *Agricultural Economic Report 824*. Washington, DC: Economic Research Service, U.S. Department of Agriculture.

Roberts, B. H. 2004. The application of industrial ecology principles and planning guidelines for the development of eco-industrial parks: an Australian case study. *Journal of Cleaner Production* 12 (8-10): 997-1010.

Rosegrant, M. W., M. S. Paisner, S. Meijer, *et al.* 2001. *2020 Global Food Outlook: Trends, Alternatives, and Choices*. Washington, D.C.: International Food Policy Research Institute.

Rossi, P. H., H. E. Freeman, and M. W. Lipsey. 2002. *Evaluation: A Systematic Approach*. Six ed. (Chinese version). Beijing: Huaxia Publishing House.

Rousseau, S., and S. Proost. 2002. The cost effectiveness of environmental policy instruments in the presence of imperfect compliance. *Working Paper Series No. 2002-04*. Belgium: Faculty of Economics and Applied Economic Sciences, Center for Economic Studies, Energy, Transport and Environment, Katholieke Universiteit Leuven.

Rozelle, S. D., and D. A. Sumner, eds. 2003. *Agricultural Trade and Policy in China: Issues, Analysis and Implications*. Edited by A. M. Chen and S. F. Song, The Chinese Economy Series. Aldershot, England: Ashgate.

Ruth, M. 2001. Dematerialization in five US metals sectors: implications for energy use and CO2 emissions. *Resources Policy* 24 (1): 1-18.

Rutherford, P. M., M. J. Dudas, and R. A. Samek. 1994. Environmental impacts of phosphogypsum. *The Science of The Total Environment* 149 (1-2): 1-38.

Schandi, H., W. Huttler, and H. Payer. 1999. Delinking of economic growth and material turnover. *Innovation* 12 (1): 31-45.

Schoer, K., H. Hoh, A. Heinze, *et al.* 2000. Material flow analysis in the framework of environmental economic accounting in Germany. *Eurostat Working Paper No. 2/2000/B/9*, European Commission.

Schütz, H., and M. J. Welfens. 2000. Sustainable development by dematerialization in production and consumption: strategy for the new environmental policy in Poland. *Wuppertal Report No.103 (ISSN 0949-5266)*. Wuppertal, Germany: Wuppertal Institute.

SCOPE. 2001. Phosphate recovery: where do we stand today? Brussels, Belgium: SCOPE (Scientific Committee on Phosphates in Europe).

Senbel, M., T. McDaniels, and H. Dowlatabadi. 2003. The ecological footprint: a non-monetary metric of human consumption applied to North America. *Global Environmental Change* 13 83-100.

SEPA. 1996. *Handbook for Industrial Pollutants Production and Discharge and Emission Coefficients*. Beijing: China Environmental Science Press.

SEPA. 2001. *Report on the State of the Environment in China (2000)*. Beijing: State Environmental Protection Administration.

SEPA. 2002. *Investigation on the Pollution of National Livestock and Poultry Breeding in Scale and Its Control Strategy*. Beijing: China Environmental Science Press. (in Chinese).

SEPA. 2003. Circular on the progress on water pollution prevention and control in key watersheds and marine areas. *huanban[2003]104*. Beijing: State Environmental Protection Administration.

SEPA. 2003. Communiqué on ecological and environmental monitoring of the Three Gorges. Beijing: State Environmental Protection Administration.

SEPA. 2004. News Press on Action of Rectification of Illegal Polluters and Safeguard of Public Health. July 13, Beijing. http://www.chinagateway.com.cn/chinese/huanjing/30081.htm.

Shen, G. X., Y. G. Wang, and D. W. Yuan. 1994. Loading amounts of animal feces and their alarming values and classification grades in Shanghai suburbs. *Acta Agricultural Shanghai* 10 (Supplement): 6-11. (in Chinese).

Shen, Q. R., ed. 2001. *General Introduction on Soil and Fertilizer*. Beijing: High Education Press. (in Chinese).

Shu, J. H., W. Y. Huang, X. Y. Gao, *et al.* 1999. The evaluation of the effect of powdered detergent containing phosphate on Tai Hu Water Area Eutrophication. *Detergent and Cosmetics* 22 (Supplement, August): 129-145. (in Chinese).

Si, Y. B., S. Q. Wang, and H. M. Chen. 2000. Nitrogen and phosphorus losses from agricultural soil and surface water eutrophication. *Journal of Soil* (4): 188-193. (in Chinese).

Silva, M. A. S. e. 2001. Environmental input-output analysis: Application to Portugal. Universidade Técnica de Lisboa, Instituto Superior Técnico.

Simonis, U. E. 1994. Industrial restructuring in industrial countries. In *Industrial Metabolism: Restructuring for Sustainable Development*, edited by R. U. Ayres and U. E. Simonis, p:31-54. Tokyo/New York/Paris: United Nations University Press.

Skjelhaugen, O. J. 1999. Closed system for local reuse of blackwater and food waste, integrated with agriculture. *Water Science and Technology* 39 (5): 161-168.

Smil, V. 1995. Who will feed China? *The China Quarterly* (143): 801-813.

Smil, V. 2000. Phosphorus in the environment: Natural flows and human interferences. *Annual Review of Energy and Environment* 25 53-88.

Smith, G. R. 2002. Lead recycling in the United States in 1998. Denver: *U.S. Geological Survey Circular 1196-F*, U.S. Department of the Interior.

Song, Z. W., X. W. He, F. S. Li, *et al.* 2003. Study on the eutrophication of water body and the effect of detergent phosphate on eutrophication in Shanghai. *Environmental Monitoring in China* 19 (5): 12-15. (in Chinese).

Spangenberg, J. H., A. Femia, F. Hinterberger, *et al.* 1998. Material flow-based indicators in environmental reporting. *Environmental Issue Report No.14*. Copenhagen: European Environmental Agency.

Stauffer, J. 1998. *The Water Crisis: Constructing Solutions to Freshwater Pollution*. London: Earthscan.

Steen, I. 1998. Phosphorus availability in the 21st century: Management of a non-renewable resource. *Phosphorus and Potassium* 217 (Sep/Oct): 25-31.

Steinfeld, H., C. de Hann, and H. Blackburn. 1998. Livestock and the environment: issues and options. In *Agriculture and the Environment: Perspectives on Sustainable Rural Development.* 1998. Washington, D.C.: The World Bank.

Stenger, A. 2000. Experimental valuation of food safety: Application to sewage sludge. *Food Policy* 25 (2): 211-218.

Suh, Y.-J., and P. Rousseaux. 2002. An LCA of alternative wastewater sludge treatment scenarios. *Resources, Conservation and Recycling* 35 (3): 191-200.

Sun, F. G., and Z. J. Li. 1999. Recent developments and prospects for fertilizer in China. In proceedings of 67th IFA Annual Conference, Manila, the Philippines, 17-20 May.

Sznopek, J. L., and T. G. Goonan. 2000. The materials flow of mercury in the economies of the United States and the world. Denver: *U.S. Geological Survey Circular 1197*, U.S. Department of the Interior.

Tao, J. F. 2002. The present status and progress of phosphorus chemical Industry in Yunnan province. *Inorganic Chemicals Industry* 34 (6): 13-15. (in Chinese).

Tiwari, R. K., and K. N. N. Rao. 1999. Periodicity in marine phosphorus burial rate. *Nature* 400 (6739): 31-32.

Udo de Haes, H. A. 2000. Weighting in life-cycle assessment: Is there a coherent perspective? *Journal of Industrial Ecology* 3 (4): 3-7.

Udo de Haes, H., E. van der Voet, and R. Kleijn. 1997. From quality to quantity: Substance flow analysis (SFA), an analytical tool for integrated chain management. *Regional and National Material Flow Accounting: From Paradigm to Practice of Sustainability, Wuppertal special 4,* edited by S. Bringezu *et al.*, p:32-42, at Leiden, The Netherlands.

UNEP. 1998. Mineral Fertilizer Production and The Environment: Part 1. The Fertilizer Industry's Manufacturing Processes and Environmental Issues. Washington, D.C.: *Technical Report No. 26-Part 1*, United Nations Publication.

UNEP. 2002. *China Human Development Report 2002: Making Green Development a Choice*. Oxford UK: Oxford University Press.

USDA, and USEPA. 1999. *Unified National Strategy for Animal Feeding Operations*. Washington, DC: U.S. Department of Agriculture, U.S. Environmental Protection Agency.

USDA. 2002. China's Food and Agriculture: Issues for the 21st Century. *Agriculture Information Bulletin No. 775*. Washington, DC: Market and Trade Economics Division, Economic Research Service, U.S. Department of Agriculture.

USEPA. 1999. Background Report on Fertilizer Use, Contaminants and Regulations. *EPA 747-R-98-003*. Washington, DC: U.S Environmental Protection Agency.

USEPA. 2002. *State Compendium: Programs and Regulatory Activities Related to Animal Feeding Operations*. Washington, DC: Office of Wastewater Management, U.S. Environmental Protection Agency.

USGS. 2004. *Mineral Commodity Summaries 2004*. Washington, DC: U.S Department of the Interior.

van der Veeren, R. J. H. M., and C. M. Lorenz. 2002. Integrated economic-ecological analysis and evaluation of management strategies on nutrient abatement in the Rhine basin. *Journal of Environmental Management* 66 361-376.

van der Voet, E. 1996. Substances from Cradle to Grave: Development of a methodology for the analysis of substance flows through the economy and the environment of a region. Ph.D. Thesis, Centre of Environmental Science, Leiden University, Leiden, the Netherlands.

van der Voet, E., J. B. Guinee, and H. A. Udo de Haes. 1999. Metals in the Netherlands: a problem solved? *CML-SSP Working Paper* 99.003, the Centre of Environmental Science, Leiden University, the Netherlands.

van Koppen, C. S. A. 2004. Eco-industrial design of biomass cycles: flows, technologies and actors. In *Resource Recovery and Reuse in Organic Solid Waste Management*, edited by P. Lens, *et al.*, p:26-45. London: IWA Publishing.

van Koppen, C. S. A., and A. P. J. Mol. 2002. Ecological modernization of industrial ecosystems. In *Water Recycling and Resource Recovery in Industry*, edited by P. Lens, *et al.*, p:132-158. London: IWA

Publishing.

Vedung, E. 1998. Policy Instruments: typologies and theories. In *Carrots, Sticks, and Sermons: Policy Instruments and Their Evaluation*, edited by M.-L. Bemelmans-Videc, R. C. Rist and E. Vedung, p:21-58. New Brunswick, NJ: Transaction Publishers.

Verfaillie, H. A., and R. Bidwell. 2000. Measuring eco-efficiency: A guide to reporting company performance. Geneva: World Business Council for Sustainable Development.

Vermeer, E. B. 1998. Industrial pollution in China and remedial policies. *The China Quarterly* 156 (December): 952-985.

VERNA, and HSB. 2000. Urine Seperation - Closing the nutrient cycle. Stockholmshem: VERNA Ecology; HSB National Federation.

Wackernagel, M., and W. E. Rees. 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. *Ecological Economics* 20 3-24.

Wackernagel, M., L. Onisto, P. Bello, *et al.* 1999. National natural capital accounting with the ecological footprint concept. *Ecological Economics* 29 375-390.

Waddell, T. E., and B. T. Bower. 1988. *Managing Agricultural Chemicals in the Environment: The Case for a Multimedia Approach*. The Conservation Foundation, Washington, D.C.

Wagner, L. A. 2002. Materials in the economy: Material flows, scarcity, and the environment. Denver: *U.S. Geological Survey Circular 1221*, U.S. Department of the Interior.

Wang, B. 2003. Study on Pollution Control Policies of Agricultural Non-Point Sources in Dianchi Lake Basin. *MSc Thesis*, Department of Environmental Science and Engineering, Tsinghua University, Beijing. (in Chinese).

Wang, F. K., S. G. Wang, and X. P. Li, eds. 1992. *Technical and Economic Handbook for Treatments of Industrial Wastewater and Urban Domestic Wastewater*. Beijing: Tsinghua University Press. (in Chinese).

Wang, H. M. 2004. Current situation and control countermeasures for pollution in scale poultry and livestock farm. *Yunnan Environmental Science* 23 (Supplement): 79-81. (in Chinese).

Wang, L. 1998. Benefit analysis of the World Bank load to Dianchi project. *Yunnan Environmental Science* 17 (1): 32-36. (in Chinese).

Wang, L. M., and J. Davis. 2000. *China's Grain Economy: The challenge of feeding more than a billion*. Aldershot, England: Ashgate.

Wang, P. W., Z. B. Wei, and Y. Wang. 2003. Suggestion and research on phosphate in detergent and water protection. *Detergent and Cosmetics* 26 (6): 10-13. (in Chinese).

Wang, Q. B., C. Halbrendt, and S. R. Johnson. 1996. Grain production and environmental management in China's fertilizer economy. *Journal of Environmental Management* 47 283-296.

Wang, T., and S. Y. Lou. 2004. Investigation of urban wastewater treatment techniques and assessment on technological and economic indicators. *Water and Wastewater Engineering* 30 (5): 1-4. (in Chinese).

Wang, X. C., and D. S. Mo. 2001. Study on the development of municipal wastewater treatment plants. *Shandong Environment* (5): 25-26. (in Chinese).

Wang, X. X., T. L. Zhang, and B. Zhang. 1999. Nutrient cycling and balance of sloping upland ecosystems on red soil. *Acta Ecological Sinica* 19(3): 335-341. (in Chinese).

Wang, Y. B. 2001. Study on Environmental Consciousness of Individuals in Dianchi Basin. Kunming: Department of Social Sciences, Yunnan University. (in Chinese).

Wang, Y. F. 1998. Apprehension and foresight of phosphorous resource in Dianchi area, Yunnan province. *Geology of Chemical Minerals* 20 (2): 195-198. (in Chinese).

Wang, Y. M. 2002. Assessment on soil feasibility of Kunming. In *Studies on Soils, Fertilizers and Agricultural Non-Point Pollution Sources in Kunming*, edited by Y. M. Wang, p:117-129. Kunming: Yunnan Science and Technology Press. (in Chinese).

Watanabe, N., and H. Ito. 1997. Analysis of waste for combustion: The case of Osaka City, Japan. *Resources, Conservation and Recycling* 20 57-69.

WB. 1999. *Accelerating China's Rural Transformation*. Washington, D.C.: The World Bank.

WB. 2001. *China: Air, Land, and Water*. Washington, D.C.: The World Bank.

WBCSD. 1998. Cleaner production and eco-efficiency: Complementary approaches to sustainable development. Geneva, Switzerland: World Business Council for Sustainable Development (WBCSD).

WBCSD. 2000. Eco-efficiency: Creating more value with less impact. Geneva, Switzerland: World Business Council for Sustainable Development (WBCSD).

Weale, A. 1992. *The New Politics of Pollution: Issues in Environmental Politics*. Manchester: Manchester University Press.

Wen, Y. 1998. A preliminary study on quantitative assessment approach to intensity of human activity. *Impact of Science on Society* (4): 55-60. (in Chinese).

Wernick, I. K., and J. H. Ausubel. 1995. National material metrics for industrial ecology. *Resources Policy* 21 (3): 189-198.

Westerman, P. W., and J. R. Bicudo. 2005. Management considerations for organic waste use in agriculture. *Bioresource Technology* 96 (2): 215-221.

Wilsenach, J., and M. van Loosdrecht. 2003. Impact of separate urine collection on wastewater treatment systems. *Water Science and Technology* 48 (1): 103-110.

Wilson, B., and B. Jones. 1994. Life Cycle Assessment of Domestic Laundry Detergent Builders. Landbank Environmental Research and Consulting.

Wolman, A. 1965. The metabolism of cities. *Scientific American* 213 179-190.

World Bank. 2001. *China: Air, Land, and Water*. Washington, DC: World Bank.

WRI. 1994. World Resources 1994-1995. Washington, D.C.: World Resources Institute.

Wu, T. M. 2001. Report on the investigation of animal husbandry in scale in Jiangsu Province. *Environmental Herald* (4): 20-22. (in Chinese).

Wu, Y. S., ed. 1992. *Urban Wastewater Treatment: the Costs and Decision Making*. Beijing: China Environmental Science Press. (in Chinese).

Xu, G. T. 2001. Comprehensive treatment of livestock wastewater in Shanghai. *China Population, Resources and Environment* 11 67-68. (in Chinese).

Xu, H. B. 1996. World Bank funded project - industrial pollution control in Dianchi Basin. *Yunnan Environmental Science* 15 (3): 24-26. (in Chinese).

Yan, J. W., J. J. Ma, and Y. M. Wang. 2002. Report on fertilizers application in Kunming. In *Studies on Soils, Fertilizers and Agricultural Non-Point Pollution Sources in Kunming*, edited by Y. M. Wang, p:61-68. Kunming: Yunnan Science and Technology Press. (in Chinese).

Yan, W. J., C. Q. Yin, P. Sun, X. Y. Han, and S. X. Xia. 1999. Phosphorus and nitrogen transfers and runoff losses from rice field wetlands of Chaohu Lake. *Chinese Journal of Applied Ecology* 10(3): 312-316. (in Chinese).

Yang, H., and Z. Y. Ren. 2000. Study on nutrition evaluation of Chinese Lakes. *Jounral of Shaanxi Normal University (Natural Science Edition)* 28 (4): 104-107. (in Chinese).

Yang, W. L., and C. L. Yang. 2002. Dianchi Water environment capacity model research and calculation results. *Yunnan Environmental Science* 21 (3): 20-23. (in Chinese).

YPPG. 1998. National Ninth-Five-Year Plan and Long-Term Program on Water Pollution Prevention and Control for Year 2010 for Dianchi Lake Basin. In *Water Pollution Prevention and Control Plans and Programs for Huaihe River, Haihe River, Liaohe River, Taihu Lake, Chaohu Lake and Dianchi Lake in China*, edited by SEPA. Beijing. (in Chinese).

Zeng, X. K. 1999. Agrochemistry of phosphorus (V). *Phosphate & Compound Fertilizer* 14 (5): 55-59. (in Chinese).

Zhan, Y. R. 1995. Sample analysis of grain losses in China. *China Grain Economics* (4): 44-47. (in Chinese).

Zhang, L. 2002. Ecologizing Industrialization in Chinese Small Towns. *PhD Thesis*, Environmental Policy Group, Department of Social Science, Wageningen University, Wageningen.

Zhang, N. M., Y. Yu, B. Hong, *et al.* 2003. Factors influencing phosphorus loss by runoff process from farmlands in the Dianchi Watershed. *Environmental Science* 24 (3): 155-157. (in Chinese).

Zhang, S. Q. 2004. Research on NPK nutrients and rational fertilization technique in China. *Phosphate & Compound Fertilizer* 19 (2): 65-69. (in Chinese).

Zhang, T. W. 2001. Survey report on integrated treatment of water pollution of Dianchi Watershed in 2000. *China Rural Survey* (2): 32-38. (in Chinese).

Zhang, T. Z., and J. N. Chen. 2003. Industrial environmental management in China. In *Greening Industrialization in Asian Transitional Economies: China and Vietnam*, edited by A. P. J. Mol and C. L. J. van Buuren, p:23-38. Oxford: Lexington Books.

Zhang, T. Z., L. Shi, and Y. Liu. 2003. Plan for Industrial Ecological Economy of Jiangsu Province in China. Beijing: Department of Environmental Science and Engineering, Tsinghua University. (in Chinese).

Zhang, W. L., A. G. Xu, H. J. Ji, *et al.* 2004. Estimation of agricultural non-point source pollution in China and the alleviating strategies III. A review of policies and practices for agricultural non-point source pollution control in China. *Scientia Agricultura Sinica* 37 (7):1026-1033. (in Chinese).

Zhang, X. D. 1999. The development and prospect of China powder detergent market. *Detergent and Cosmetics* (5): 17-20. (in Chinese).

Zhang, X. M., T. Liang, and Y. B. Geng. 2001. The significance of sedimentary phosphorus in estuary-bay in studies on global change and regional response. *Progress in Geography* 20 (2): 161-168. (in Chinese).

Zhang, Z. X. 1996. Water pollution prevention and control of livestock husbandry in cities in China. *Urban Environment & Urban Ecology* 9 (1): 48-54. (in Chinese).

Zhang, Z. X., and Y. Qian, eds. 1998. *Urban Sustainable Development and Water Pollution Control Strategy*. Beijing: China Construction Industry Press. (in Chinese).

Zhou, X. Y. 2001. Demand of feed grains and adjustment of productive distribution in China. *Chinese Journal of Animal Science* 37 (2): 45-46. (in Chinese).

Zhou, Y. X., and L. J. C. Ma. 2003. China's urbanization levels: Reconstructing a baseline from the fifth population census. *The China Quarterly* (173): 176-196.

Zhuang, Y. H. 1997. Advances in global biogeochemical cycling studies. *Earth Science Frontiers* 4 (1-2): 163-168. (in Chinese).

Appendices

Appendix I: List of Fieldwork Participators

Participators involved in the first fieldwork, July 30 – August 10, 2000

Staff at Kunming Institute of Environmental Science, Kunming, Yunnan Province

Mr. He, Shu Zhuang (project co-manager)
Mr. Lu, Yi Feng (project coordinator)
Ms. Zhang, Kun Lin; Ms. He, Lin Hui; Ms. Xu, Yi Lei

Graduates at Department of Environmental Science and Engineering, Tsinghua University, Beijing

Prof. Chen, Ji Ning (project manager)
Mr. Liu, Yi (group coordinator)
Ms. Song, Lei; Mr. Yu Xin; Mr. Liu, Chao Xiang; Ms. Chen Qing; Mr. Qiu, Xiang Yang

Participators involved in the second fieldwork, July 2 – July 17, 2002

Participators at Department of Environmental Science and Engineering, Tsinghua University, Beijing

Prof. Zhang, Tian Zhu (project co-manager)
Mr. Liu, Yi (group coordinator)
Ms. Liu, Yan; Mr. Wang Bo; Mr. Yang, Bo Jing

Graduates (volunteers) at Southwest Forestry College, Kunming, Yunnan Province

Mr. He, Jia Fei; Ms. Lu, Mei; Mr. Wang, Tao; Mr. Jiang, Yanbin; Mr. Wu, Guo Fu; Ms. Li, Hong; Mr. Liu, Jie; Ms. He, Chun Yan; Mr. Wu, Xian Jun; Mr. Jiang, Xiao Lin; Mr. Lin, Jun Yu; Mr. Wang, Yuan Xing

Appendix II: List of Interviewees

At Kunming Municipal Level

Mr. He, Shu Zhuang, head, Kunming Institute of Environmental Science
Ms. Zhang Kun Lin, staff, Kunming Institute of Environmental Science
Mr. Lu Yi Feng, staff, Kunming Institute of Environmental Science
Mr. Zhang, Yu Feng, chief, Executive Office of the project "Non-Point Sources Pollution Control in the Dianchi Lake"
Mr. HuangPu, deputy director, Kunming Land and Resources Bureau
Mr. Cao, director; Kunming Environmental Protection Bureau
Mr. Wei, chief, Station for Agricultural Technology of Kunming
Mr. Liu, division chief, Administrative Office of Vegetables & Flowers, Kunming Agricultural Bureau
Mr. Jin, division chief, Administrative Office of Vegetables & Flowers, Kunming Agricultural Bureau
Mr. Zhang, division chief, Administrative Office of Energy & Rural Environment, Kunming Agricultural Bureau
Mr. Liu, division chief, Administrative Office of Economic Crops, Kunming Agricultural Bureau
Mr. Zhang, chief, Livestock & Veterinary Institute of Kunming Agricultural Bureau
Mr. Feng, division chief, Livestock Office of Kunming Agricultural Bureau
Mr. Wang, Yi Min, chief, Soil & Fertilizers Station of Kunming Agricultural Bureau
Mr. Zhou, division chief, Irrigation Works Construction Office of Kunming Water Conservancy Bureau
Mr. Yang, official, Irrigation Works Construction Office of Kunming Water Conservancy Bureau
Mr. Zhang, division chief, Energy Sources & Environment Office of Kunming Water Conservancy Bureau
Mr. Zhang, chief, Kunming Institute of Dairy Cow Breeding
Mr. Ming, Da Zeng, general manager, Sanhuan Chemical Industry Co. Ltd of Yunnan Province (Yunnan Phosphate Fertilizer Plant)
Mr. Liu, Shi Wen, environmental official, Sanhuan Chemical Industry Co. Ltd of Yunnan Province (Yunnan Phosphate Fertilizer Plant)
Mr. Shi, Jian Sheng, section chief, Kunyang Phosphorus Mine Corporation
Mr. Zhang, chief, Kunyang Phosphorus Mine Corporation
Ms. Chai, Li Qiong, chief executive, Kunyang Phosphate Fertilizers Plant
Anonymous officials, Kunming Dianchi Administration Bureau

At County Level

Mr. Duan, Ri Fu, director, Environmental Protection Bureau of Xishan District
Mr. Li, Xiang Qun, deputy director, Environmental Protection Bureau of Guandu District
Mr. Zhang, Shao Hua, director, Environmental Protection Bureau of Chenggong County
Mr. Zhang, Zeng Fu, Agricultural Development Office, Dounan Township of Chenggong County
Mr. Zhao, chief, Duty Office of Jinning Government
Mr. Zhao, Chun Quan, director, Environmental Protection Office, Enterprises Bureau of Jinning County
Mr. Teng, Xingliang, chief, Environmental Protection Office, Enterprises Bureau of Jinning County
Mr. Chen, director, Forestry Bureau of Jinning County
Mr. Li, the Party branch secretary, Forestry Bureau of Jinning County
Mr. Chen, Zhao Liu, director, Environmental Protection Bureau of Jinning County

Mr. Chen, Tian Lei, official, Environmental Protection Bureau of Jinning County

Anonymous officials, Development Planning Commission of Jinning County; Land and Resources Bureau of Jinning County; Jinning Agricultural Department; Water Conservancy Bureau of Jinning County; Water Affairs Bureau of Jinning County

Mr. Fan, chief, Water & Soil Conservancy Station of Jinning County

Mr. Yang, Xiao Qiang, deputy director, Agricultural Bureau of Jinning County

Mr. Zhang, deputy director, Agricultural Bureau of Jinning County

Mr. Yang, Shu Nan, chief, Agricultural Bureau of Jinning County

Mr. Cui, Guo Wu, chief executive, Gucheng Township, Jinning County

Anonymous official, Enterprises Duty Office, Shangsuan Township of Jinning County

Mr. Yang, Yong, Government of Xinjie Township, Jinning County

Mr. Huang, Ke Hong, Dairy Cow Village, Sunjiaba Township of Jinning County

At Village Level

Representative of villagers, Haifeng Village, Luchaiwan Village, Haimen Village, Baiyu Village, Haikou Township of Xishan District; Huiwan Village, Shanyi Village, Xihua Village, Biji Township of Xishan District; Yiliu Village, Guansuo Village, Ziwei Village, Zhaoxi Village, Yiliu Township of Guandu District; Baofeng Village, Zhongying Village, Haidong Village, Yaojiaba Village, Guandu Township of Guandu District; Xinghai Village, Liujia Township of Guandu District; Huilong Village, Xinjie Township of Jinning County; Chuying Village, Dunzi Village, Zhonghe Township of Jinning County; Hebo Village, Niulian Village, Shangsuan Township of Jinning County; Yao, Cun Hui, deputy chief, Qudong Village, Zhonghe Township of Jinning County

Mr. Tian, Cun Jin, the Party branch secretary, Dayu Village, Dayu Township of Chenggong County

Mr. Li, Chun Hai, the Party branch secretary, Dounan Village, Dounan Township of Chenggong County

Mr. Wang, Yong Guo, deputy executive, Dounan Village, Dounan Township of Chenggong County

Representative of villagers, Jiangwei Village, Dounan Township of Chenggong County

Mr. Zhang, Kunhong; Wulong Village, Dounan Township of Chenggong County

Mr. Zhang, Shou, Dianxin Village, Gucheng Township of Jinning County

Mr. Wang, the Party branch secretary, Baisha Village, Jincheng Township of Jinning County

Mr. Zhao, Dairy Cow Cooperative, Baisha Village, Jincheng Township of Jinning County

Anonymous owner, Dairy Cow Cooperation, Shangsuan Township of Jinning County

Mr. Li, Ru, Majiatang Village, Shangsuan Township of Jinning County

Mr. Du, chief, Sanhe Village, Xinjie Township of Jinning County

Notes: interviewees are listed according to the date of interview at different levels.

Appendix III: Data Sources for the PHOSFAD (1.1) Model

Industrial Sectors	Phosphate Reserve and Mining	Luo (1997); Deng (1998); Wang (1998); Zhao & Zhu (1999); Chen (2000); Huang (2001)
	Industrial Wastes and Wastewater	Wang (1998); KEPB (2002a); KEPB (2002b)
Agricultural Sectors	Crops Production	KAB & KAA (1999); Cao (2000); Cai (2000); KSB (2001); Xie & Tan (2001); Jia et al. (2002); Chen (2002)
	Fertilizers Application	Zhang & Qian (1998); KAB & KAA (1999); KSB (2001)
	Livestock Husbandry	Zhuang et al. (1996); Zhang & Qian (1998); Shu et al. (1999); KAB & KAA (1999); KSB (2001); Jia et al. (2002); KESRI (2002)
Urban and Rural Population	Human's Bio-Metabolism	Zhang & Qian (1998); Shu et al. (1999); KAB & KAA (1999); Shen (2001); KSB (2001); Interview (2002); Chen (2002)
	Domestic Waste and Wastewater	KAB & KAA (1999); Bian (2000); KSB (2002)
	Washing Powder Uses	Shu et al. (1999); KAB & KAA (1999); Qian (2000); KSB (2001); Interview (2002)

References for Data Collection:

Bian, Y. S., ed. 2000. *Wastes Disposal and Reuse for Ecological Agriculture*. Beijing: Chemical Industrial Press. (in Chinese).

Cai, X. M., ed. 2000. *General Introduction on Vegetable Cultivation*. Beijing: China Agriculture Press. (in Chinese).

Cao, Z., ed. 2000. *General Introduction on Crops Cultivation*. Beijing: China Agriculture Press. (in Chinese).

Chen, F., ed. 2002. *Agricultural Ecology*. Beijing: China Agricultural University Press. (in Chinese).

Chen, L. K. 2000. Problems in the development of phosphate rock resources in Yunnan province and the suggestions. *IM&P*, 5: 1-4. (in Chinese).

Deng, Q. 1998. Status quo and protection measures of the eco-environment along the Dianchi basin. *Yunnan Environmental Science*, 17(3): 32-34. (in Chinese).

Huang, Z. Q. 2001. Phosphate resources and its industrial development in Yunnan province. *Conservation and Utilization of Mineral Resources*, 2: 17-21. (in Chinese).

Interview with an anonymous official in KIES, July. 2002.

Jia X. H., Y. F. Huang, and J. T. Xu, eds. 2002. *Processing and Application of Organic Fertilizers*. Chemical Industrial Press, Beijing. (in Chinese).

KAB, and KAA. 1999. Survey report on agricultural non-point source pollution and control measures of Dianchi Watershed. Kunming: Kunming Agricultural Bureau and Kunming Agricultural Academy. (in Chinese).

KEPB. 2002a. *The Tenth Fiver Year Plan for Pollution Control for Dianchi Watershed*. Kumming: Kunming Environmental Protection Bureau. (in Chinese).

KEPB. 2002b. Monitoring Statistical Data for the Deadline Action for Industrial Pollution Sources Control in Dianchi Watershed (*Dianchi Lingdian Xingdong*). Kumming: Kunming Environmental Protection Bureau. (in Chinese).

KIES. 2002. Investigation Report on Livestock and Poultry Husbandry in Dianchi Watershed. Kunming: Kunming Institute of Environmental Science. (in Chinese).

KSB. 2001. *Kunming Statistical Yearbook 2000*. Beijing: China Statistics Press. (in Chinese).

Luo, F. Y. 1997. A discussion on the position and the development of the phosphate rock in Yunnan. *Yunnan Chemical Industry*, 4: 8-12. (in Chinese).

Qian, T. M. 2000. Consideration of the changes of phosphorus loading in water before and after prohibiting (limiting) phosphorus in detergent. *Chongqing Environmental Science*, 22(5): 38-40. (in Chinese).

Shen, Q. R. ed. 2001. *General Introduction on Soil and Fertilizer*. Beijing: High Education Press. (in Chinese).

Shu, J. H., W. Y. Huang, X. Y. Gao, et al. 1999. The evaluation of the effect of powdered detergent containing phosphate on Tai Hu Water Area Eutrophication. *Detergent and Cosmetics (Riyong Huaxuepin Kexue)* 22 (Supplement, August): 129-145. (in Chinese).

Wang, Y. F. 1998. Apprehension and foresight of phosphorous resource in Dianchi area, Yunnan province. *Geology of Chemical Minerals*, 20(2): 195-198. (in Chinese).

Xie, R. L., and H. W. Tan. 2001. The present and future demand of phosphate fertilizer in China in the light of agricultural production. *Phosphate and Compound Fertilizer*, 16(2): 6-9. (in Chinese).

Zhang, Z. X., and Y. Qian. 1998. *Urban Sustainable Development and Water Pollution Control Strategy*. Beijing: China Constructive Industry Press. (in Chinese).

Zhao, M. Y., and W. J. Zhu. 1999. Take advantage of comparison and promote the development of Yunnan chemical industry. *Yunnan Chemical Industry*, 1: 15-19. (in Chinese).

Zhuang, Y. H. *et al*. 1996. *Elements Flow of Complex Ecological System: A Case Study on Haihe River*. Beijing: China Environmental Science Press. (in Chinese).

Appendix IV: Data Sources for the PHOSFLOW (3.0) Model

Subsystem	Contents	Data Sources
Resource & Mining	Resource reserve, extraction and trading	NBS (2003a, 2003b); FAO (2003a); CNCIC (2002); JSDLR (2003)
Phosphate Industry	Production	NBS (2001a); CNCIC (2002); OCFI (2001); CNLIA (2001); Nie and Wang (2001); Chen (2002a); Qian (2000)
	Import and export	FAO (2003a); CNCIC (2002)
	Wastes and wastewater	SEPA (1996); CNCIC (2002); CEDT (1998); Nie (2000); Jiang (1999); Chen (2002b); Zhang et al. (2002)
Crops Farming	Production and trading	FAO (2003b); NBS (2001a); CGHF (1979); INFH (1992); Cao (2000); Jiang (2000); Chen (2002)
	Cultivated area and Fertilizers application	ISF (2003); NBS (2001a)
	Residues utilization and nutrient leaching	DRSD and CAAMS (2001); Lai (2004)
Animal Husbandry	Feed consumption	NBS (2001a, 2001b); NBS (2000); Tu and Huo (1997); Luo (2000); Lu (1997)
	Import and export	NBS (2001a); FAO (2003b); Fan (1983); Liu (1993)
	Waste disposal and nutrient losses	SEPA (2002); WB (2001); NBS (2000); Chen (2002); Jia et al. (2002); Shen (2001)
Urban and rural population	Food consumption	FAO (2003b); NBS (2001a, 2001b)
	Waste Treatment and nutrient losses	NBS (2001a); MOC (2001); MOH (2001); Bian (2000); Nie (2000); Yuan (2003); Pan et al. (1995)

References for Data Collection:

Bian, Y. S., ed. 2000. *Treatment and Recovery of Wastes in Ecological Agriculture*. Beijing: Chemical Industry Press. (in Chinese).

Cao, Z., ed. 2000. *Summa of Crops Planting Science*. Beijing: China Agricultural Press. (in Chinese).

CEDT, ed. 1998. *Handbook of Environmental Protection of Chemical Industry*. Beijing: Chemical Industry Press. (in Chinese).

Chen, F., ed. 2002. *Agricultural Ecology*. Beijing: China Agricultural University Press. (in Chinese).

Chen, S. J. 2002a. Prospects for the phosphorus industry in China. *Phosphate & Compound Fertilizer* 17 (5): 43-46. (in Chinese).

Chen, S. J. 2002b. The current production status and consumption ways of phosphorus in China. *Chemical Industry and Engineering Progress* 21 (10): 776-778. (in Chinese).

CGHF, ed. 1979. *Handbook of Fertilizers*. Beijing: China Agricultural Press. (in Chinese).

CNCIC, ed. 2002. *China Chemical Industry Yearbook 2001-2002*. Beijing: China National Chemical Information Center.

CNLIA, ed. 2001. *China Light Industry Yearbook 2001*. Beijing: Society of China Light Industry Yearbook.

DRSD, and CAAMS, ed. 2001. *Yearbook of China Agricultural Products Processing Industries 2001*.

Beijing: China Agricultural Press.

Fan, J. F., ed. 1983. *Food Nutrition and Health*. Shenyang: Liaoning People's Press. (in Chinese).

FAO. 2003a. *FAOSTAT Online Database*. Download on 12 December 2003.

FAO. 2003b. Food Balance Sheet for China in 2000. *FAOSTAT Online Database*. Download on 12 December 2003.

INFH, ed. 1992. *Food Composition Table*. Beijing: People's Hygiene Publisher. (in Chinese).

Jia, X. H., Y. F. Huang, and J. T. Xu, eds. 2002. *Manure Process and Utilization*. Beijing: Chemical Industry Press. (in Chinese).

Jiang, S. X., ed. 1999. *Phosphoric Acid, Phosphorous Fertilizer and Compound Fertilizer*. Beijing: Chemical Industry Press. (in Chinese).

Jiang, X. M., ed. 2000. *Summa of Vegetables Planting Science*. Beijing: China Agricultural Press. (in Chinese).

JSDLR. 2003. *Extraction and Utilization of Mineral Resources in China*. Nanjing: Jiangsu Department of Land and Resources. (in Chinese).

Lai, S. Y. 2004. Study on Non-point Sources Pollution Investigation and Evaluation Method and Its Application. *MSc Thesis*. Department of Environmental Science and Engineering, Tsinghua University, Beijing. (in Chinese).

Liu, Z. G., ed. 1993. *Food Nutrition*. Beijing: China Light Industry Press. (in Chinese).

Lu, W. G. 1997. Estimates on grain consumption for feed in China. *China Grain Economics* (3): 38-40. (in Chinese).

Luo, L. 2000. Feed industry in China: A challenge to the next millennium. *Feed Industry* 21 (5): 5-6. (in Chinese).

MOC, ed. 2001. *Urban Construction Statistics Yearbook of China Cities 2001*. Beijing: China Building Materials Industry Press.

MOH, ed. 2001. *Health Yearbook of China 2000*. Beijing: People’s Medical Publishing House.

NBS, ed. 2000. *China Statistical Yearbook 2000*. Beijing: China Statistics Press.

NBS, ed. 2001a. *China Statistical Yearbook 2001*. Beijing: China Statistics Press.

NBS, ed. 2001b. *Rural Statistical Yearbook of China 2001*. Beijing: China Statistics Press.

NBS, ed. 2003a. *China Industry Economy Statistical Yearbook 2003*. Beijing: China Statistics Press.

NBS, ed. 2003b. *China Statistical Yearbook 2003*. Beijing: China Statistics Press.

Nie, J. L., and P. L. Wang. 2001. Analysis of phosphor production and consumption. *Modern Chemical Industry* 21 (10): 48-52. (in Chinese).

Nie, Y. F., ed. 2000. *Handbook of Engineering and Techniques of Three-Wastes Treatment*. Beijing: Chemical Industry Press. (in Chinese).

OCFI, 2001. Review of feed industry production in China. Beijing: Office of China Feed Industry. Available at http://old.chinafeed.org.cn/slgj/Templates/2001-13/nair/2001-13-01.htm. (in Chinese).

Pan, S. C., G. H. Xu, Y. Z. Wu, *et al.* 1995. A background survey and future strategies of latrines and nightsoil treatment in rural China. *Journal of Hygiene Research* 24 (Supplement): 1-10. (in Chinese).

Qian, T. M. 2000. Consideration of the changes of phosphorus loading in water before and after prohibiting (limiting) phosphorus in detergent. *Chongqing Environmental Science* 22 (5): 38-40. (in Chinese).

SEPA. 1996. *Handbook for Industrial Pollutants Production and Discharge and Emission Coefficients*. Beijing: China Environmental Science Press. (in Chinese).

SEPA. 2002. *Investigation on the Pollution of National Livestock and Poultry Breeding in Scale and Its Control Strategy*. Beijing: China Environmental Science Press. (in Chinese).

Shen, Q. R., ed. 2001. *General Introduction on Soil and Fertilizer*. Beijing: High Education Press. (in Chinese).

Tu, Y., and Q. G. Huo. 1997. Phosphorus content in feeds and bioavailability. *Feed Industry* 18 (11): 27-30. (in Chinese).

WB. 2001. *China: Air, Land, and Water*. Washington, D.C.: The World Bank.

Yuan, Y. 2003. Present situation of our country's urban domestic garbage and countermeasures. *Sci/Tech Information Development and Economy* 13 (7): 59-60. (in Chinese).

Zhang, Y. X., S. H. Xu, and J. Li. 2002. Phosphorus pollution and its prevention and control measures. *Environmental Protection of Chemical Industry* 22 (2): 68-70. (in Chinese).

Appendix V: Refinements of the PHOSFAD (2.2) Model

(Unit: metric tons P)

Flow		Value		Description and Modification
Vs. 1.1	Vs. 2.2	Vs. 1.1	Vs. 2.2	
A. Mining and Industry Subsystems				
R_S	R_S	270.6*10^4	1976*10^3	Extracted minerals: revision
S_1	S_1	135.3*10^4	954*10^3	Mining tailings: revision
M_1	M_1	67.4*10^4	756*10^3	Industrial processed minerals: revision
M_2	E_1	67.6*10^4	266*10^3	Output minerals: revision
I_1	E_2	42.5*10^4	506*10^3	Output industrial products: rebalance
I_2	P_1	4049.6	4589	Chemical fertilizer application: rename & revision
I_3	P_2	500.6	252	Detergent demand of urban population: revision
I_4	P_3	218.4	118	Detergent demand of rural population: revision
B. Farming Subsystem				
F_2	F_1	1387.0	1387	Crop harvest: rename
F_3	F_2	383.7	384	Crop residues: rename
F_5	F_3	123.8	21	Residues reuse for farmlands: rename & rebalance
F_6	F_4	40.4	40	Residues reuse for household fuel: rename
	F_5		55	Residues reuse for scaled breeding: recruitment
F_4	F_6	186.9	132	Residues reuse for family breeding: rebalance
F_7		841.9		Composing manure: removal
	E_3		191	Crop products output: recruitment
F_1	F_S	5089.9	5120	Nutrient deposit in soil: rename & rebalance
N_1+N_3	N_1+N_3	3085.6	1485	Food/grain demand of urban population: rebalance
N_2+N_4	N_2+N_4	697.6	679	Food/grain demand of rural population: rebalance
N_7	N_5+N_7	351.2	497	Grain demand of scaled breeding animal: rebalance
N_8	N_6+N_8		1597	Grain demand of family breeding animal: rebalance
	S_3		69	Crop wastes during processing: recruitment
	S_4		119	Residues dumping: recruitment
W_6	W_3	202.5	351	Farmlands leaching: rename & revision
W_{10}	W_4	16.1	16	Residues leaching: rename
C. Livestock Subsystem				
L_4	L_1	184.8	484	Reuse of scaled-breeding animal manure: rename & revision
	L_2		40	Livestock products demand by urban population: recruitment
L_1	L_3	1199.2	1534	Reuse of family-breeding animal manure: rename & revision
L_2		537.2		Composting manure of family-breeding animal: removal
	L_4		63	Livestock products demand by urban population: recruitment
	L_5		18	Livestock products demand by rural population: recruitment
L_3	L_6	110.9	98	Animal excreta reuse for biogas generation: revision
	E_4		14	Livestock products output: recruitment
	S_5		92	Dumping of family-breeding animal dung: recruitment

Flow		Value		Description and Modification
Vs. 1.1	Vs. 2.2	Vs. 1.1	Vs. 2.2	
W_{12}	W_5	166.3	28	Nutrient loss of scaled feedlots: rename & rebalance
W_{11}	W_6	226.0	108	Nutrient loss of family farms: rename & rebalance
D. Rural Population Subsystem				
R_2	R_1	698.3	420	Food consumption: rename & revision
R_1	R_2	825.2	324	Domestic garbage: rename & revision
R_3	R_3	218.4	118	Detergent consumption: revision
R_6	R_4	37.4	23	Human excreta reuse for biogas generation: removal
R_4	R_5	403.9	352	Recycling of human excreta: revision
R_5		180.9		Composing of human excreta: removal
N_9				Organic garbage reuse for biogas generation: removal
N_{10}	R_6		199	Kitchen wastes reuse for feedstuff: rename & refinement
	R_7		21	Organic garbage reuse to farmlands: recruitment
	R_8		121	Biogas wastes reuse to farmlands: recruitment
	S_6		21	Dumping of human excreta: recruitment
	S_7		77	Dumping of domestic garbage: recruitment
W_7	W_{10}	76.1	25	Leaching of human excreta: rename & rebalance
W_9	W_{11}	188.0	28	Leaching of garbage: rename & rebalance
W_8	W_{12}	218.4	118	Discharge of grey water: rename & rebalance
E. Urban Population Subsystem				
U_1	U_1	1600.8	895	Food consumption: revision
U_2	U_2	1389.0	698	Domestic garbage: revision
U_3	U_3	500.6	252	Detergent consumption: rebalance
U_4	U_4	1329.5	200	Collection and transportation of human excreta: revision
U_5	U_5	271.3	377	Treatment of black wastewater: rename & revision
U_6	U_6	1161.0	556	Collection of domestic garbage: rename & revision
U_7	U_7	309.4	156	Treatment of grey wastewater: rename & rebalance
	U_8		86	Recycling of human excreta: recruitment
	U_9		34	Recycling of urban garbage: recruitment
	S_8		97	Uncollected domestic garbage: recruitment
S_3	S_9	2490.5	756	Disposal of urban wastes: rename & rebalance
S_4	S_{10}	430.3	394	Disposal of WWTP sludge: rename & rebalance
	W_6		233	Direct discharge of domestic wastewater: recruitment
W_5	W_7	228.0	11	Leaching of garbage: rename & rebalance
W_4	W_8	191.2	96	Direct discharge of grey water: rename & rebalance
W_3	W_9	150.4	138	Discharge of treated wastewater: rename & rebalance

Notes: black water refers to efflux of household toilet; grey water refers to domestic wastewater discharged by showering, washing, cooking, etc.

Summary

The substantial shifts in the structure of domestic phosphorus (P) throughput, during a crucial period of social and economic transitions, brings forth a challenge for curbing the extensive eutrophication of surface waters in China. The lack of an overall knowledge on the societal metabolism of P has contributed to the absence of eutrophication policy and efficient measures in China. Therefore, it is vital to reflect on the fundamental economic and environmental characteristics of physical P flows, in order to reinvent ecologically rational solutions for a successful eutrophication control in China.

This study develops a comprehensive research approach for inquiring into the societal P cycles within China. The methodology, combining a substance flow analysis (SFA) approach and an environmental policy evaluation method, starts with a quantitative analysis of the entire life cycle of societal P metabolism. Subsequently, it identifies the critical P flows with crucial environmental consequences and then relates them to an evaluation of the environmental policies directed at these P flows. The Dianchi Basin, one of the national key environmental protection watersheds, has been selected as a case study area, both for characterizing the social P metabolism and for evaluating the policies to reduce eutrophication.

Two static SFA models for national (PHOSFLOW) and regional (PHOSFAD) P regimes have been developed, respectively, for quantitatively estimating unknown but critical flows. These models reveal how a nation or a region extracts P from nature, metabolizes it via various economic activities, and finally excretes P as waste into the environment. The analysis of the P metabolic structure in general shows a clear form: the societal P metabolisms, both within the national and local economies, have developed into a once-through system where P is mobilized in large quantities from nonrenewable rocks and subsequently leaks into either the (natural or cultivated) soil or surface water. In analyzing the ecological restructuring of China, that is, examining the environmental effectiveness and economic efficiency of critical P flows throughout time, it was found that China has become more efficient in the use of P materials over the last two decades, measured per unit of national economic growth. But the absolute quantities of these P flows have increased, which is of course especially relevant for the environment.

The SFA models were deliberately modified by applying a set of data processing principles and new data series. While the updated P fluxes of the static SFA models are arguably better reflecting their counterparts in the natural cycle, the process itself has demonstrated a gradual increase in the knowledge on societal P cycles. The modification also facilitates a normative (re)formulation of the main findings of these SFA models. This has enabled an understanding of the similarities and differences of the physical characteristics of P throughput between the national and the local economies: 1) the local P throughput in Dianchi Basin was basically export-oriented while relying on domestic reserves; 2) the local P fluxes were more intensive in quantity compared to those at the national level; 3) the import and export of agricultural goods was, in general, balanced at both the national and local levels, except for the crop trading in Dianchi Basin; 4) the increasingly widespread cultivation of vegetables and flowers in Dianchi Basin resulted in an intensive application of P fertilizer; 5) the large-scale feedlots and farms in Dianchi Basin did less harm to the environment than the small family-scale units; 6) the physical configurations of P flows associated with urban and rural residents were mostly similar, the exception being the urban sewage system, which proved more efficient in terms of P removal in Dianchi Basin.

The SFA models enable the identification of critical P flows that should be the central object of concern for environmental policy and regulation. The P flows associated with intensive livestock, urban centralized sewage systems and P-detergents are the appropriate candidates for this concern. These flows also have the attention of environmental authorities in contemporary China.

In order to assess how and to what extent the environmental policies have successfully (re)directed these P flows to generate environmental benefits for eutrophication control, a policy evaluation framework proposed by the European Environment Agency (EEA) is applied. This methodology facilitates a systematic investigation into both the policy design in terms of the relevance of policy objectives and the policy implementation with regard to evaluating issues of economic efficiency, environmental effectiveness and overall social impact.

The evaluated environmental policies contain two main societal responses to P- caused eutrophication: an endeavor to prevent P from flowing into waterbodies, and an attempt to reduce P use at the household level. The end-of-pipe approaches that remove P from effluent do not necessarily result in a reduction of P input or an increase in P recycling, both of which are underlying propositions of an ecologically rational eutrophication policy. At the same time, the current and potential environmental effects of these approaches in terms of P outflow reduction are limited, due to a large number of economic, social and institutional barriers. The ban of P-detergents – a source-control strategy – could considerably reduce the P input in the economy. As reduced P consumption results in the introduction of substitute materials, this policy could lead to undesired negative environmental impacts. In this way, the P control approach at source could become less promising. Furthermore, the P-detergent ban, like the other two end-of-pipe strategies, relies mainly on government-oriented regulatory measures, while other measures, especially market-based economic incentives, are to a large extent ignored.

The application of the approach and methodology of this study proves its innovative value for China. First, this research produces two P flow models and a normative evaluation of three clusters of P-related environmental policies, which can be understood as an innovation in contemporary research on P in China. The comprehensive insight to P flows, at both national and local levels, provides a scientific basis for strengthening and further developing the eutrophication control framework. Second, by introducing the substance flow analysis method and the policy evaluation method into China, this thesis offers a promising research paradigm for reflecting on the fundamental societal material cycles, which is particularly important for the ongoing campaign of establishing a circular economy in China.

Samenvatting

De substantiële veranderingen in de structuur van de binnenlandse fosfor (P) productie tijdens een periode van sociale en economische transitie brengt een uitdaging om de grootschalige eutrofiering van de Chinese oppervlaktewateren tegen te gaan. Het gebrek aan kennis over het lot van P na de winning va fosfaaterts en aanwending in verschillende productieprocessen heeft bijgedragen aan de afwezigheid van een effectief beleid en maatregelen op het gebied van eutrofiering. Daarom is het belangrijk om de economische en milieuhygiënische karakteristieken van P stofstromen nader te bestuderen om te komen tot adequate oplossingen om eutrofiering in China te beteugelen.

In dit onderzoek is een uitgebreide onderzoeksmethodiek ontwikkeld om meer inzicht te krijgen in de maatschappelijke P cycli in China. Deze onderzoeksmethodiek combineert twee verschillende methodes: een “substance flow analysis” (SFA) en een milieubeleid evaluatiemethode. De methodiek begint met een kwantitatieve analyse van de gehele levenscyclus van het maatschappelijke P metabolisme. Daaropvolgend worden de P stromen met belangrijke milieuhygiënische consequenties geïdentificeerd en deze worden gekoppeld aan een milieubeleidsanalyse op het gebied van deze P stromen. Naast een nationale analyse is het Dianchi bekken gekozen als regionaal onderzoeksgebied, zowel op het gebied van P stromen als op het gebied van de milieubeleidevaluatie.

Twee statische SFA modellen voor nationale (PHOSFLOW) en regionale (PHOSFAD) stofstromen zijn gebruikt om onbekende, maar belangrijke, P stromen te kwantificeren. Deze modellen laten zien hoe een land of regio P wint, dit via verschillende economische activiteiten omzet en het uiteindelijk weer in het milieu uitstoot. De analyse van de stofstromen van P laat een duidelijk, consistent beeld zien. De maatschappelijke P stofstromen in de nationale en de regionale economieën zijn stromen waarin P slechts één maal gebruikt wordt. P wordt in deze stromen gemobiliseerd uit gesteente en wordt daarna uitgestoten naar de bodem of het oppervlaktewater. Wanneer de milieuefficiëntie en de economische efficiëntie van belangrijke P stromen worden geanalyseerd, wordt het duidelijk dat China de afgelopen 20 jaar een hogere efficiëntie heeft behaald gemeten per eenheid nationale groei. De absolute hoeveelheden in de P stromen zijn echter gegroeid, hetgeen natuurlijk relevant is voor de uiteindelijke effecten van deze stromen op het milieu.

In het onderzoek zijn de bovenstaande SFA modellen vervolgens aangepast door het gebruik van nieuwe data en door het toepassen van een groep principes op het gebied van dataverwerking. Daar deze aangepaste P fluxen van de statische SFA modellen beter in staat zijn om de natuurlijke P cycli te simuleren, heeft dit proces een geleidelijke groei van kennis over de maatschappelijke P cyclus opgeleverd. Deze aanpassing vergemakkelijkt ook een normatieve herformulering van de belangrijkste uitkomst van deze modellen. Dit heeft een beter begrip opgeleverd van (de overeenkomsten en verschillen tussen) de fysieke P stromen in de regionale en nationale economie: 1) de locale P stofstromen in het Dianchi bekken waren vooral gericht op de export, terwijl vooral gebruik werd gemaakt van eigen reserves voor P productie; 2) de lokale P fluxen waren groter dan de fluxen op nationaal niveau; 3) de import en export van agrarische producten was in het algemeen in evenwicht. Een uitzondering hierop vormde de handel in gewassen in het Dianchi bekken; 4) de toenemende teelt van groenten en bloemen (vooral in het Dianchi bekken) heeft geleid tot een intensiever gebruik van P meststoffen; 5) de grootschalige percelen en boerderijen in het Dianchi bekken brachten minder schade toe aan het milieu dan de kleine familiepercelen; 6) de fysieke vormen van P stromen van plattelandsbewoners en stadsbewoners waren grotendeels gelijk; een uitzondering hierop was het stedelijke rioleringsysteem dat meer efficiënt was in het verwijderen van P in het Dianchi bekken.

De SFA modellen maakten het mogelijk de belangrijkste P stromen voor milieubeleid en -regulering te identificeren. Tot deze belangrijkste P stromen behoren de stromen die zijn gerelateerd aan intensieve veehouderij, stedelijke rioleringsystemen en fosfaathoudende wasmiddelen. Deze stromen zijn ook aandachtspunten in het huidige Chinese beleid.

Om nader te onderzoeken hoe, en in welke mate, het milieubeleid effectief is geweest op het gebied van het reguleren van P stromen is een beleidsevaluatie methode gebruikt van het Europees Milieu Agentschap (European Environment Agency, EEA). Deze methode omvat een systematische analyse van de beleidsontwikkeling, van de relevantie van de beleidsdoelen en van de implementatie van het beleid. In deze methode wordt daarbij gekeken naar de economische efficiëntie, milieueffectiviteit en de sociale gevolgen van dit beleid.

Het geëvalueerde beleid kent twee belangrijke type maatregelen voor P gerelateerde eutrofiering: emissiegerichte maatregelen om te voorkomen dat P in water terechtkomt en brongerichte maatregelen om het gebruik van fosfor op huishoudelijk niveau terug te brengen. De eerste categorie end-of-pipe oplossingen, die P verwijderen uit afvalwater, hoeven niet noodzakelijkerwijs te leiden tot een verminderde P uitstoot of een toename van P hergebruik, hetgeen wel uitgangspunten zijn van een ecologisch rationeel eutrofiëringbeleid. Daarnaast zijn de huidige en potentiële milieueffecten van deze end-of-pipe maatregelen beperkt door verschillende economische, sociale en institutionele barrières. Een verbod op fosfaathoudende wasmiddelen zou de P instroom in de economie aanzienlijk kunnen verkleinen. Verminderde P consumptie leidt echter tot gebruik van vervangende materialen en stoffen en dit zou kunnen leiden tot ongewenste negatieve milieueffecten. Daarom kan deze brongerichte methode minder aanlokkelijk zijn. Ook is deze methode, net als de andere end-of-pipe methodes, grotendeels georiënteerd op de regelgeving door overheid terwijl andere type instrumenten, zoals economische maatregelen, voor een groot deel over het hoofd worden gezien.

De ontwikkeling en het gebruik van de aanpak in deze studie bewijst haar innovatieve waarde voor China. Ten eerste zorgt dit onderzoek voor twee P stroommodellen en een normatieve evaluatie van drie clusters van maatregelen in het P gerelateerde milieubeleid. Dit is een innovatie in het huidige onderzoek naar P in China. Dit uitgebreide inzicht in P stromen, op nationaal en regionaal niveau, levert een wetenschappelijke basis om een nieuw raamwerk voor het tegengaan van eutrofiering te ontwikkelen. Ten tweede zorgt de introductie van een "substance flow analysis" en een beleidsevaluatie in China voor een veelbelovend onderzoeksparadigma om meer inzicht te krijgen in de fundamentele maatschappelijke stofstromen. Dit is vooral belangrijk in het licht van de voortgaande campagne om in China een economie met gesloten kringlopen te vestigen.

About the Author

Yi Liu was born on 22 October 1975 in Lanzhou city, Gansu province, China. He obtained his Bachelor and PhD degree in Environmental Engineering at Tsinghua University in July 1999 and July 2004, respectively. In March 2001, he came to Wageningen University to pursuit a second doctorate.

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