WHAT'S AT STEAK?

ECOLOGICAL ECONOMIC SUSTAINABILITY AND THE ETHICAL, ENVIRONMENTAL, AND POLICY IMPLICATIONS FOR GLOBAL LIVESTOCK PRODUCTION

by

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I do not know much about gods; but I think that the river
Is a strong brown god - sullen, untamed and intractable,
Patient to some degree, at first recognized as a frontier;
Useful, untrustworthy, as a conveyor of commerce;
Then only a problem confronting the builder of bridges.
The problem once solved, the brown god is almost forgotten
By the dwellers in cities - ever, however, implacable.
Keeping his seasons and rages, destroyer, reminder
Of what men choose to forget. Unhonoured, unpropitiated
By worshippers of the machine, but waiting, watching and waiting.
His rhythm was present in the nursery bedroom,
In the rank ailanthus of the April dooryard,
In the smell of grapes on the autumn table,
And the evening circle in the winter gaslight.

The river is within us, the sea is all about us.

- T.S. Eliot

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ABSTRACT

Achieving environmental sustainability in human organization is the defining challenge of the modern era. In light of the inability of the existing economic paradigm to provide for sustainability objectives, novel approaches to understanding and managing economic activities are required. Towards this end, the emergent field of ecological economics provides an alternative paradigm that expressly prioritizes the development of the theory and tools necessary to operationalize environmental sustainability in economic activity, which is viewed as prerequisite to sustainability in any other sphere. Here, I advance an internally consistent framework for understanding and implementing the core ecological economic sustainability criteria: appropriate scale relative to biocapacity; distributive justice; and efficient allocation. This framework includes: (1) an ecological communitarian conception of distributive justice which recognizes environmental sustainability as the first principle of distributive justice; (2) the rationale for biophysically-consistent ecological economic modeling of human activities as a basis for environmentally-enlightened policy and management; and (3) an appeal for scale-oriented environmental governance as could potentially be operationalized by a strong, centralized World Environment Organization. I further apply this framework to evaluating the current and future status of livestock production systems at regional and global scales with respect to efficiency considerations as well as their relationships to sustainability boundary conditions for human activities as a whole. It is suggested that the current and projected scale of the livestock sector is fundamentally unsustainable, and that all leverage points must be exploited to rein in this sector in the interest of preventing irreversible ecological change. This must include, but cannot be limited to, strong eco-efficiency measures and changes in production technologies, species substitutions, and consumption patterns and volumes. Outcomes are interpreted in terms of their implications for environmental policy and governance oriented towards the sustainability objective.

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"If I have seen further, it is only by standing on the shoulders of giants"

- Isaac Newton

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CHAPTER 1: INTRODUCTION

1.1 Dissertation Overview

The rapid rise and globalization of industrial society has been accompanied by widespread and increasingly destabilizing environmental degradation (Goldblatt 1996; Sklair 2002; MEA 2005). Concerns regarding the long-term sustainability of contemporary society have thus come to occupy a prominent position in global political discourse. An important, although insufficiently prominent, component of this discourse hinges on the extent to which our contemporary form of economic organization contributes to these challenges. A central question is whether sustainability can be achieved within the existing economic paradigm, or if the paradigm itself is pathological (Goldblatt 1996; Sklair 2002). If the latter is true, we must ask, what alternative models of economic organization might we entertain in the interest of sustainability?

According to an oft-cited definition, economics is the study of the allocation of scarce resources among competing ends. Such a definition begs several questions: What ends are appropriate; what scarce resources does their attainment require; and by what criteria should the allocation of means to ends be prioritized (Daly and Farley 2004)? At a societal level, the answers to these questions collectively contribute to our understanding and administration of economic activity and legitimize the resultant form of economic organization. Since they effectively delineate and circumscribe accepted norms for material relations, both within the recognized constellation of economic actors and between those actors and the artifacts and processes upon which economic activities impinge, they are critical to defining and understanding the worldview of a given society. They are similarly instrumental to determining its consequences in the material world and the scope of policies that might be entertained to mitigate undesirable consequences within the existing paradigmatic boundaries.

The purposes of this dissertation are several, and complementary in their pursuit of understanding the minimum conditions necessary for ensuring the environmental sustainability of economic activity. They may be broken down into six key foci.

First is to compare and contrast two paradigms that provide competing conceptions of the appropriate theoretical and practical foundations for the interpretation and organization of economic activity with respect to sustainability objectives. One is the currently dominant neoclassical economic paradigm. The other is the emergent paradigm of ecological economics, which has its genesis in perceived deficiencies of the neoclassical model in its vision of the economic actor and the relationships between economic activity and the biosphere. More specifically, the central question here is the practical efficacy of these two paradigms in serving the overarching objective of structuring economic activity in the interest of long-term environmental sustainability. This requires examination of both the positive and normative aspects of these models and their implications for sustainability objectives. To this end, Chapter Two briefly describes the major tenets of the neoclassical economic model. This description is provided in broad strokes only, in order to illustrate the core suite of assumptions that are the basis of critique. Chapter Three outlines the emerging paradigm of ecological economics, and compares and contrasts its theoretical orientations with those of the neoclassical model.

The utility of any body of economic theory is determined by its capacity to provide an internally consistent basis for understanding and managing the interactions of economic actors and the implications of their behaviours for specified societal objectives. The overarching objective of ecological economics is ensuring the environmental sustainability of economic activities. Ecological economists prescribe three criteria for environmentally sustainable economic activity: appropriate scale (relative to environmental carrying capacity), efficient allocation, and just distribution (Daly 1992; Daly and Farley 2004). Satisfying the first two criteria demands an empirical approach in order to accurately describe the relationships between the material and energy demands and emission intensities of discrete and cumulative economic activities and the resource provisioning and waste assimilatory capacity of host ecosystems. This requires biophysical accounting methodologies consistent with the thermodynamic basis of ecological economic theory. The third (just distribution) criterion is a normative issue whose resolution must proceed from the articulation of an ethically grounded vision for economic organization. Specifically, for ecological economics, such a framework must provide prescriptive guidance for an environmentally sustainable mode of economic organization that simultaneously resolves issues of intra- and intergenerational distributive justice for both human communities and the broader web of mutually constituting biotic communities and abiotic variables of which we are part. Furthermore, its articulation is prerequisite to defining the criteria for efficient allocation, and for legitimizing the relevance of both the scale criterion as well as the ecological economic orientation towards sustainability, generally. To date, it appears that a broadly accepted normative framework has not been advanced in support of the ecological economic vision.

The second purpose of this dissertation is therefore to sketch out the potential contours of such a vision. Chapter Four surveys the major traditions of distributive justice in western moral philosophy, as well as the emerging environmental ethics literature. To the extent that they have been articulated, the limited treatment of normative orientations in the ecological economics literature is reported, and these are contrasted with the normative elements of the neoclassical model. It is proposed that ecological economics as a normative enterprise oriented towards sustainability is best legitimized by an ecological communitarian ethic based on a conception of the economic actor as ecological community member. From this perspective environmental sustainability, which is seen to be prerequisite to sustainability in any other sphere, is recognized as the first principle of distributive justice. Here, ethics are viewed as evolutionary-adaptive phenomena which collectively promote the flourishing of the community in which they arise, in part by informing behavioural norms consistent with context-specific ecological constraints. For industrial society, as for all self-organizing systems, the thermodynamic principles central to ecological economics are seen to provide core guiding principles for the structuring and maintenance of an ethically-oriented ecological political economy.

Since ecological economics focuses on the relationships between economic activities and the biophysical environment, tools conducive to evaluating these relationships are required. Specifically, ecological economists require accounting frameworks appropriate to elucidating the environmental dimensions of economic activity relative to efficiency and scale considerations. Chapter Five reviews the spectrum of contemporary ecological economic biophysical accountancy-type instruments that have been developed to date, with a focus on the life cycle assessment framework, which is applied as the basis of the analyses reported in subsequent chapters.

Despite the increased prevalence of biophysical accounting using these instruments, and the rising prominence of the ecological economic perspective generally, a review of current practice suggests that much of this work is being conducted in a sub-optimal manner. As argued here, the prevalent use of price-based market information in biophysical accounting exercises introduces the distorting influences of the neoclassical economic model into what are otherwise intended to be biophysical models of human activities. As a consequence, the information produced often simply reflects and reinforces existing, environmentally myopic market signals rather than providing the biophysically consistent information necessary to understand and manage the environmental dimensions of economic activity for sustainability objectives.

The third purpose of this thesis is thus to advance a critique of contemporary practice, and a viable alternative for future research. Chapter Six presents a critique of the currently prevalent use of market information in biophysical accounting endeavours, using life cycle assessment research of food systems as an example. Specifically, the rationale and methods for biophysically-consistent application of life cycle assessment models to understanding the environmental dimensions of human food production activities are developed.

The fourth purpose of this work is to assess the implications of the ecological economic perspective for environmental policy and governance, which is ultimately necessary to operationalize an ecological political economy. While this will require institutions commensurate with the scale of human/environment interactions of concern, the focus here is on governance of the macroscale environmental dimensions of economic activities, which require commensurate, global scale governance institutions. Chapter Seven hence applies the ecological economic perspective to explain and offer partial recourse to the failures of contemporary global environmental governance. It is argued that a scale-based approach to governing the global environmental commons, operationalized by a strong World Environment Organization with appropriate decision making, legislating and enforcement capacities, is prerequisite to ensuring the environmental sustainability of industrial society.

In concert, the proposed ethical (ecological communitarian), methodological (biophysically-consistent ecological economic modeling) and policy (scale-based global environmental governance) elements advanced are intended to provide an internally consistent framework for the positioning, execution and interpretation of ecological economic research. On this basis, the fifth purpose of this dissertation is to apply this ecological economic research framework to elucidating a subset of the environmental dimensions of a specific sector of economic activity, and the attendant implications for environmental policy and management oriented towards the sustainability objective.

The economic sector of interest is the global animal husbandry industry, which has been implicated as contributing to a variety of critical environmental issues at local, regional and global scales (Steinfeld *et al.* 2006). Chapter Eight reviews the role of food systems generally, and livestock production specifically, in anthropogenic environmental change, along with recent biophysical accountancy-type food system sustainability research. The rationale is provided for assessing the contributions of the livestock sector to three macroscale environmental concerns, and for using life cycle assessment to evaluate livestock production systems using four core impact assessment methods along with three additional biophysical “return on investment” metrics. This core suite of metrics provides a working example of ecological economic accounting and its relevance to understanding and managing economic activities for environmental sustainability objectives.

On this basis, the next objective is to evaluate the current and projected contributions of global livestock production and consumption in aggregate to anthropogenic environmental change from the perspective of sustainable scale. Towards this end, Chapter Nine describes a scenario modeling exercise which couples published estimates of aggregate greenhouse gas emissions, biomass appropriation, and reactive nitrogen mobilization associated with the global livestock sector in 2000 with conservative estimates of marginal impacts per unit production beyond year 2000 levels and United Nations Food and Agriculture Organization (UNFAO) estimates of livestock production and consumption in 2050. Aggregate impacts in 2050 are estimated, and the mitigation potentials of several scenarios involving product and protein source substitution are evaluated. These scenarios are related to estimates of sustainability boundary conditions for anthropogenic activities as a whole in each of these spheres. The outcomes are couched in terms of their implications for food and environmental policy.

Having established the relationships between the livestock sector and current understandings of sustainable scale for three critical dimensions of global-scale environmental concern, the next objective is to consider livestock production from the perspective of efficiency. Chapters Ten through Twelve report three separate life cycle assessment research projects that evaluate three major intensive animal husbandry sectors, along with several emerging alternative production technologies, as currently practiced in the United States. The focus here is on describing the flows of material, energy, and wastes associated with these production technologies, identifying key leverage points for eco-efficiency improvements within existing production systems, and making comparisons between competing production technologies along four important dimensions of environmental performance: cumulative energy use, greenhouse gas emissions, eutrophying emissions, and ecological footprint. In addition, three further biophysically-based energy-return-on-investment (EROI) metrics are applied to evaluate these systems in terms of non-renewable and renewable energy resource efficiencies from both anthropocentric and ecological perspectives. In concert, these seven measures are intended to contribute to a basis for understanding and managing livestock systems for ecoefficiency objectives in order to serve the overarching imperative of sustainable scale.

They further provide a strong example of the kinds of biophysical considerations that must be brought to bear in understanding and managing economic activities, generally, for environmental objectives.

The sixth and final purpose of this dissertation is to synthesize these ecological economic analyses of livestock production as seen through the lens of the ecological economic sustainability criteria of appropriate scale, just distribution, and efficient allocation. Towards this end, Chapter Thirteen provides an integrated discussion of the work presented in this dissertation. It begins with a review of the ecological economic research framework developed. This is followed by an assessment of the key insights that emerged from the application of this framework to evaluating livestock production at global and regional scales. This includes: (1) the ethical imperatives that must be brought to bear in determining the extent to which current and alternative future configurations of livestock production and consumption serve the ecological communitarian vision of environmental sustainability as the first principle of distributive justice; (2) the core insights, with respect to sustainable scale and efficient allocation, which flow from the empirical models of global and regional livestock production presented here; and (3) the attendant implications for global environmental governance looking forward. The latter discussion addresses the current state of global environmental governance regimes and their limited capacity to respond to the issues presented. It is concluded that redressing sustainability concerns in the livestock sector will likely require the intervention of a strong, centralized global environmental governance body. The limitations of the current work, as well as directions for future research, are also presented.

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CHAPTER 2: NEOCLASSICAL ECONOMICS: THE PARADIGM OF EGO AND ABUNDANCE

2.1 Philosophical Roots Of Neoclassical Economics

The roots of neoclassical economics, which comprises the body of theory, tools and practice central to the dominant, contemporary economic paradigm, can be traced to the moral philosophy of the Enlightenment period (Costanza *et al.* 1997; Holder 2000). In many ways, this era represented the most important watershed in Western philosophical thought. It was during this time that fundamental notions regarding the nature of the self were redefined and arguments with respect to the necessity of material security to moral progress gained currency (Costanza *et al.* 1997). Also in flux were long-standing ideas of the responsibility of individuals in contributing to social well-being and the relationship of humans to nature. Key contributors included John Locke, Rene Descartes, Isaac Newton and Adam Smith (Canterbury 1987; Nelson 1991).

According to Locke, a just and efficient social order requires enclosure of the commons via private property rights guaranteed by the state (Barry 1999). Recognizing the motivation derived from ownership of the fruits of one's labour, Locke based his position on the argument that property is a natural right essential to liberty. He further argued that the accumulation of private property is justified, so long as 'as much and as good' is left for others. A profound implication of this philosophy was the legitimacy it provided for European colonial conquest of the New World, and private appropriation of the seemingly boundless material frontiers afforded.

The rise of scientific empiricism, based on the subject/object dualism of Descartes and Newtonian atomism, further promised a manipulable world in which novel technology would allow leverage of limitless riches from a bountiful nature. This shift in perception of the human/nature relationship supported a view of non-human nature as nothing more than instrumental resources, wasted if left fallow. The self/other dichotomy further served to elevate the ideal of the individual as a discrete and isolated entity over that of the individual as community member (Lazlo 2001).

Adam Smith provided the keystone for contemporary economic theory with his argument that individual self-interest is actually socially beneficial (Lux 1990; Costanza *et al.* 1997). According to Smith, since no rational agent would agree to an economic transaction that was not in his/her interest, the inevitable consequence of fully-informed market exchange is the betterment of all parties involved. In this light, the free market is seen to act as an 'invisible hand,' increasing overall welfare without recourse to coercive state intervention.

In concert, these theories provided the philosophical foundations and moral legitimacy for a mechanistic economics governed solely by the dictum of self-interest as expressed in the marketplace. Here, society was reduced to the sum of its parts, and societal good equated with the individual pursuit of unlimited material prosperity in a world of limitless resources (Costanza *et al.* 1997).

2.2 The Neoclassical Economic Model

From these premises, neoclassical economics has evolved and been formalized into an internally consistent set of theory and tools for the interpretation and structuring of economic activity. Neoclassical economics begins with a specific conception of the economic actor. Termed *Homo economicus*, this actor is viewed as a rational, self-interested, utility-maximizing agent whose income-constrained market preferences, subject to a number of restrictive assumptions, cumulatively discipline an efficient marketplace (Persky 1995; Becker 2006). Here, efficiency is defined in terms of Pareto-optimality – a state where resources are allocated such that no change in allocation will make one agent better off without worsening the condition of another agent (Sen 1993; Barr 2004).

In neoclassical economics, the economy is understood as a circular flow linking production (firms) and consumption (household) units through the forces of supply and demand (Daly and Farley 2004). On the supply side, economic inputs are referred to as ‘factors of production.’ These factors, which traditionally included land, labour and capital (although land has since been excluded from consideration), are deemed to be ultimately substitutable in the production of economic goods and services. When scarcities arise prices rise in complement, spurring technological innovation and the development of substitutes (Sloman 1999; Mankiw 2006). On the demand side, consumer willingness-to-pay expresses preference, and determines the optimal allocation of resources to productive ends. This model thus evinces the fundamental relationship between production and consumption, whereby patterns of supply-and-demand emerge from the respective goals of firms and households for profit and utility maximization (Daly and Farley 2004). It further predicts interactions of supply and demand under variable conditions, and how these determine prices as well as influence changes in the allocation of factors to producing different goods and services (Sloman 1999; Daly and Farley 2004; Mankiw 2006).

Since it contains all factors of production, the circular flow model effectively views the economy as whole unto itself. Moreover, because the factors of production are assumed to be freely substitutable, and the boundaries of the economy are not considered to be limited by any encompassing system, it is assumed that economic growth potential is similarly unconstrained (Daly and Farley 2004). This assumption is compatible with, and indeed necessary to, the pursuit of unlimited material prosperity as the organizing principle of social progress. At the same time, the prima facie acceptance of private property rights means that initial endowments are not considered relevant to the neoclassical calculus of economic efficiency. To the extent that inequalities are recognized, it is assumed that continued growth (with its attendant spill-over effects) rather than redistribution is the logical recourse. For this reason, the focus of neoclassical economics is largely on microeconomic phenomena. Neoclassical macroeconomics are principally concerned with ensuring smoothly functioning, free markets in facilitation of sustained growth (Daly and Farley 2004).

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CHAPTER 3: ECOLOGICAL ECONOMICS: THE PARADIGM OF COMMUNITY AND CONSTRAINTS

"The law that entropy increases – the Second Law of Thermodynamics – holds, I think, the supreme position among the laws of Nature. If someone points out to you that your pet theory of the Universe is in disagreement with Maxwell's equations – then so much the worse for Maxwell's equations. If it is found to be contradicted by observation – well, these experimentalists do bungle things sometimes. But if your theory is found to be against the Second Law of Thermodynamics I can give you no hope; there is nothing for it but to collapse in deepest humiliation."

- Sir Arthur Eddington (1933)

"A theory is the more impressive the greater the simplicity of its premises is, the more different kinds of things it relates, and the more extended is its area of applicability. Therefore the deep impression which classical thermodynamics made upon me. It is the only physical theory of universal content which I am convinced that, within the framework of the applicability of its basic concepts, will never be overthrown."

- Albert Einstein (1970)

3.1 The Ecological Economics Of Sustainable Scale

Ecological economics shares with neoclassical economics a preoccupation with the allocation of scarce means among competing ends. It further accepts the desirability of using means efficiently to attain those ends (Daly and Farley 2004). However, the answers provided by ecological economics with respect to defining scarcities, ranking needs, interpreting efficiency and allocating means to ends are strikingly different.

Ecological economics begins by challenging the notion that ever-greater material consumption provided by on-going economic growth is the overarching, desirable end for society (Daly and Farley 2004). This challenge is two-part. The first recognizes that the economy is a subsystem of the larger but finite global ecosystem that both sustains and constrains it. Following from this observation is a series of critiques regarding positive dimensions of the neoclassical economic model, including the assumptions that the market can efficiently allocate all types of ecosystem goods and services, and that technology will always find a substitute for scarce resources. These critiques are well established in the ecological economic literature (Ayres 1989; Cleveland 1991; Costanza 1991; Ayres 1993; Goodland and Daly 1996; Raskin *et al.* 1998; Rees 1999; Daly and Farley 2004; Ayres 2007a), and will be briefly summarized below.

The second challenges the normative assumptions of neoclassical economics. Specifically, this second strand questions the model of the economic actor espoused by neoclassical economics, and its implications for just and efficient resource allocation. In the words of Costanza *et al.* (1997; pp. 24) "Among the multiplicative factors affecting environmental degradation, the role of materialism and its relation to moral behavior is rarely discussed and is in need of broader, more serious scientific and public discourse." Ecological economics rejects the neoclassical assumption that growth is a panacea for inequality, and that self-interest is the principal defining motive of the economic actor (Daly and Farley 2004). Certain aspects of this critique have been vigorously debated in the ecological economic literature, whereas others have received notably little attention. More broadly, however, it appears that a core set of shared principles from which the ecological economic worldview might derive its legitimacy has not yet been adopted in the field. For this reason, the normative dimensions of ecological economics are treated in depth in a subsequent stand-alone chapter.

In neoclassical economics, the macroeconomy is thought to be whole onto itself. Non-human nature, to the extent that it is considered as resource inputs and as a repository for wastes, is seen as part of the whole. Accordingly, the neoclassical economic model does not recognize bounds on the growth potential of the macroeconomy (Daly and Farley 2004). In contrast, ecological economics sees the economy as a subsystem of the encompassing and supporting biosphere, which provides the material and energy resources that underpin *all* economic activity, absorbs the *inevitable* entropic waste produced, and creates the conditions in which human life is viable. For this reason, sustainable scale is a central preoccupation of the ecological economic approach (Daly 1991; Costanza *et al.* 1997; Daly and Farley 2004).

The ecological economic emphasis on appropriate scale is rooted in a recognition of the limitations implied by the Laws of Thermodynamics for the growth potential of the human economy (Daly 1991; Ayres 1998; Baumgartner *et al.* 2006). The First Law states that energy can neither be created nor destroyed, but only changed from one form to another. The Second Law states that every such transformation results in the degradation of usable forms of energy into lower quality, higher entropy forms (i.e. waste). In this light, every economic activity inevitably requires inputs of material/energy and results in the production of waste. Because the economy is an open system (characterized by material and energy throughput) embedded in a finite and relatively closed system (a materially closed planet with a discrete flux of solar/radiant energy), the scale of the human economy is ultimately constrained by the material and energy resource provisioning and entropic waste assimilatory capacity of host ecosystems and of the biosphere as a whole (Daly and Farley 2004).

This relationship has been expressed formally in the theory of dissipative structures advanced by Ilya Prigogine (Prigogine 1961; Prigogine and Nicolis 1977; Prigogine and Stengers 1984) and subsequently applied to economy/environment interactions (Georgescu-Roegen 1971; Binswanger 1993; Rees 1995; Baumgartner *et al.* 2006). An open system can only be maintained and evolve in a state far from thermodynamic equilibrium through the irreversible dissipation of matter and energy, which is taken from the surrounding environment and returned to the environment as a net increase in high-entropy waste. As the economy grows, it inevitably appropriates ever greater shares of the biosphere, both in terms of energy and material inputs and waste assimilatory capacity. What is encompassed represents the opportunity cost of economic growth (Daly and Farley 2004). Since the finite biosphere is itself a self-organizing system that constitutes the life-sustaining context for all human and non-human nature, continuous economic growth is neither physically feasible nor socially desirable (Daly 1991). Rather, what emerges from this perspective is the importance of identifying optimal scale for economic activities, where the marginal costs of growth do not exceed the marginal benefits obtained (Daly and Farley 2004).

Optimal scale is a central concept in neoclassical microeconomics. However, because the neoclassical paradigm does not recognize constraints to the growth of the economy as a whole, there can be no perceived opportunity costs and hence no concept of optimal scale from the neoclassical macroeconomic perspective (Daly and Farley 2004). Ecological economics therefore takes as its departure point the centrality of optimal scale to sustainability. Moreover, ecological economists believe that the scale of the economy has already reached the point where it is large relative to the size of the biosphere (Daly 1991; Daly and Farley 2004). In contrast to an 'empty world' vision, where the opportunity costs of economic growth are trivial, this 'full world' vision recognizes that continued economic growth occurs at substantial opportunity cost with respect to global ecological integrity and the welfare of both current and future human generations (Farina *et al.* 2003; Daly and Farley 2004). The implications of the sustainable scale criterion for human organization are profound. In contrast to the contemporary conception of sustainability as a juggling act between equally important socio-economic and environmental considerations, the paramountcy of appropriate scale repositions the latter as prerequisite to the former.

In the words of McNeill (2000; pp. 337) "the overarching priority of economic growth was easily the most important idea of the twentieth century." From the ecological economic perspective, constraining growth relative to biocapacity must necessarily be the priority of the twenty-first. What is immediately apparent, then, is the necessity of re-examining the assumptions of the neoclassical model with respect to how it proposes to resolve sustainability concerns.

3.2 Ecological Economic Critiques Of The Neoclassical Model

The primary (technical) remedy to environmental externalities and environmental scarcity advanced by neoclassicists is the extension of the market mechanism to encompass ecosystem goods and services, which have previously been external to the calculus of economic costs and benefits. In recent decades, the sub-discipline of environmental economics has emerged to serve in this capacity. Inclusion of ecosystem goods and services in the market requires the development of property rights and prices for these resources such that they might be efficiently allocated, or the estimation of 'shadow prices' where markets do not exist in order to include environmental criteria in cost-benefit analyses (Vatn and Bromley 1994).

Neoclassical economists generally recognize a suite of conditions that must be met in order for the price/market mechanism to function efficiently in resource allocation. These assumptions are that (1) the factors of production are perfectly substitutable, (2) external costs and benefits are negligible, (3) all goods are market goods (rival and excludable), (4) market participants have access to perfect information, (5) all markets are fully competitive, and (6) transaction costs are negligible (Daly and Farley 2004).

Given the restrictive nature of these assumptions, it is to be anticipated that most markets are less than perfectly efficient. Market failure occurs when inefficiencies are substantial due to an inability to meet some of all of the requisite conditions for market efficiency. Each of these assumptions has been scrutinized and debated at length in the ecological economic literature with respect to marketing ecosystem goods and services. The practice of shadow pricing via preference elicitation and other techniques has been similarly scrutinized. The following section briefly summarizes these critiques, and concludes with an evaluation of the potential efficacy of the market mechanism or non-market environmental valuation in efficiently allocating ecological resources and hence ensuring their sustainability.

3.2.1 Substitutability

In ecological economics, the term 'capital' refers to stocks that give rise to flows of goods or services (Daly and Farley 2004). The debate regarding the substitutability of capital inputs in economic activity points to one of the most important schisms in the discursive struggle to define sustainability between the neoclassical and ecological economic camps. These respective positions have become known as 'weak' and 'strong' sustainability, which differ according to the degree to which each believes that natural capital (the range of ecosystem goods and services provided by the natural environment) can be substituted with other kinds of capital (human, social and manufactured) whilst sustaining human welfare in perpetuity (Ekins *et al.* 2003; Krysiak 2006; Ayres 2007b).

As previously described, neoclassicists see economic activity as sustainable so long as aggregate per capita capital stocks do not decline over time (Ekins *et al.* 2003). Taken to its logical extreme, such technological optimism suggests the viability of replacing all ecosystem goods and services with manufactured equivalents. Indeed, some proponents have advanced claims to the effect that environmental variables in no way limit the growth potential of the human economy (Myers and Simon 1994).

Supporters of the strong sustainability position reject the assumption of universal capital substitutability based on an evaluation of the nature and functions of natural capital. Specifically, natural capital is thought to serve four major functions: material and energy provision; waste assimilation; amenity values; and basic life support functions (Ekins 1993). Ecological economics recognizes that the first three functions are, to limited degrees, substitutable in the short term. In keeping with the Laws of Thermodynamics, however, every economic activity requires inputs of material and energy and produces waste. In other words, neither human nor manufactured capital can, in the long run substitute for natural capital because they are themselves derivative of natural capital. For this reason, the factors of production must ultimately be seen as complements rather than substitutes. Moreover, it is maintained that the life support functions provided by global biogeochemical cycles (called critical natural capital) are not substitutable. This assertion derives from an appreciation of the entropic implications of replicating the resource provisioning and waste assimilatory services of biogeochemical cycles, and of the prevalent environmental characteristics of irreversibility and uncertainty. Ecological economics therefore demands the maintenance of 'critical' natural capital stocks far above potential carrying capacity thresholds (Deutsch *et al.* 2003; Ekins 2003; Ekins *et al.* 2003).

3.2.2 Externalities

Externalities are harms or benefits generated by a market transaction for which the recipients are not compensated and for which the beneficiaries do not pay (Templet 2000). Market prices, which balance supply and demand, typically reflect human preferences and other influences, but rarely do they capture the environmental costs and benefits associated with their provision. These hidden costs (negative externalities), which benefit specific economic actors at the expense of society as a whole, help explain the seeming contradiction between apparently healthy, growing economies and the accompanying environmental degradation. However, rather than exceptions, the thermodynamic perspective reveals that externalities associated with increased entropy are an inevitable consequence of economic activity (Faber *et al.* 1998). As the factors of production (energy/matter, labour and capital) are brought to bear in the delivery of a desired economic good or service, the transformations of these factors produce both the desired product/service along with undesired resource depletion and waste products that, in sufficient quantity, result in environmental degradation (for a formal proof, see Krysiak 2006). The challenges to meaningfully accounting for these environmental variables via the traditional market mechanism mean that externalities are pervasive, such that the neoclassical economic requirement for negligible externalities in efficient markets can rarely be met.

3.2.3 Ecosystem Goods And Services As Market Goods

Neoclassical economics treats all economic inputs in a homogeneous fashion by assuming that all are marketable and substitutable. In contrast, ecological economics distinguishes between human, manufactured, and natural capital and the unique features of the latter that render many ecosystem goods and services non-marketable and non-substitutable (Gustafsson 1998). A key distinction is made between stock-flow and fund-service resources (Daly and Farley 2004). Stock-flow resources, which may be renewable or non-renewable, provide the material basis of production. They exist in finite quantities that may be stockpiled, and they are transformed through the production process (Daly and Farley 2004). Fund-service resources provide a flow of service per unit time that cannot be stockpiled. This distinction is very important to understanding how ecosystem goods and services do or do not meet the conditions of excludability and rivalness characteristic of marketable goods (Daly and Farley 2004).

Excludability refers to the potential for an owner of a good or service to prevent others from using it. Generally, some kind of institution such as a property right is required. Hence, to be excludable, it must be possible to assign user rights (Daly and Farley 2004). Rivalness is a characteristic of a good or service whereby its use by one person reduces the amount available for use by others. Rivalness is hence essential to scarcity, which is a determining factor in the value of market goods. Ecological economists further distinguish between the specific kinds of abiotic and biotic resources that possess these qualities, and hence may be considered marketable goods, and those that do not (Daly and Farley 2004).

Abiotic resources include mineral resources, fossil fuels, water, land and solar energy. The first two of these are stock-flow resources that are both rival and excludable within generations. Minerals, because they are partially recyclable, are only partially rival between generations, whereas fossil fuels are fully rival. It is likely that fossil fuels are substitutable, but not so for mineral resources. The latter two are fund-service resources. Of these, land is rival and excludable within generations but solar energy is not. Neither is rival between generations. Both are ultimately non-substitutable. Water is a special case, since its characterization as a stock-flow or fund-service resource, as well as whether it can be made excludable and rival is context-dependent. For most purposes, it is ultimately nonsubstitutable (Daly and Farley 2004).

Biotic resources include renewable resources, ecosystem services and waste absorption capacity. Renewable resources are stock-flow resources that are both rival and excludable within generations. Depending on their rate of use, they may also be rival between generations. Although they are substitutable at the margin, they are ultimately non-substitutable (Daly and Farley 2004). Both ecosystem services and waste absorption capacity are fund-service resources. Most ecosystem services are neither rival nor excludable within and between generations. Although they may have low substitutability at the margin, they are largely non-substitutable. In contrast, waste absorption capacity is excludable and rival within generations, and may be rival between generations depending on rate of use. These resources are moderately substitutable at the margin, but ultimately non-substitutable (Daly and Farley 2004).

These unique attributes of various ecosystem goods and services implies that most are not market goods. Rather, many are pure public goods. For this reason, they cannot be efficiently allocated via the market mechanism and will hence be chronically undervalued and undersupplied within the existing neoclassical economic market system. This also presents challenges to monetary valuation of ecosystem goods and services, which requires that we implicitly treat them as commodities, or marketable goods.

3.2.4 Perfect Information

In neoclassical economics, the market can function efficiently only when market agents have access to perfect information (necessary to fully-informed exchange). While restrictive even for traditional market goods, this requirement is more problematic still when dealing with ecological inputs to economic activity, most of which have historically been largely ignored in the calculus of market exchange.

Price represents a hybrid proxy for exchange value which captures only those values that the market can effectively communicate. Beyond the previously discussed factors, attempts to derive monetary values for ecosystem goods and services are inevitably frustrated by their inherent complexity and multiple attributes (Vatn and Bromley 1994). Valuation necessitates treating ecosystem functions as isolated entities. This is in clear conflict with our understanding of natural systems as tightly interconnected processes in which the value of a component can only be understood in terms of the relationships in which it participates. Furthermore, even if we were to assume perfect information, the act of compressing this complexity into a simple monetary metric results in the loss of important information, thus stripping the metric of its legitimacy (Vatn and Bromley, 1994).

The valuation challenge is multiplied by the functional transparency of many ecosystem services, which means that their contribution is unknowable until they cease to function (Vatn and Bromley, 1994). Systems theory predicts that, when subjected to stress, a dynamic system will oscillate within prescribed boundaries until, upon reaching a critical limit, it will undergo sudden and irreversible change (Capra 1997). In other words, the impairment of ecosystem function cannot be expected to occur in a linear and predictable manner. Assigning monetary values to nature will in no way provide safe measures of ecological scarcity because the marginal value of a given service will not predict its breaking point (Rees 1995). For these reasons, the assumption of consumer access to perfect information that fully reflects the ecological costs and benefits associated with specific market transactions is unrealistic.

3.2.5 Perfect Competition

Perfect competition occurs when neither buyers nor sellers have disproportionate market influence. Among other things, fully competitive markets require numerous independent firms and households with the respective willingness and ability to supply and consume products at specific prices, low market entry and exit barriers, and that both consumers and producers have access to perfect information (McNulty 1967; Novshek and Sonnenschein 1987). As previously described, the latter perfect information criterion is rarely feasible. Moreover, the nature of various ecosystem goods and services is also frequently incompatible with the former criteria. This is because many ecosystem goods and services are neither rival nor excludable; hence they can cannot be privatized and sold in the market. These 'pure public goods,' such as fresh air or the protection afforded by an intact ozone layer, may be used by anyone regardless of who pays for it. Since competitive markets cannot be established for these goods/services, the market will systematically underprovide them (Daly and Farley 2004). In addition, even where specific resources are rival and excludable, it may not be feasible to have numerous firms responsible for their provision. For example, in the case of fresh water or energy distribution, the considerable infrastructure required means that monopolies are often preferable (Daly and Farley 2004). Finally, access to ecosystem goods and services is an intergenerational concern. Since future generations cannot participate in today's markets, fully competitive markets cannot be established. Ecological economics therefore recognizes the various constraints to perfect competition for ecosystem goods and services, and the attendant limitations to market allocative efficiency (Daly and Farley 2004).

3.2.6 Negligible Transaction Costs

A transaction cost is the cost of coming to an agreement in a market exchange (Daly and Farley 2004). The potential for negligible transactions costs with respect to internalizing the ecological dimensions of economic activities in market prices is severely hampered on several counts. First, since the negative externalities may be experienced by numerous actors (for example, in the case of climate change), negotiations between all relevant agents is challenging. Moreover, their inevitable generation (due to the entropy law) means this would be required for all market transactions. Second, since externalities may also affect future generations, transaction costs between generations are infinite (Daly and Farley 2004). Finally, lack of perfect information for the reasons previously discussed precludes realistic negotiations. For these reasons, ecological economics similarly rejects the feasibility of this condition for market efficiency.

In short, ecological economics holds that, in most cases, none of the conditions for efficient market allocation stipulated in the neoclassical model can be fulfilled for the majority of ecosystem goods and services. As a result, markets alone cannot reasonably be expected to serve environmental sustainability objectives.

Despite these criticisms, valuation of ecosystem goods and services is an increasingly common practice (Vatn and Bromley 1994; Norton and Noonan 2007). Various techniques have emerged based on implicit pricing methods, which estimate non-market values using surrogate markets or other means (Smith 2000). Contingent valuation methods are survey-based techniques for the valuation of non-market resources that focus on revealed preferences by asking how much people would be willing to pay/be compensated for the existence/loss of some environmental feature (Hoehn and Randall 1987; Baral *et al.* 2008). Hedonic pricing is used to estimate the economic values of ecosystem goods/services that directly affect market prices (Hamilton 2007; Michael 2007; Poor *et al.* 2007). However, a series of theoretical and methodological challenges renders their utility questionable (Sagoff 1981; Vatn and Bromley 1994; Diamond and Hausman 1994; Sagoff 1998; Nunes and van den Bergh 2001). Although pricing of certain ecosystem goods and services is sometimes feasible and may be of significant value in specific contexts, ecological economics recognizes the limited utility of this approach to managing for sustainability objectives.

3.3 Revisiting The Circular Flow Model Of The Economy

As previously described, neoclassical economics imagines the economy as a circular flow linking production (firms) and consumption (household) units through the forces of supply and demand. Although useful, the circular flow model of the economy is incomplete because it is abstracted from the encompassing ecosystem whose resource provisioning and waste assimilation services provide the metabolic foundation which underpins the flow of exchange value (Costanza *et al.* 1997; Daly and Farley 2004). A central focus of ecological economics is therefore to elucidate the relationships between specific kinds and quantities of economic activity, and the capacity of the biosphere to provide the material and energy inputs that underpin all economic goods and services, as well as assimilate the inevitable entropic waste produced. The model of the economy is thus expanded to include the encompassing biosphere, which provides the necessary material/energy and waste metabolic functions.

3.4 Summary: Ecological Economics As Macroeconomics

Microeconomics focuses on market efficiency. Macroeconomics considers the economy as a whole. The overarching objective of neoclassical macroeconomics is market-driven economic growth, which may be facilitated via fiscal and monetary policy interventions. In ecological economics, the sustainability of the encompassing ecosystem is the primary macroeconomic objective. In this light, optimal scale replaces continuous growth as the organizing imperative (Daly and Farley 2004). Within this context, ecological economics prescribes two additional macroeconomic objectives. The first is just distribution. Due to the inadequacy of markets in allocating most ecosystem goods and services, the second is the establishment and implementation of non-market allocative principles and procedures as a matter of macroeconomic policy (Daly and Farley 2004).

In neoclassical economics, scarcity of means is a function of existing technologies and the mobilization of capital. Since all forms of capital are considered substitutable and the macroeconomy is viewed as whole onto itself, scarcities are relative rather than absolute. In contrast, ecological economics sees low-entropy matter-energy as the basic common denominator by which means must be measured in an ecological economy. In this light, the ultimate mean is provided by the gradient between low and high-entropy matter-energy that facilitates all economic transformations (Daly and Farley 2004). It is precisely because sources of low-entropy matter-energy in the biosphere are finite, as is the assimilatory capacity of sinks for high entropy wastes, that ecological economics recognizes the scarcity of means as a defining principle.

Neoclassical economics views the market as a sufficient mechanism for the allocation of means. In other words, all goods are market goods. Ecological economics agrees that the market provides a useful institution for allocating many means in the service of ends, but argues that many goods (in particular, many ecosystem goods and services) are not amenable to market allocation (Daly and Farley 2004). Allocation in the service of ends hence requires regulative intervention structured in the interest of sustainability objectives. Moreover, whereas neoclassical economics defines efficiency strictly in terms of the allocation of resources to least-cost productive means in the service of maximizing individual consumptive utility, ecological economics expands the meaning of efficiency to include resource use and waste considerations.

Defining and ranking ends requires determining and prioritizing that which we value. In neoclassical economics, ends are defined and ranked according to the principle of utility maximization, where value is expressed via individual consumer preference in the marketplace. While ecological economics rejects this narrow delineation of value for a variety of reasons, the problem of ranking ends has not been resolved in a comparable manner. Identification of an ultimate end, however, has coalesced around the ecological economic conception of sustainability, whereby the life-supporting capacity of the biosphere is ensured by constraining the scale of the human economy at a steady state well within the buffering capacity of planetary biogeochemical cycles (Daly 1991). The challenge of linking intermediate ends with this ultimate end in the ecological economy, as well as the philosophical premises from which the sustainability norm might derive its legitimacy, will be explored at length in the next chapter.

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CHAPTER 4: ENVIRONMENTAL SUSTAINABILITY AS THE FIRST PRINCIPLE OF DISTRIBUTIVE JUSTICE: TOWARDS AN ECOLOGICAL COMMUNITARIAN NORMATIVE FOUNATION FOR ECOLOGICAL ECONOMICS.

4.1 Publication Information

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4.2 Abstract

The ecological economic concern with environmental sustainability embodies the normative orientations of the field. This concern is foremost a matter of distributive justice, the definition of which determines the relevance of the appropriate scale and efficient allocation criteria. Yet it would appear that the discipline lacks a shared, internally consistent set of ethical premises by which this concern might be legitimized. Various authors have embraced a Rawlsian conception of liberal justice as the appropriate banner for ecological economics in place of the consequentialist-libertarian/neoliberal foundations of neoclassical economics (including environmental economics). It is argued here that this is insufficient in so far as it is premised on a vision of a discrete, self-sufficient economic actor. Instead, it is posited that an ecological economic ethic must proceed from an understanding of the economic actor as community member – a recognition implicit in recent ecological economic contributions focused on discourse ethics and deliberative democracy. An ecological communitarian conception of distributive justice, which views the well-being of the individual as inseparable from the integrity of its implicate, mutually constituting human and non-human natural communities, is advanced as the appropriate basis for the ecological economic world-view. In this light, the thermodynamic foundations of ecological economics are seen to provide the necessary departure point for normative decision-making oriented towards ensuring environmental sustainability in economic organization.

4.3 Introduction

Economic systems embody the rules by which scarce resources are allocated across relevant populations. Allocation presupposes the resolution of distributive norms that both define the constellation of economic actors as well as how and to whom the costs and benefits of economic activities accrue. These norms must thus derive from specific conceptions of distributive justice in material relations. Given that the meaning and application of just distribution must, in turn, be founded on shared interpretations of the right or good, an economic system operationalizes a core component of a society's moral universe.

The neoclassical economic model has been roundly criticized for (among other things): its analytical myopia with respect to the foundational role of ecosystem goods and services in economic activity; the attendant environmental degradation resulting from the omission of this consideration in economic policy; and the implications for both economic and environmental sustainability within and across generations. In large part, these failures reflect a fundamental incompatibility between the underlying ethical premises that inform neoclassical economic conceptions of distributive justice, incommensurability in the spectrum of values which must be weighed, and the operational realities of economic activity in a finite biosphere. The discipline of ecological economics arose precisely to address this deficiency in contemporary economic theory and practice. The overarching goal of ecological economics is to advance a body of theory and tools conducive to operationalizing an environmentally sustainable mode of economic organization (Soderbaum 1999).

Of the three conditions ecological economists prescribe as necessary for sustainable economic activity (appropriate scale, efficient allocation and just distribution) (Daly 1992), the just distribution clause embodies the discipline's normative orientations. This is implicit in the sustainability objective, which is concerned with maintaining the integrity of the ecological systems foundational to the well-being of both current and future generations. Moreover, a shared understanding of distributive justice is prerequisite to establishing the related criteria for efficient allocation, as well as for legitimizing the relevance of both the scale and efficiency variables (Martinez-Alier and O'Connor 1996).

Given the central importance of just distribution as the normative foundation for ecological economics, the topic has received remarkably little treatment in the disciplinary literature. In the absence of a clear, shared understanding of distributive justice and how it animates the balance of ecological economic theory and practice, disciplinary cohesion, evolution and operationalization are impeded. This paper presents an attempt to address this lacuna. I begin with a brief overview of the major conceptions of distributive justice, each of which is moored in a specific interpretation of the (deontological) right or (consequentialist) good. I next review the normative dimensions of neoclassical economics, and compare and contrast these with the limited treatment of distributive justice in the ecological economics literature. I suggest that both are undermined by a fundamental misunderstanding of the nature of the individual. Turning to the core insights of ecology as a basis for the necessary reconception of the economic actor as ecological community member, I posit that an expanded ecological communitarianism, which views maintaining the integrity of the web of biotic and abiotic processes and communities that mutually constitute the biosphere as the first principle of distributive justice, offers the most appropriate departure point for ecological economics as a normative enterprise. In this light, the thermodynamic foundations of ecological economics are shown to provide the necessary reference point for normative decision making in economic organization.

4.4 Discussion

4.4.1 Distributive Justice

Justice, in the normative (as opposed to procedural) sense, is the principle of moral rightness or goodness. Views of what constitute justice are therefore highly variable because they are moored in competing conceptions of the (deontological) right or (consequentialist) good, which is prior, and how it is achieved. In most cases, these can be traced to the major traditions of normative moral philosophy.

Following Lamont and Favor (2007), principles of distributive justice are “normative principles designed to guide the allocation of the benefits and burdens of economic activity.” The western tradition of normative political philosophy comprises several major interpretations of distributive justice along a continuum encompassing libertarian, socialist and liberal theories. Also notable are emerging theories of environmental justice (Sterba 2003; Okereke 2006).

According to libertarianism, individual liberty (i.e. the state of being unconstrained by other persons in pursuing one’s self-interest) is the ultimate moral and political ideal (Sterba 2003). This typically entails an ideological commitment to private property rights within (although not necessarily between) generations, with distributive justice as that which provides freedom to individuals to pursue their respective material desires. In this light, the free market is viewed as inherently just because it expresses the sum of selfish desires. Any redistributive interventions by non-market forces are seen as violations of the individual’s basic right to liberty (Hayek 1976). These consequentialist libertarian premises are central to the free-market principles of Chicago School-style neoclassical economics (Friedman 1953, 1962), and bear similarities to neoliberal orientations.

The socialist conception of justice views equality rather than liberty as the moral imperative. Following Marx, just distribution conforms to the principle ‘from each according to his ability, to each according to his need.’ Socialist justice therefore demands egalitarian access to resources (Sterba 2003), and is distinctly deontological in flavour.

Justice in the classical liberal democratic tradition attempts to balance the ideals of liberty and equality. Liberal distributive justice has been approached under various guises ranging from free market neoliberalism to welfare liberalism. The most popular contemporary construction of contractual liberal justice is that advanced in Rawls’ seminal *Justice as Fairness* (1971). Rawls argues that just distributive principles for society must be acceptable to free and rational persons under initial conditions of equality, which is ensured when we adopt a 'veil of ignorance' with respect to our initial conditions in deciding on fair agreements (Rawls 1971, 1999). This conception of moral equality in the right to resources is often evoked in support of the Brundtland version of sustainable development, which expressly prioritizes the equitable satisfaction of human needs within and across generations (Brown 2000; Langhelle 2000; Wenar 2001).

Theories of environmental ethics are inherently theories of distributive justice because they largely center on delineations of moral considerability and the definition of moral communities across species boundaries. How environmental ethicists have variously championed specific delineations may be usefully characterized into four categories along two key demarcations. The first demarcation separates anthropocentric from non-anthropocentric orientations (along with the attendant debate regarding the attribution of intrinsic value to nature). The second delineates individualistic from holistic theories. Throughout the history of Western philosophical thought, the focus of ethical concern has largely been interhuman behavioural norms. The animal rights and welfare positions simply extend distributive concerns from humanistic ethics to individual animals based on arguments of utility or agency (Singer 1975; Regan 1985). Biocentric ethics extend distributive concerns based on the criteria of being alive, with the focus remaining the individual (Taylor 1981). Holism has been championed on both anthropocentric (Norton 2005) and non-anthropocentric grounds (Callicott 1984, 87, 89, 99; Naess 1973, 1989; Westra 1994). Here, distributive justice requires allocative strategies which serve to maintain the integrity of host ecosystems and, by association, the well-being of those entities dependent upon them.

4.4.2 Distributive Justice In Neoclassical Economics

Economic systems serve two primary purposes – the allocation of resources amongst competing uses, and the distribution of costs and benefits among market participants. In neoclassical economics the former is the domain of microeconomics, which emphasizes the development of market policies to maximize allocative efficiency. The latter is the domain of macroeconomics, and requires political decision making reflective of a society's distributive ideals. Since the positivist revolution in economics, it has been assumed that microeconomic questions can and should be neatly divorced from broader, macroeconomic distributive objectives (Lipsey 1983, Routh 1989, Arrow *et al.* 2004). However, an examination of the assumptions underpinning the neoclassical conception of allocative efficiency reveal that efficiency cannot be defined in the absence of pre-existing distributive ideals (van de Veer and Pierce 1997). Although frequently portrayed as value neutral, even a superficial consideration of the neoclassical treatment of efficiency points to the numerous normative assumptions upon which it depends for its legitimacy. Thus distributive issues are considered irrelevant in neoclassical microeconomics not because they have no bearing but because they have already been, a priori, resolved. Specifically, neoclassical economics assumes: that initial endowments are irrelevant to the calculus of welfare; that the aggregate utility (a concept derived from utilitarianism, which occupies a minor position in humanistic ethics) of rational, self-interested utility maximizers is a defensible metric by which it is assessed; that individual preference communicated through the market mechanism constitutes an ethically sound basis for social organization; and that the calculus of well-being need only consider human agents (van de Veer and Pierce 1997). It is for this reason that Keita (1997) argues, in defiance of positivism, that neoclassical economics should be viewed as a branch of applied ethics.

If any of these assumptions are rejected, then the legitimacy of the model itself is called into question. It would appear that each has, in fact, been richly explored and contested by numerous authors (for example, see van de Veer and Pierce 1997; Sagoff 1981, 1998, 2004; Vatn and Bromley 1994; Norton and Noonan 2007), whose detailed arguments need not be repeated here, but from whose work it is apparent that the value assumptions underpinning the neoclassical model are morally bankrupt. It is therefore constructive to entertain what might be more appropriate ethical premises for economic organization.

4.4.3 Distributive Justice In Ecological Economics

Ecological economics is concerned with understanding the environmental dimensions of economic activity *in order to advance the objective of environmentally sustainable economic organization* (Soderbaum 1999; Daly and Farley 2004). By environmental sustainability is meant the maintenance of the integrity of the biophysical environment, including its resource provisioning and waste assimilatory capacities, in perpetuity. Since this concern with sustainability presupposes a specific vision of how the world ought to be, ecological economics is an inherently normative enterprise. Moreover, because the concern with sustainability is, by nature, a concern with the perpetuation of a set of conditions relevant to the interests of some constellation of intended beneficiaries, it must be informed by a specific conception of distributive justice (Luks and Stewen 1999; White 2000). What is less clear is the set of ethical arguments by which the ecological economic concern with environmental sustainability is justified. Also unclear is the delineation of the community of actors to whom the attendant conception of distributive justice applies. Although various authors have contributed partial and often contradictory accounts, no single clearly articulated, unifying vision has emerged. Given the importance of distributive justice to the ecological economic worldview, this lack of clarity represents a significant theoretical gap.

The first necessary distinction in assessing the treatment of distributive justice in ecological economics is the anthropocentric versus non-anthropocentric divide (Van den Bergh 1997). On the one hand, there is the anthropocentric concern with intra- and intergenerational distributive justice. Here, environmental integrity is deemed important only to the extent that human welfare is impacted. In this sense, distributive concerns arise where changes in the allocation of resources over time and space may frustrate the potential needs satisfaction of human economic agents. The second is a non-anthropocentric concern for nature, generally. Although imperfect, this distinction provides a useful starting point for assessing the treatment of distributive justice in the ecological economics literature and the extent to which the arguments advanced differ from neoclassical economic orientations.

4.4.4 Sustainability As An Anthropocentric Concern In Ecological Economics

The implicit acceptance of anthropocentrism as the appropriate departure point for sustainability concerns is commonplace in the ecological economics literature. This is evident in the treatment of several central themes, including the appropriate conception of the economic actor, the practice of discounting, property rights issues, and the distribution of environmental impacts. It is unclear whether this represents a carry-over from neoclassical economics, an acceptance of enlightened self-interest as sufficient grounds to legitimize the field, or if it follows from deeper reflection on the nature of moral reasoning and moral agents.

The Economic Actor

A recurring theme in the ecological economics literature is criticism of the inadequate portrayal of the economic actor in the neoclassical model (Janssen and Jager 2000). In general, since *Homo economicus* is strictly self-referential, this conception does not provide space for a systematic concern about others (Becker 2006). Moreover, the ideal of unlimited needs satisfaction, which is taken as the *modus operandi* of *Homo economicus,* is directly at odds with the ecological economic imperative to restrict present consumption patterns in the interests of future generations. Further problems with the neoclassical conception include the assumption that everything of value to the economic actor is commensurable and can be expressed via the price numeraire, and that preferences are fixed.

The importance of broader relational considerations in economic decision making has been suggested with reference to a number of philosophical traditions and supported by a range of empirical studies (see Janssen and Jager, 2000; Gintis, 2000; and Siebenhuner, 2000 for reviews of the pertinent literature). This distinction has focused largely on interhuman relations, on which basis several authors have argued for a reconception of the economic actor. Siebenhauer (2000) describes a *Homo sustinens* based on the human propensity for cooperation and communication as suggested by evolutionary biology and psychology. Gintis (2002) invokes a *Homo reciprocans* to explain empirical observations that humans are actually strong reciprocators. The *Homo politicus* of Faber *et al.* (2002) is characterized by its pursuit of political justice which, in the spirit of Aristotle, is deemed an inherent attribute of a rational being. Underlying these critiques is the idea that the economic actor belongs to a human community and that her economic decisions are made, at least in part, from the perspective of a community member. The implicit anthropocentrism is unquestioned.

Discounting

A second distributive theme that has attracted significant attention is the neoclassical practice of discounting. Numerous authors have taken exception with this practice, ostensibly on ethical grounds. Lumley (1997) argues that discounting reduces questions of what is best to questions of what is most profitable, and that this is inadequate where a complex range of non-commensurate values must be considered. It also privileges the present at the expense of the future, which directly contradicts the objective of intergenerational equity.

Azar and Sterner (1996) distinguish between two approaches to the discount rate: one based on the opportunity cost of capital (marginal rate of return on investment), and the other given by the social rate of time preference. It is argued that the former must account for the constraints to economic growth implied by the sustainable scale criterion. As the rate of economic growth slows towards a steady state, the discount rate must similarly tend towards zero. In this light, the future costs of environmental degradation become much higher and hence favor constraint in the present. With respect to the latter, it is argued that a rate of time preference is only relevant to individuals, since individuals have limited life spans, whereas society as a whole enjoys continuity. For this reason, (as argued by numerous economists and philosophers including Spash and d'Arge, 1989; Broome, 1992; Cline, 1992), there is no sound ethical justification for applying a social rate of time preference larger than zero. Again, these arguments center exclusively on the ethics of interhuman relations.

Property Rights

Also of note is the ecological economic concern with questions of property rights. Whereas neoclassical economics takes initial distributions as given, ecological economics stresses the importance of redistribution over growth as the necessary means of meeting the needs of the impoverished whilst respecting environmental sustainability objectives (Daly and Farley 2004). This requires a significant departure from the libertarian and neoliberal roots of neoclassical economics, which view private property as sacrosanct (Bromley 1989; Daly and Cobb 1989; Ostrom 1992; Hanna and Munasinghe 1995; Hanna *et al.* 1996). Treatment of this issue in the ecological economics literature has largely been restricted to interhuman property rights and distributive concerns.

The Distribution Of Environmental Impacts

A fourth common theme with distributive implications relates to the intra- and intergenerational environmental impacts associated with economic growth, particularly with reference to global climate change. Azar and Sterner (1996) argue that resolving how best to approach the problem of climate change has important intergenerational distributive implications. Paavlova and Adger (2006) point out that anthropogenic climate change is caused mainly by greenhouse gases emitted by developed countries, but that climate change impacts will disproportionately burden developing countries. Similarly, a significant body of literature discusses the outsourcing of the environmental impacts of western-style consumption with reference to the Environmental Kuznets Curve (Stern *et al.* 1996; Rothman 1998; Suri and Chapman 1998; Torras and Boyce 1998; Bagliani *et al.* 2008). Again, the focus is the distribution of environmental impacts between human actors.

Justice As Fairness In Ecological Economics

While all of the aforementioned authors either implicitly or explicitly recognize the ethical dimensions of their critiques, none have presented a sound philosophical grounding for their concerns or arguments which such an ethical foundation might demand. In general, however, it would appear that most derive from a liberal interpretation of justice as fairness, as characterized by the work of John Rawls (1971, 1999). This is evident in the apparent interchangeability of the concepts of 'just' and 'fair' in the literature. For example, Lumley (1997) argues that the inclusion of ethics in decision making (in place of cost-benefit analyses employing discounting as a proxy for intergenerational concerns) would facilitate achieving objectives of intra- and intergenerational equity. Paavlova and Adger (2006) similarly describe the distribution of climate change impacts as a question of fairness. Elsewhere, Daly (1992) describes a 'good' distribution as one that is 'just or fair' and Aubauer (2006) frames the just and efficient reduction of resource throughput as a matter of fairness between generations. Barrett (1996) and DeCanio and Niemen (2006) have also discussed intergenerational distribution in terms of equity and both Ferguson (2002) and Haberl *et al.* (2004) frame international fairness as equal per capita consumption of resources. However, only Norton (1989) advances a detailed application of Rawlsian liberal justice to determine optimal strategies for ensuring intergenerational equity in environmental decision making. Following Rawls' 'veil of ignorance' condition, Norton (1989; pp. 137) argues that a rational chooser ignorant of the generation he will inhabit would adopt a naturalist-preservationist strategy, which "imposes constraints on the pervasive exploitation of ecological systems and thereby protects biological diversity in the long run." Norton's article appeared in the first volume of Ecological Economics. It would appear that many subsequent authors have simply accepted the banner of distributive justice as fairness in place of the libertarian/neoliberal stance of neoclassical economics as the appropriate standard for the ecological economic worldview. Interestingly, Rawls did not advocate applying his model of justice as fairness beyond the nation state, to say nothing of intergenerational considerations, although several authors outside of the ecological economics literature have championed this position (for example, see Beitz 1999, 2000, 2001). Norton has also subsequently moved away from this position in favour of an ecologically enlightened anthropocentric communitarianism (Norton 2005).

4.4.5 Sustainability As An Ecocentric Concern In Ecological Economics

According to Becker (2006), the ecological economics literature has been decidedly quiet with respect to the potential ethical dimensions of human/nature relations beyond their immediate implications for human welfare. Although several authors have indicated the possibility and/or desirability of non-anthropocentric concern for nature as a normative orientation in the field (Costanza *et al.* 1996, van den Bergh 1997, Becker 2006) the advancement of a philosophical platform for such a position is notably absent. Some partial developments may be found in Becker's third of three relations of the human being. According to Becker (2006), the three basic relations that define the human being are the relation to oneself, the relation to community, and the relation to nature. The relation to nature is presented as distinct from self-interest or the pursuit of biological survival. Instead, it derives from respect for and sympathy towards nature, the alignment of one's creativity with that found in nature, and a personal, experience-based relation with nature. This *Homo ecologicus*, Becker argues, is not merely a biological organism concerned with survival but also a moral being concerned with the good life and human excellence. Referring to virtue ethics and romantic natural philosophy, Becker implies the existence of intrinsic value in nature, the recognition of which promotes human fulfillment.

The relative paucity of theorizing regarding the ethical dimensions of human/nature relations among ecological economists is somewhat surprising given that the field is defined by its preoccupation with environmental sustainability. Even if there was consensus that anthropocentrism was the appropriate departure point from which to consider the ethical implications of specific economic configurations, one would expect, in the least, an adequate defense for such a position and clear arguments for the rejection of competing orientations. However, it is by no means clear that ecological economists are united in a specific defense of this perspective nor that it might offer the most appropriate and legitimate philosophical foundation for the field. The remainder of this discussion, then, is intended to outline and describe a potential alternative foundation which transcends what may be, for all practical purposes, an unnecessary dichotomy between anthropocentric and ecocentric orientations. It begins with a return to the basic premises of ecological economics, its key metaphors, and the debate regarding the nature of the human economic actor that has galvanized much of the ecological economic critique of the neoclassical model.

4.4.6 Defining Minimum Necessary Conditions For An Ecological Economic Ethic

Ecology, broadly conceived, is the science of community, where adaptation and change are seen to maintain, in dynamic tension, the integrity of the community and its constituent elements. A more formal definition of ecology is "the scientific study of the distribution and abundance of life and the interactions between organisms and their natural environment," which includes the sum of biotic and abiotic variables (Begon *et al.* 2006). The pivotal insight here is that *the individual cannot be understood in isolation* but, rather, is defined in relation to the organisms and processes that constitute its environment and the conditions necessary to its existence. Similarly, no sub-community can be understood in isolation from the broader ecological context in which it has its existence. Ecological economics attempts to bring the ecological perspective to bear on understanding and managing the environmental dimensions of economic activity. Indeed, the ecological economic concern with constraining the scale of economic activity in order to ensure environmental sustainability is premised on a basic appreciation of the necessity of maintaining the critical ecosystem goods and services provided by intact ecosystems, which are requisite to human well-being.

For this reason, as a starting point, ecological economics must necessarily reject the neoclassical economic conception of the discrete, utility-maximizing economic agent. This conception, which views economic actors as autonomous referents interacting only through market transactions, is an abstraction of little value from an ecological perspective. The prevalence of environmental economics published in the discipline's flagship journal suggests that, despite the clear orientations of some ecological economists, this is currently not the case. Instead, the ecological economic actor must be understood first as a member of the relevant human *and non-human natural* communities that mutually constitute the conditions necessary to her existence and well-being. Although the normative implications of the economic agent as human community member have been recognized in the ecological economics literature, the broader implications of the economic agent as ecological community member have not been satisfactorily developed.

To locate potential normative mooring points for this vision of the economic actor as ecological community member, it is useful to assess its compatibility with existing ethical theories. What is immediately apparent is that traditional humanistic ethics, in which the discrete human individual is the locus of moral concern, are inadequate since these are founded on an ecologically untenable conception of the individual. For the same reason, individualist environmental ethics have little to offer ecological economics. This is true of rights and welfare-based environmental ethics whose focus is individual animals (Singer 1975; Regan 1985). Individualist biocentric ethics (Taylor, 1981) are also of questionable utility in so far as they are restricted to individual entities and expressly exclude ecological communities and processes. What is required, instead, is a platform that begins from an ecologically-informed conception of the economic actor. Towards this end, I briefly describe first the rationale for such an ethic as offered up by Leopold's Land Ethic and Naess's Deep Ecology. I next point to the divergent philosophical justifications for adopting such an ethic and suggest a middle ground for ecological economics.

The Land Ethic

According to Leopold (1949), an ethic in the ecological sense is essentially a constraint on individual freedoms in the struggle for existence. As evolutionary, adaptive phenomena, ethics therefore originate in the proclivity of interdependent individuals to progressively codify cooperative norms that enhance both individual and group success. In this light, systems of political and economic organization represent advanced symbioses structured according to mutual constraints, which have adaptive value within specific contexts. In other words, the ethics of specific human societies evolve to embody behavioral norms that serve to minimize conflict between individuals, their communities, and their socio-ecological milieu in the interests of mutual flourishing. Since the need for ethical mediation arises along lines of conflict, as socio-ecological conditions change the lines of conflict change. So too do the necessary norms and mechanisms of conflict resolution. As the insights of modern ecology reveal the interdependencies of human and non-human nature, the necessary evolution is a commensurate expansion of our ethics

(Leopold 1949; Callicott 1987, 1989). From this perspective, morally right actions are those which promote communal flourishing.

Deep Ecology

Based on Norwegian philosopher Arne Naess's doctrine of biospheric egalitarianism, which attributes equal importance to human and non-human nature across scales of organization, Deep Ecology also recognizes that humankind is an integral part of its environment, and emphasizes the importance of species, ecosystems and processes over individuals. According to Naess, the central failure of existing ethical theories resides in the narrow identification of the self – what Watts (1961) has termed the "skin-encapsulated ego." Deep Ecology therefore calls for a process of self-realization culminating in a mature experience of "oneness in diversity." By broadening the scope of that with which we identify, it is argued, we naturally develop an ethic of care for what we recognize as our greater self (Naess 1973, 1989).

4.4.7 Ecological Communitarianism As Distributive Justice

What both of these perspectives share in common is the notion of holism, which calls for the maintenance of ecological integrity in light of the ecological reality of interdependence. As noted earlier, holism has been championed under both anthropocentric and non-anthropocentric forms – a discursive struggle that has defined much of environmental ethics since the emergence of the field in the early 1970's (Norton 1991; Minteer 2009). On the one hand are non-anthropocentrist holists such as Callicott (1984, 1987, 1999) and Westra (1994), who defend an ecocentric ethic based on the attribution of non-instrumental, intrinsic value to nature (i.e. that nature is valuable in its own right, and has moral standing regardless of its usefulness to humans). On the other hand are anthropocentric holists like Norton (1984, 2005), who hold that valuing is a uniquely human phenomenon. From the perspective of Norton's "weak anthropocentric" holism (Norton 1984), protecting the full range of human values, which include non-instrumental values, similarly demands the maintenance of ecological integrity.

The seemingly intractable and increasingly acrimonious pitched battle between these two camps (Minteer 2009) is perhaps one that ecological economists need not fight. It might be argued that an ecologically informed conception of the economic actor simply dissolves the traditional anthropocentric/ecocentric dichotomy and that the terms themselves have become impediments. Indeed, proponents on either side would appear, at least in part, to endorse this proposition. To quote Westra (1994), "When we recognize the primacy of that commonality and the ways in which ecological integrity supports it for all, globally, then we are ecocentrists, or biocentric holists, because our anthropocentrism has been so weakened as to be non-existent, dissolved into the reality of our presence first and foremost, as part of the biota of natural systems." Norton (2005) calls for a transcendence of the dualistic terms of reference that characterize this debate. Simply put, if ecological economists accept ecological interdependence, then right action is that which maintains ecological integrity, regardless of the motivations that are invoked. In many respects, such a position mirrors Norton's contested "convergence hypothesis", which suggests that ecologically informed stakeholders of any stripe will, more often than not, endorse the same policy prescriptions (Norton 1989; Minteer 2009). If ecological economists can agree that the principles of ecology, when applied to humans, dissolve these distinctions, what remains is to articulate the vision of distributive justice that flows from this recognition.

In searching for an ecological economic vision of distributive justice, traditional conceptions founded in individualist, humanistic ethics are once again of little use since they are untenable from an ecological perspective. This includes the libertarian/neoliberal underpinnings of the currently prevalent Chicago School-style neoclassical economics, whose primary concern is the self-interested individual. It applies equally to the Rawlsian liberal conception of distributive justice as fairness which has been seemingly embraced by some as the appropriate banner for ecological economics. While presenting a less competitive vision of individual rights and responsibilities than libertarianism/neoliberalism, Rawlsian liberal justice is none the less founded on the same fallible conception of the autonomous agent. This point has been richly explored by

Sandel (1984), who points to the failure of liberalism, with its focus on individual rights, to account for the community in which the very entertainment of rights becomes possible.

According to the liberal view, community arises out of autonomous choice for participation for mutual advantage. In contrast, ecology sees community as prior to choice, or the background against which choice becomes possible. It is precisely this recognition of the necessity of sustaining the ecological community (which provides the context for human well-being in any other sphere) that animates ecological economics. This distinction highlights the tension inherent in dichotomies of what is good and what is right. To overcome the internal contradictions which accompany the exclusive embrace of either of these poles, ecological economics must recognize that environmental sustainability is simultaneously an overarching good as well as prerequisite to the conditions in which individual rights might be entertained. What is required, then, is a theory of distributive justice which explicitly accounts for the interdependence of human and non-human nature and the overarching value of community.

Communitarianism

Communitarianism is a relatively novel theory of distributive justice which originated in critical response to the Rawlsian theory of justice and its de-emphasis of community (Bell 2009). It is most often associated with the works of political philosophers Alasdair MacIntyre (1978), Michael Sandel (1984) and Charles Taylor (1989). In particular, communitarian philosophers take issue with the liberal assumption that the central goal of justice is to serve the individual's pursuit of liberty through securing and fairly distributing rights and resources (Bell 2009). Communitarians advance three core critiques of liberalism related to: (1) epistemological assumptions regarding social context in moral reasoning; (2) the narrow liberal conception of the self; and (3) the nature and value of community (Bell 2009).

In contrast to the universalism of Rawlsian justice, communitarians argue that principles of justice are context specific, and can only be understood relative to the conditions of the culture in question (Sandel 1984; Miller 1995). According to MacIntyre (1978) and

Taylor (1985, 1989), beliefs, practices and institutions mutually constitute the framework within which each experiences and interprets their world; thus, abstract universalism is of little use in formulating grounded principles of distributive justice.

Communitarians further challenge the individualistic conception of the self that constitutes the locus of concern in Rawlsian justice (Sandel 1984; Taylor 1989, 1992). Following the Aristotelian view that humans are social animals defined in relation to a polis rather than self-sufficient entities, it is argued that the politics of distribution must respect the need to sustain and promote the communities essential to our sense of self, rather than the simple conditions necessary for autonomous choice (Bell 2009). For communitarians, the rights of individuals must therefore be balanced against the need to foster the well-being of the communities in which we experience our lives. As a political project, the objective is to identify, protect and promote valued forms of community in place of an exclusive pursuit of individual interests (Bell 2009).

Deliberative Democracy And Discourse Ethics As Communitarianism in Ecological Economics

Emphasis by a variety of authors on the nature of the economic actor as community member suggests that ecological economics has a strongly communitarian orientation. This is even more explicitly apparent in recent contributions to the field which emphasize the value of deliberative democracy and discourse ethics (Howarth and Zografos 2008; Randhir and Shriver 2009). This approach underscores the desirability of community discourse, moral pluralism, and consensus seeking to inform policy oriented towards sustainability. For example, Randhir and Shriver (2009) investigate the use of non-price based ranking of attributes through participatory deliberation for managing complex systems such as watersheds. Such work complements the position so admirably advanced by Vatn and Bromley's 1994 "Choices, Without Prices, Without Apologies," lending further weight to arguments regarding the inadequacy of relying on environmental valuation as a basis for policy. Deliberative democratic principles are perhaps best represented by Brian Norton's contextualist approach to adaptive management for sustainability (Norton 2005).

Ecological Communitarianism

Sustainability is by nature a community concern, hence communitarianism likely provides the most compelling conception of distributive justice for ecological economics. Although the ecological dimensions of community have received little treatment in the communitarian literature, it is a modest leap to expand the principles of communitarianism congruent with the insights of modern ecology. Indeed, a mature appreciation of the ultimate dependence of human well-being on ecological integrity demands it.

The deliberative turn in ecological economics towards communitarianism is therefore welcome. In many respects, however, this communitarianism appears to ascribe to the traditional Brundtland version of sustainability, where equally important economic, social and environmental costs and benefits are weighed and balanced. For example, Norton's contextualism recognizes that human and ecological well-being are inseparable, yet there is nonetheless the caveat that ecological integrity should be conserved only where the costs (defined in a plurality of ways) are bearable. However, to the extent that the costs of diminished ecological integrity are always *unbearable* for the ecological economic actor, this caveat represents a slippery slope. Ecological sustainability provides the necessary context for any further measure of human well-being. The anthropocentric concern becomes arbitrary, not because human values are unimportant, but because the distinction itself is arbitrary. Since community values are informed by community knowledge, this strongly underscores the necessity of promoting ecological literacy.

4.4.8 Towards A Unifying Normative Framework For Ecological Economics

Although the communitarian interpretation of ethics suggests context-specificity, a macro-ecological perspective which takes into account the biosphere as a whole also points toward the possibility of unifying principles that unite the otherwise disparate ecological communitarian ethics of specific societies across socio-ecological conditions. Moreover, in an increasingly globalized society, an ethic of commensurate scope is imperative. With this in mind, might there be common conditions necessary to the objective of sustainability across scales?

Faber *et al.* (1995) develop a comprehensive theoretical basis for a teleological interpretation of the emergence and perpetuation of living entities, and advance this as an appropriate operational foundation for ecological economic theory and practice. They find support for their model in the core theoretical concepts of ecological economics. Prigogine's work on self-organizing systems (Prigogine 1961, 1977; Jantsch, 1980; Prigogine and Stengers 1984), which views all living entities as patterns of organization existing far from thermodynamic equilibrium via the mobilization of a continuous flow of low-entropy resources, is seen to embody the basic conditions necessary for the teleological realization of the individual. In turn, individuals are themselves available as the constituent elements of the higher order self-organizing systems that constitute communities (the second tele) and ultimately the ecological community as a whole (the third tele). The implications of the Entropy Law (Georgescu-Roegen, 1971), which is foundational in ecological economics, dictate the constraints necessary for sustainability in a finite biosphere, which is achieved by a balance amongst these levels of organization. The authors further posit that our current crisis of sustainability reflects an emphasis on the first tele (the individual) in our current economic system at the expense of the second and third (Faber *et al.* 1995).

Although Aristotelian teleology contributes to a specific vision of human ethics, Faber *et al.* stop short of developing the normative dimensions of their model. Moreover, I do not adopt here the teleological interpretation of ethics, although this may perhaps constitute a fruitful direction well suited to the demands of the ecological economic worldview. However, the empirical relationships described, which are foundational to ecological economics, certainly provide strong support for an ecological communitarian ethical perspective. From this vantage point, distributive justice in economic organization begins with the promotion of economic configurations that ensure the well being of the ecological community as a whole. Informed by thermodynamic principles, which dictate the most basic conditions necessary to ecological integrity across scales of organization, this conception of distributive justice clearly legitimizes the relevance and normative weight of the appropriate scale and efficient allocation criteria. The necessity of constraining the scale of economic activity relative to biocapacity has normative weight in informing the structuring of economic activities in aggregate precisely because it is prerequisite to socio-ecological sustainability – both in the local-scale contexts of specific ecosystems and for the macroscale context of the biosphere as a whole. Subject to the satisfaction of the sustainable scale criterion, the efficient allocation criterion (more broadly understood to include measures of resource and waste intensity per unit economic good or service provided) has normative weight in informing the execution of specific economic activities in that it serves the end of meeting the competing needs of interlinked human and non-human natural communities within and across generations. For the ecological communitarian economic actor, this further implies boundary conditions for right action. Here, the moral obligation is towards individual production or consumption choices which place the least possible strain on ecological systems and lifestyle orientations which recognize that the scale of our activities in aggregate must correspond to the dictates of sustainable scale. Within this context, the ecological economic actor, and communities as a whole, can then further refine their choices based on a context-specific plurality of moral considerations.

4.5. Conclusions

What we conceive of as the great environmental crises of the modern era are, in fact, simply symptoms of a single pervasive crisis. This is the crisis of identity that plagues the western industrial social model - a model that is fast becoming the dominant mode of human civilization. The story of who we are in relation to one another and the myriad other beings and processes with whom we mutually constitute the conditions of life on earth is an evolutionary anachronism that has now become the single most destructive force in the history of life. This is the story of the atomistic self of Descartes, the self-centered, instrumental rationality of neoclassical economics, the skin-encapsulated ego of the western philosophical tradition. This is the story of a self-conscious social organism in the throes of a cultural delusion in which the individual is paramount and the sum of selfish desires expressed through consumer preference in the marketplace is thought to constitute a reasonable proxy for enlightened democracy. This is the story of the belief that all values have a price. It is the story of an economic system premised on unconstrained growth embedded in a finite, encompassing and supporting ecosystem. It is the story of the loss of community, and the evolutionary-adaptive checks and balances that successful community must create and encode in ethical norms to constrain, in dynamic tension, the predispositions of its constituents with respect to the biocapacity of its ecological milieu. This is a story that must be rewritten.

Any economic discipline must provide an internally consistent account of economic activities and behaviours conducive to managing our economies in the interest of explicit, shared objectives. Neoclassical economics offers an internally consistent account, but one that is both morally bankrupt and unable to satisfactorily account for many of the most important dimensions of our activities – particularly with respect to the human and non-human natural communities and processes of which we are part. Ecological economics is well-positioned to provide a competing account which might resolve many of these deficiencies.

Sustainability is, by nature, a community rather than an individual concern. The ecological economic ethical framework sketched in broad, preliminary strokes here, and its attendant ecological communitarian conception of distributive justice, provides a strong departure point for mediating human/nature relations in economic organization towards a plurality of objectives, regardless of how these are framed. However, it must be recognized that this position requires further development. Ecological communitarianism may well represent a viable alternative to the overemphasis on individualism which permeates conventional economics, but I have not developed, beyond the most basic of assertions, why this alternative is more legitimate nor how such an alternative might be operationalized. Certainly, within our current cultural context of hyper-individualism, charges of implicit authoritarianism might be anticipated. In short, there is a need here for on-going development of the epistemological and ontological foundations of ecological communitarianism.

Moreover, this framework does not pretend to provide for the adjudication of all ethical dilemmas. Rather, it is primarily intended to serve the environmental sustainability objective, which is prerequisite to sustainability in any other sphere. As such, this vision of distributive justice should be seen as providing the baseline which circumscribes and constrains the range of choices that may be entertained by the economic actor and by society as a whole according to the overarching imperative of ecological community sustainability. Within this context, there is ample scope and, indeed, requirement, for a rich suite of ethical norms – for example, with respect to direct interhuman and interspecies behaviors - relevant within the intended frameworks of their applicability.

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CHAPTER 5: ECOLOGICAL ECONOMIC ACCOUNTING

5.1 Rationale For Ecological Economic Accounting

The increasing importance of sustainability as a policy objective has necessitated the development of accounting frameworks to establish baselines and measures of progress towards sustainability goals. These have included both monetary and biophysical techniques, as well as numerous composite indicators (Gasparatos *et al.* 2008). The focus here is biophysical techniques, which constitute one of the primary sets of tools characteristic of ecological economic accounting.

In neoclassical economics, accounting refers to the measurement, reporting, or confirmation of the robustness of financial information (Elliot and Elliot 2004). This information is used primarily by decision makers as a basis for resource allocation decisions. Ecological economics is similarly concerned with providing information to inform efficient resource allocation. However, in contrast to the pure focus on allocative efficiency in neoclassical economic accounting, ecological economic accounting has two intended applications. The first is to provide information relevant to managing economic activity according to the dictates of sustainable scale. Within this context, the second (subservient) function is to inform ecologically efficient resource allocation. Ecological economics thus employs accounting frameworks intended to serve either or both of these applications.

Ecological economics recognizes the utility of neoclassical economic measures of allocative efficiency for market goods within the intended framework of their application. However, it also requires an expanded conception of efficiency. In neoclassical economics, a state is efficient where: (a) no one can be made better off without making another worse off (Pareto-optimality); (b) further output cannot be obtained without increasing relative input; or (c) production occurs at the least possible cost per unit output. To this, ecological economics adds the concept of eco-efficiency, which refers to the resource and emissions intensity per unit economic good or service rendered relative to opportunity costs. The term eco-efficiency was first coined by the World Business Council on Sustainable Development as the provision of “competitively priced goods and services that satisfy human needs and bring quality of life while progressively reducing the environmental impacts of goods and resource intensity throughout the entire life-cycle to a level at least in line with the Earth's estimated carrying capacity" (WBSCD 1992). However, whereas this definition is frequently interpreted as maximizing resource efficiency per unit of revenue generated, ecological economics views eco-efficiency strictly in biophysical terms.

Instead of interpreting economic activities in terms of financial flows, the focus of ecological economics is thus the elucidation of the relationships between specific kinds and quantities of economic activity and the capacity of the supporting ecosystem to furnish the requisite material and energy resources and absorb the resultant wastes produced over variable spatial and temporal horizons. In other words, ecological economic accounting strives to reinterpret economic activity in terms of biophysically relevant currencies such as measures of resource dependencies and waste emission intensities.

The utility of such measures is a function of the extent to which they provide guidance in satisfying human needs in a maximally ecoefficient manner whilst ensuring that cumulative throughput is well within the limits of sustainable scale. To this end, a variety of ecological economic accounting tools have evolved or been adopted from outside the discipline that speak to various aspects and dimensions of ecoefficiency and sustainable scale. They may be roughly divided into tools addressing energy flows, material flows, carrying capacity, or some combination of these.

5.2 Energy Flow Accounting

Ecologist Charles Elton first noted that energy is the currency of the economy of nature (Worster 1994). Since the human economy is a subsystem of the biosphere, energy is similarly, by default, the fundamental currency of human economies. The First Law of

Thermodynamics informs us that, as a finite system that is energetically closed save for a discrete flux of incident solar radiation and outflow of radiative energy, useable, high quality energy resources in the biosphere are of limited quantity. The Second Law dictates that all energy transformations increase entropy, which presents scale concerns regarding both resources and wastes in a finite system. Methods to track the efficiency and scale of energy use are thus of obvious utility to sustainability objectives, and several accounting frameworks that evaluate energy flows in unique ways have been developed.

5.2.1 Energy Analysis

Energy analysis is "the process of determining the energy required directly and indirectly to allow a system to produce a specified good or service" (Brown and Herendeen 1996; pp. 220). This includes energy inputs transformed at all stages of the production process. Although energy analysis has a long history, it rose to prominence during the Arab oil embargo of the early 1970's, when energy use became an important indicator of economic vulnerability. Since the combustion of fossil fuels results in environmentally problematic emissions, energy use has also been suggested as an environmental indicator (Brown and Herendeen 1996). In recent years, the use of energy analysis has seen a resurgence due to widespread interest in energy return on investment (EROI) ratios in biofuel production (Farrel *et al.* 2006; Hill *et al.* 2006; Schmer *et al.* 2008; Mulder and Hagens 2008). This concept has also been used to document the declining productivity of conventional fossil fuel extraction (Cleveland, 2005; Gately 2007) and to study energy efficiencies in a broad range of economic activities (for example, see Giampetro *et al.* 1994; Kok *et al.* 2006; Crawford *et al.* 2006).

Given the critical role of energy in all economic activities, and the variable environmental implications of producing and consuming different forms of energy, the importance of energy analysis as an ecological economic accounting instrument is clear. It also has relevance vis-à-vis the rival nature of non-renewable fossil fuels and the intergenerational implications of depleting this resource. Although simplistic, energy analysis provides a useful indicator of ecoefficiency and a convenient proxy for several combustion-related environmental impacts. When applied at an aggregate level, it also facilitates projections of demand relative to supply.

5.2.2 Emergy Analysis

Based on the principles of energetics (Lotka 1922, 1945), systems theory (von Bertalanffy 1968) and systems ecology (Odum 1967, 1975, 1988, 1996), emergy analysis (EMA) is a quantitative analytical technique for determining "the values of nonmonied and monied resources, services and commodities in common units of the solar energy it took to make them" (Brown and Herendeen 1996; pp. 220). Emergy analysis proceeds from the recognition that much of the human enterprise depends on flows of solar energy, which ultimately limit the emergy storage and flow capacity of the global economic system. In contrast to energy, emergy is not conserved. Moreover, it measures the energy quality rather than quantity of a product/service (Herendeen 2004; Sciubba and Ulgiati 2005).

The solar emergy of a resource or commodity is calculated by expressing all of the resource and energy inputs to its production in terms of their underpinning solar energy inputs (emJoules or emj) (Odum 1975, 1988, 1996, 2000). The resulting total can then be used to calculate the "transformity" for the resource or commodity, which is a ratio of the total emergy used relative to the energy produced (emj/J). Transformities have been calculated for a wide range of materials and energy sources, and are typically used to inform analyses of other product/service systems to which the materials/energy sources are themselves inputs (Brown and Herendeen 1996, Odum 2000).

The use of average transformities is convenient and time-effective, but may compromise the accuracy of analyses, depending on geographical and temporal representivity. Nonetheless, this technique does provide valuable insight into the energy performance of economic activities in terms of our primary renewable energy resource. Of particular interest is the signals it supplies regarding current cumulative rates of non-renewable energy consumption relative to available renewable sources, which is of value to forecasting future energy scenarios (Hau and Bakshi 2004; Mayer 2008).

In theory, emergy analysis can be applied to systems across scales, although in practice necessary data are unavailable for many scales and low-resolution data compromises accuracy at larger scales (Odum *et al.* 2000; Brown and Ulgiati 2001; Brandt-Williams 2002; Mayer 2008). To date, emergy analysis has been and is increasingly applied to evaluate a variety of systems including geographical regions (for example, see Pulselli *et al.* 2008a,b; Lei *et al.* 2008), food production (for example, see Maud 2007; Rotolo *et al.* 2007; Vassallo *et al.* 2007) and industrial processes (for example, see Brown and McClanahan 1996; Min and Feng 2008; Pulselli *et al.* 2008b). Critics point to the challenge of defining emergy values for many abiotic materials, and question the physical validity of the methodology (Ayres 1998; Cleveland *et al.* 2000; Hammond 2007).

5.2.3 Exergy Analysis

Exergy is a measure of energy quality or potential that reflects the total amount of work that can be extracted as a system moves from a reference state towards thermodynamic equilibrium. Contrary to energy, which is conserved in any transformation, exergy is diminished in the sense that it is no longer available to do useful work. The concept can be traced to the work of Carnot and Gibbs during the 19th century, although the term was not coined until the mid 1950's (Gasparatos *et al.* 2008).

Exergy analysis is used to study the energy metabolism of a system. By quantifying potential energy losses, this analytical approach provides information relevant to ecoefficiency measures – both in terms of resource use efficiency and the minimization of higher entropy waste (Bejan 2002). In particular, because it enables the determination of the location, type and magnitude of waste and loss, it is useful in the development of systems that efficiently use nonrenewable energy sources (Moran and Sciubba 1994). Applied at broader scales, it speaks to thermodynamic optimization with respect to global constraints (Bejan 2002). For an in-depth description of methodology used to calculate the exergetic content of diverse substances and products, see Szargut *et al.* (1988).

Like energy analysis, exergy analysis rose to prominence during the energy price shocks of the 1970's. By the 1990's it had become the foremost technique for thermodynamic analysis in systems engineering (Bejan 2002). Numerous published reports of exergy analyses are available (for example, see Erstsvag 2001; Hammond and Stapleton 2001; Toonssen *et al.* 2008).

5.3 Material Flow Analysis

Economies are underpinned by flows of both material and energy, and by the waste assimilation services of biogeochemical cycles. The previously discussed ecological economic accounting techniques focus exclusively on the energetic dimensions of these flows. A second major category of techniques, collectively termed material flow analyses, are concerned with their material dimensions.

The structure and evolution of human societies has long been closely coupled with changes in the control, extraction and use of natural resources (Behrens *et al.* 2008). The rapid increase in resource extraction in the industrial era is now considered a major threat to both societal and ecological integrity (MEA 2005; Stern 2007; IPCC 2007). Since both extraction-related impacts and increased waste generation is related to the scale of material use, a pressing challenge is arriving at a sustainable level of material throughput (Matthews *et al.* 2000, Behrens *et al.* 2007).

Material flow analysis encompasses a suite of related techniques. Schmidt-Bleek developed Material Intensity Per Unit Service (MIPS) (Schmidt-Bleek 1993; Hinterberger *et al.* 1997, Hinterberger and Schmidt-Bleek 1999). Other material input indicators focus on economy-level descriptions of physical inputs, including Total Material Input, Direct Material Input, and Total Material Requirement (EUROSTAT 2001). All of these are primarily concerned with quantifying the gross mass of material and energy carrier throughput associated with the production of discrete or aggregate economic goods and services from a life cycle perspective (for example, see Patel *et al.* 1998 or Kytzia *et al.* 2004). It should be noted that material flow analyses typically track unweighted mass units, and do not consider qualitative differences between material flows. Material flow analyses thus signal environmental pressures, but do not describe the impact potentials of specific flows (Behrens *et al.* 2007). For a qualitative treatment of the environmental implications of material flows, see Reijnders (1998) or van der Voet *et al.* (2008).

The perceived importance of material flow analyses to increasing resource productivity and eco-efficiency is reflected in the adoption of material efficiency targets by governments such as the European Commission, Germany and Japan. This is similarly evident in the development of national material flows accounts for numerous countries including the USA, Japan, Austria, Germany, the Netherlands and the Czech Republic (Adriaanse *et al.* 1997, Matthews *et al.* 2000, Scasny *et al.* 2003, Weisz *et al.* 2006).

Beyond eco-efficiency considerations, material flow accounting can also be used to evaluate the scale of global resource extraction, which is conducive to elucidating distance-to-target for sustainable extraction limits. It further provides a basis for challenging simplistic assertions regarding dematerialization through economic development, suggested by the Environmental Kuznets Curve hypothesis, by tracing global resource flows between developed and developing regions (Fischer-Kowalski and Amann 2001, Bringezu *et al.* 2004). An analogous but less common technique called Environmental Space compares societal per capita resource consumption relative to world averages (Moffet 1996; Alcantara and Roca 1999; Hanley *et al.* 1999; Spangenberg 2001; Spangenberg and Lorek 2002). This technique has similarly been applied to debunking the Environmental Kuznets Curve hypothesis (Spangenberg 2001).

5.4 Carrying Capacity Analysis

Although applicable to questions of sustainable scale, the previously discussed ecological economic accounting frameworks are typically applied primarily to eco-efficiency questions. However, since the time of Jevons and Malthus, numerous authors have recognized that eco-efficiency alone is insufficient to achieving sustainability objectives, since increases in absolute throughput will inevitably outweigh relative efficiency gains (Arrow *et al.* 1995; Aubauer 2006; Reuveny and Decker 2000). For this reason, ecological economics has perennially been concerned with questions of carrying capacity. Borrowed from ecology, the concept of carrying capacity, as applied to human activities, refers to the maximum service a resource can provide in perpetuity without suffering declines in productivity. Since the environmental resource base underpinning economic activity is finite, understanding the scale of activity that natural systems can accommodate is prerequisite to ensuring sustainable limits are not transgressed.

5.4.1 Ecological Footprint Analysis

To date, the most widely applied indicator of carrying capacity is the Ecological Footprint (EF) analysis method developed by Rees and Wackernagel (1994). In the words of its creators, Ecological Footprint Analysis is "an accounting tool that estimates the resource consumption and waste assimilation requirements of a defined human population or economy in terms of a corresponding productive land area" (Wackernagel and Rees 1996; pp. 9).

Essentially, an Ecological Footprint is calculated by inventorying the material and energy flows required to support a given population or activity and re-expressing these flows as area of productive land required to furnish the requisite resources and absorb a subset of the resultant wastes (typically limited to CO_2 emissions) (Wackernagel and Rees 1996). The indicator thus provides a measure of resource dependency expressed in a common currency, which can be used to compare performance between systems both spatially and temporally (Wackernagel *et al.* 2004). More importantly, since ecologically productive land is ultimately limited in quantity, it also provides insight regarding cumulative consumption relative to biocapacity (Rees 1999).

Footprints can be calculated for systems at any scale, and have been variously determined for urban areas (Folke *et al.* 1997; Wackernagel *et al.* 2006), nation states (Ko *et al.* 1998; Monfreda *et al.* 2004; Moran *et al.* 2008) and numerous product/service systems (Kissinger and Rees 2007; Niccolucci *et al.* 2008). It has also been applied at a planetary scale, resulting in estimates that humanity is currently appropriating resources and producing waste at a rate equivalent to 130% of biocapacity (White 2007; GFN 2008). This ecological overshoot suggests a drawdown of the natural capital essential to the long term well-being of both human and non-human nature.

The intuitive appeal of representing consumption using a spatial metric and its usefulness as a heuristic device has attracted the attention of both fans and critics of the EF methodology (McManus and Haughton 2006). Among other things, the methodology has been criticized for: ignoring means of carbon sequestration other than by terrestrial vegetation, particularly oceanic carbon sequestration; the exclusive focus on greenhouse gases from fossil fuel combustion at the expense of considering the other environmental impacts associated with emissions such as nitrous oxide and sulfur dioxide; the inability to treat inputs such as minerals, which cannot be expressed in terms of productive land; the treatment of all land use as equivalent; and implicitly suggesting that trade is undesirable as a means for EF deficit areas to increase their carrying capacity by exchanging locally abundant ecological services for those in surfeit (Ayres 2000; Opschoor 2000; van Kooten and Bulte, 2000; Herendeen 2000). The latter issue is of particular interest and raises the important question as to whether EF measures are relevant to carrying capacity considerations at less than global scales. For these reasons, argue critics, the results are of limited utility to informing policy. Proponents respond that, despite its limitations, the Ecological Footprint is broadly applicable to policy questions about sustainability in that it describes a minimum necessary condition: that footprints, which tend to be conservative due to methodological limitations, must be smaller than the available ecological capacity. Moreover, while a certain level of trade may be permissible, a sustainable economy cannot rely on increasing imports of what is an inherently finite quantity of global ecological capacity (Rees 2000; Wackernagel and Silverstein 2000).

5.5 Hybrid Ecological Economic Accounting Techniques

All of the aforementioned techniques focus on a single dimension of eco-efficiency or scale. As suggested by Opschoor (2000), however, sustainability is a multi-dimensional objective, and no single tool is likely to provide sufficient guidance for the management of economic activity. Rather, to the extent possible, we will be better served by multi-criteria measures that facilitate simultaneous consideration of the environmental performance of discrete and cumulative economic activities along various dimensions of sustainability (Pelletier and Tyedmers 2008). Given that sustainability involves a combination of ecological, social and economic considerations, the development of a single tool which provides a rich and robust suite of information across these domains is unlikely. Within the more limited domain of biophysical environmental sustainability, which ecological economics sees as prerequisite to sustainability in any other sphere, such measures are not only feasible but also highly desirable.

5.5.1 Life Cycle Assessment

Life Cycle Assessment is an ISO-standardized methodology for inventorying the material and energy inputs and emissions associated with each stage of a product or service life cycle and translating this inventory data in terms of resource dependencies and globally problematic chemical emissions (Guinee *et al.* 2001; Baummann and Tillman 2004). This approach facilitates the identification of life cycle stages that contribute disproportionately to specific areas of environmental concern. When consistent methodological decisions are applied to different product or service systems, it is also conducive to comparing the relative ecological efficiency of competing technologies. Finally, if applied at an aggregate level, it can be used to estimate the environmental implications of industrial throughput with respect to scale considerations (Pelletier *et al.* 2008).

An ISO-compliant life cycle assessment consists of four phases, which are executed in a sequential and, frequently, iterative fashion. The Goal and Scope Definition phase involves identifying the product or service system of interest and the set of questions which motivate the analysis. Typically these relate to the performance of the system with respect to specific areas of environmental concern. The functional unit, which is the reference unit in terms of which the analytical outcomes will be expressed, is chosen. System boundaries delineating which aspects of the life cycle will be included/excluded from the analysis are identified, and methodological assumptions are decided as well (Guinee *et al.* 2001; Baummann and Tillman 2004).

The Life Cycle Inventory phase requires collecting inventory data representative of the material and energy inputs and emissions associated with each stage of the life cycle that falls within the system boundaries (Hischier *et al.* 2001). Data are expressed in terms of the functional unit. Data may be derived from a variety of sources, including primary research, peer-reviewed literature, statistical reports, persons with relevant expertise, or private companies. Several public and commercial databases of inventory data for common materials, energy carriers and transport modes are in various stages of development and are frequently drawn upon for representative background system data points (as distinguished from foreground data, which are unique to the system of interest) (Guinee *et al.* 2001; Baumman and Tillman 2004).

The Life Cycle Impact Assessment phase involves calculating the quantitative contribution of inventory data points to a suite of resource use and environmental impact categories. This requires first choosing a subset of impact categories relevant to answering the research questions established during the Goal and Scope Definition phase. A wide variety of impact categories, each with an associated impact assessment methodology, has been established for use in LCA research. These methods variously address a range of resource depletion and emission-related impacts (Dreyer *et al.* 2003).

For example, the most commonly employed impact assessment methodology for assessing the impact potential of greenhouse gases emitted during the product/service life cycle corresponds to the greenhouse gas accounting protocols endorsed by the Intergovernmental Panel on Climate Change (IPCC 2006). Alternately, a researcher may choose to construct an impact methodology relevant to answering a specific research question (Guinee *et al.* 2001; Pennington *et al.* 2004).

Once the set of impact categories has been chosen, the inventory data must be classified according to which categories each data point contributes (Pennington *et al.* 2004). In many cases, a single data point may contribute to more than one impact category. For example, an emission of nitrous oxide would contribute to both global warming and acidification impact potentials. The next step is to characterize each data point using an equivalency factor which expresses its impact potential relative to a reference species for the impact category of concern (Pennington *et al.* 2004). In the case of greenhouse gas emissions, for example, carbon dioxide is the relevant reference species. All other greenhouse gases are thus expressed in terms of carbon dioxide equivalents by multiplying each according to the appropriate equivalency factor. Impact assessment calculations may be conducted manually following the appropriate assessment protocol. Alternatively, a variety of public and commercial software platforms are available for life cycle impact assessments.

The Life Cycle Interpretation phase requires systematically reviewing the results of the Life Cycle Impact Assessment phase with respect to the research questions (Baumman and Tillman 2004). Typically this involves identifying life cycle stages that contribute disproportionately to specific impact categories of interest, and identifying leverage points for system improvements. Comparisons may also be made between production technologies or scenarios. The influence of key modeling assumptions are typically evaluated during this stage, and may inform model refinements and further analyses (Guinee *et al.* 2001; Pennington *et al.* 2004).

The strength of LCA as an ecological economic analytical framework is that it brings a variety of biophysical accounting techniques under the umbrella of a consistent methodological framework. All of the preceding techniques may be (and, with the exception of Emergy Analysis, have been) incorporated into a single life cycle assessment along with diverse other measures (for example, see Huijbregts *et al.* 2008). This allows simultaneous assessments of multiple dimensions of environmental performance, and affords rich insights with respect to the tradeoffs inherent to optimizing systems in terms of single indicators.

It should be noted, however, that the framework is most conducive to relating the environmental performance of the product/service system to globally problematic phenomena and is typically not used to evaluate local-level impacts. This is implicit in the commonly employed mid-point category indicator approach, where the environmental implications of resource use and emissions are expressed in terms of impact potentials rather than actual damage effects (Jolliet *et al.* 2004). This is clearly justifiable in the case of certain indicators such as greenhouse gas emissions, which have the same climate change potential regardless of where they are emitted. It is less defensible where the realization of reported impact potentials are contingent on local (for example, eutrophication) or regional (for example, acidification) conditions (Pelletier *et al.* 2006). End-point indicators, which link the product/service system to predicted damage effects (typically expressed in terms of human health and/or morbidity) are also available (Baumman and Tillman 2004) but introduce considerable additional uncertainty.

Life cycle assessment originated as an eco-efficiency tool for industrial manufacturing and waste management during the 1970's (Baumman and Tillman 2004). Subsequent to the Malmo Declaration (UNEP 2000), which stated that "Our efforts must be linked to the development of cleaner and more resource efficient technologies for a life-cycle economy," the LCA framework has been embraced by policy makers as a key tool for advancing sustainable production and consumption objectives (UNEP 2008). Further commitments made at the World Summit on Sustainable Development in Johannesburg (WSSD 2002) and in Marrakech, as well as the completion of the ISO 14040 standards for LCA in 2002 has lent further impetus to methodological development and application. Both the number and scope of published LCA studies is increasing rapidly, although the tool has yet to be widely adopted by ecological economists.

Based on the strengths of this method as an ecological economic accounting framework, it is chosen as the preferred framework for modeling a subset of the environmental dimensions of livestock production, as reported in Chapters 10-12. However, the currently prevalent practice of incorporating market information into life cycle assessment models seriously undermines the utility of the information provided to managing for sustainability objectives. The next chapter presents a detailed critique of this practice, and advances the case for biophysically-consistent application of life cycle assessment modeling techniques.

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CHAPTER 6: THE USE OF MARKET INFORMATION IN LIFE CYCLE ASSESSMENT: REARRANGING DECK CHAIRS ON THE TITANIC?

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6.2 Abstract

The rising prominence of life cycle assessment (LCA) and similar environmental accounting frameworks reflects increasing awareness of the pressing necessity of managing both for eco-efficiency and with respect to the macroscale, environmental dimensions of the material/energy flows and emissions which underpin all economic activity. However, by relying on environmentally myopic market signals to inform evaluations of the biophysical dimensions of economic activity through the use of economic allocation (in attributional LCA) and market-based system delimitation (in consequential LCA), we are concerned that researchers greatly compromise the value of their work to furthering these objectives. We herein critique the use of market information in both attributional and consequential LCA, and present the case for an ecological economic approach to the execution, interpretation and application of biophysically-consistent life cycle assessment research specifically intended to elucidate the environmental dimensions of meeting human needs. We further argue that, although LCA has historically been limited to informing eco-efficiency considerations, it can and should also be used to manage for sustainable scale, which is the first condition of sustainability.

6.3 Introduction

It is increasingly clear that the cumulative metabolic throughput of industrial society greatly exceeds the resource provisioning and waste assimilatory capacity of global ecosystems and biogeochemical cycles across scales (MEA 2005). Intellectual and political responses to discrete manifestations of systemic overshoot, for example "environmental crises" including ozone depletion, global warming, and biodiversity loss along with various forms and scales of resource depletion, have varied along with their relative success. Overwhelmingly though, political responses have manifested a powerful allegiance to standard growth-oriented development models founded on neoclassical economic principles. In practice, this has meant adopting strategies that attempt to reduce environmental harm per unit of economic throughput (an understanding of eco-efficiency as advanced by the World Business Council on Sustainable Development) while encouraging continued economic growth as the engine of "greener" development (Redclift 1997; Luke 2005). The rapidly emerging ecological economic perspective, starting from a basic recognition of the implications of the Laws of Thermodynamics for economic activity in a finite biosphere, presents a fundamental challenge to this growth-oriented world view and a platform from which to address the underlying drivers of overshoot (Georgescu-Roegen 1971; Cleveland and Ruth 1997; Pelletier *in press*). Specifically, it underscores the essential "blindness" of the neoclassical economic model and the market signals generated within our current economic system to the pivotal role of ecosystem goods and services in economic activity, and the central importance of managing economic activities from the perspective of biophysically sustainable scale (Daly 1991, 1996, 1999; Ayres 1998; Krysiak 2006).

As practicing ecological economists, we are encouraged by the rising prominence of a range of biophysical accounting frameworks (e.g. life cycle assessment (LCA), energy analysis, ecological footprint analysis, carbon footprinting, etc.) designed to further our understanding of an important subset of the environmental implications of economic activities. We are, however, deeply concerned that the efficacy of these methodologies to informing environmental management may be inadvertently curtailed by the prevalent and uncritical use of market information in systems modeling. It is our thesis that by introducing environmentally myopic signals generated within the current market system (whether through economic allocation in attributional LCA or market-based system delimitation in consequential LCA) into a framework ostensibly intended to evaluate the biophysical dimensions of economic activities, researchers, at best, risk compromising the utility of their work. At worst, it may result in policy prescriptions that exacerbate current problems while providing the patina of biophysically enlightened decision-making. We therefore present the case for an ecological economic approach to the execution, interpretation and application of biophysical systems modeling, using the example of life cycle assessment (LCA) research in food systems.

Given the diverse backgrounds of those currently engaged in biophysical modeling practice, we begin with a brief discussion of key weaknesses of the neoclassical economic paradigm (which governs the current market system) in accounting for the foundational role of ecosystem goods and services in economic activity. Specifically, it is argued that current economic values and other market signals do not, in any meaningful way, reflect the environmental dimensions of economic activity. We subsequently explain how ecological economics attempts to obviate this analytical and practical deficiency through the application of a thermodynamic perspective on the nature of human activities.

We then critically review the role of life cycle assessment research in contributing to environmental management for either eco-efficiency (understood as resource and waste intensity per physical unit rather than monetary value produced) or sustainable scale objectives. In particular, we focus our critique on the common usage of market information in both attributional and consequential LCA, with particular attention to allocation decision-making and market-based system delimitation. We then describe what we see as the fundamental limitations of these approaches in representing material/energy flows and their associated environmental impact potentials in a realistic manner. Consequently, we propose an alternative, biophysically-consistent approach to life cycle modeling which, we argue, provides a much more realistic and useful representation of the environmental dimensions of meeting human needs through economic activity than do the currently prevalent, market-based approaches.

6.4 Discussion

The global economic system comprises the sum of activities through which humans mobilize and transform the material/energy resources which underpin economic goods and services, and the principles and mechanisms by which they are distributed (Fischer-Kowalski and Haberl 1997). Importantly, the guiding principles for understanding and managing economic activities within the contemporary neoclassical economic paradigm are largely oblivious to the former (biophysical) aspect, focusing almost exclusively on the latter (political economic) aspect of exchange (Hall and Klitgaard 2006).

According to standard (neoclassical) economic theory, for markets to efficiently allocate goods and services, several criteria must be met. These include that: the goods in question be marketable goods (rival and excludable); the factors of production (natural, manufactured and human capital) are perfect substitutes for one another; externalized costs and benefits are negligible; market participants have access to perfect information; transaction costs are low; and all markets are fully competitive. Suffice to say that a considerable body of literature has been advanced detailing how and why ecosystem goods and services typically fail to meet these criteria (for example, see Barbier 1993; Cleveland *et al.* 2001; Daly and Farley 2004). As a result, efficient markets for ecosystem goods and services are rarely, if ever, realized. Since *all* economic activities are underpinned by ecosystem goods and services, we are thus faced with a state of chronic market failure in part attributable to the inability of our existing economic system to adequately account for changes in environmental amenities. For this reason, market information, as communicated in prices and other market signals, rarely embodies environmentally relevant information.

The trans-discipline of ecological economics arose precisely to address this analytical deficiency. Ecological economics proceeds from the basic recognition that the human economy is a subsystem of the encompassing and supporting economy of nature. Moreover, it is recognized that as a materially closed system subject to a discrete flux of solar and radiant energy, the biosphere has an inherently finite resource provisioning and waste assimilatory capacity (Georgescu-Roegen 1971). Since the Laws of Thermodynamics dictate that economic activities are inextricably underpinned by material/energy flows and inevitably result in the production of waste, achieving sustainability ultimately requires that the scale of the human economy be constrained relative to biocapacity (Daly 1991, 1996; Cleveland and Ruth 1997; Daly and Farley 2004).

This fundamental basis for understanding the relationships between energy, matter and waste has long been considered incontrovertible in the natural sciences (Faber *et al.* 1998; Baumgartner *et al.* 2006), but has been largely overlooked in neoclassical economics. The implication for social organization is, however, profound: as the implicate order for all human activity, the sustainability of the biophysical environment is prerequisite to sustainability in any other sphere. In contrast to the prevalent conception of sustainable development as a balancing of equally important environmental, social, and economic objectives, the focus is necessarily shifted towards prioritizing the former in order to ensure the possibility of the latter.

An important role of ecological economics, then, is to elucidate the biophysical dimensions of economic activity – essentially considering human activities from the same perspective as the ecologist considers any other species. By re-expressing economic activity in terms of flows of resource inputs and waste emissions, ecological economists provide the information necessary to understand and improve the eco-efficiency (understood here in biophysical as opposed to monetary terms) with which human needs are met, and to ensure that the limits of sustainable scale are not transgressed.

6.4.1 LCA As An Ecological Economic Accounting Framework

Life cycle assessment is an ISO-standardized biophysical accounting framework used to (1) inventory the material and energy flows associated with each stage of a product/service "life cycle" and (2) quantify how these contribute to a suite of resource use and emissions-related environmental impacts. Since the concern is with understanding the environmental effects of human-mediated flows of matter/energy and waste, LCA distinguishes system boundaries between the "ecosphere" and the "technosphere." As a tool for ecological economic analysis, LCA is particularly attractive in that it provides an internally consistent framework for reinterpreting economic activities in terms of biophysically relevant currencies, which is essential to managing for either eco-efficiency or scale objectives.

To date, two distinct methodological approaches to life cycle assessment have been advanced. These are known as attributional and consequential LCA. An attributional LCA aims at isolating and describing the average environmental properties of a product/service life cycle (Curran *et al.* 2002). In other words, it is used to model the direct environmental impacts of a product's life cycle, but does not consider indirect, market-mediated effects. In contrast, a consequential LCA is explicitly oriented towards describing the environmental consequences of changes in output as a result of market-mediated changes in production and consumption (Weidema 2003). The use of market information to inform system modeling is currently prevalent in the application of both attributional and consequential methodologies.

As we have already established, however, market information generated within the current economic system is largely devoid of environmentally relevant content. Accordingly, we question the utility of LCA models whose outcomes are strongly influenced by the explicit or implicit use of market information– particularly given that such analyses are undertaken largely as a direct response to the recognized inadequacy of existing market signals to assist in managing the environmental dimensions of our activities. The next section addresses both attributional and consequential methods, and emphasizes the preconditions for each to serve as useful sources of information for effective environmental management.

6.4.2 Co-Product Allocation In Attributional LCA Of Multi-Output Systems

Attributional LCA was first referred to in Curran *et al.* (2002; pp. 5), building upon earlier work by Frischknect (2000) and Tillman (2000), where it was defined as an attempt to answer "how are things (pollutants, resources, and exchanges among processes) flowing within the chosen temporal window?" In other words, attributional LCA attempts to explicitly quantify material/energy flows and emissions associated with specific economic activities when considered in isolation. Depending on the scale of analysis and the number of units assessed, this often involves using average data representing multiple production units.

Representing these flows and their consequences in a realistic and useful manner requires several conditions be met. First, it is necessary to collect representative data for the inputs and outputs characteristic of the modeled systems, and to employ robust impact assessment methodologies. Although data quality is a function of the thoroughness of the researcher, the appropriateness of the system boundaries, and the limits of availability, it is certainly possible to achieve a high degree of representivity. Similarly, many of the impact assessment methods commonly used in LCA are based on robust, peer-reviewed protocols, thus ensuring that the inventory data are translated into impact assessment results with defensible predictive ability. A second determining factor resides in how a researcher decides to allocate environmental burdens among co-products in attributional LCAs which address multi-output systems. Indeed, this variable can be critical, as results can hinge on the allocation rule applied.

The ISO 14044 standards (ISO 2006; pp.4) for LCA define allocation as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" - in other words, isolating the share of flows attributable to the co-product of interest. Here, a co-product is understood as any of two or more products coming from the same unit process or product system (ISO 2006), which is quite distinct from the understanding of a co-product as one that contributes to the income of the producer (as defined by Weidema (2003)).

Certainly, the question of appropriate allocation criteria has perennially been an important methodological issue in attributional LCAs of multi-output systems and elsewhere, with numerous papers devoted to the topic (Heijungs and Frischknect 1998; Guinee 1999; Frischknect 2000; Ekvall and Finnveden 2001; Guinee *et al.* 2004; Curran 2007; Ayer *et al.* 2007). In large part this contention exists because allocation decisions are inevitably somewhat subjective (Frischknect 2000). Since there is no one unambiguous basis for allocation, a decision is required that is subject to or influenced by the analyst's perspective/world view.

Frischknect (2000) suggests that it is therefore necessary to establish a well-defined rationale by which to justify the choice of a particular allocation method. Following Schumpeter (1950) we would add that such a rationale must flow logically from the "pre-analytic vision" of the analyst. Indeed, a review of the allocation literature suggests that linking allocation decisions with the broader objectives of life cycle assessments is considered highly desirable (Ekvall and Finnvedan 2001; Curran 2007).We do not think it controversial to suggest that an overarching objective in the choice of allocation criteria in LCA research should be to produce information that allows us to better understand and manage the environmental implications of the material/energy flows associated with meeting human needs via economic activity. This normative decision, which is implicit to the methodological framework itself, then provides a context in which specific allocation criteria may be considered and defended.

Towards this end, the desirability of basing allocation criteria on relational properties which link system inputs and co-product outputs in a logical manner is widely recognized. Allocation procedures should therefore approximate as much as possible such fundamental input/output relationship and characteristics (ISO 2006). Within this context, "causality" is generally accepted as the most appropriate relational property by which to justify allocation decisions (Frischknect 2000; Guinee *et al.* 2002; Feitz *et al.* 2007). Tillman (2000) differentiates between "cause-oriented causalities" and "effect-oriented causalities." For obvious reasons, physical (a kind of cause-oriented) causality (i.e. some physical property of the inputs which determines the proportion of outputs produced) is deemed preferable. However, it is widely held that physical causality is not definable in multi-output systems where the inputs and outputs cannot be independently varied, which is the case for all joint production (Ekvall and Finnvedan 2001).

In lieu of physical causality, socio-economic (cause-oriented) causality has historically been advanced as the next best alternative (Guinee *et al.* 2002; Guinee *et al.* 2004). In this context, the economic value of co-products is the most commonly employed socio-economic allocation criterion. Its use is justified according to the rationale that the economic value derived from the production of specific co-products provides the causal impetus for the existence of the product/service system (Tillman 2000). For example, where the processing of fish yields both fillets destined for human consumption as well as trimmings utilized by the animal feeds industry, economic allocation results in the attribution of burdens in proportion to the revenue streams from these two co-products. Hence, in most instances, the largest share of burdens is assigned to the more valuable fillets. The use of other relationships, such as the relative mass or energy content of co-products, as the basis for allocation has often been deemed arbitrary because they are not based on any such "causal" relationships (Huppes and Schneider 1994; Ekvall and Finnvedan 2001; Feitz *et al.* 2007).

Certainly, researchers have variously employed a range of allocation criteria (for a review of the pertinent literature for food systems, see Ayer *et al.* 2007 and Schau and Fet 2008), and sensitivity analyses to test the impact of different allocation choices are common (for example, see Cederberg and Stadig 2003 and Feitz *et al.* 2007). Nonetheless, it is interesting to note that few of these discussions appear to consider whether such allocation decisions produce results that represent the flows of material/energy inputs and emissions in a biophysically realistic manner, much less a manner meaningful to managing how we meet human needs within environmental sustainability constraints.

Certainly, this does not appear to be considered in the ISO prescription for allocation decisions, nor has it been generally discussed by researchers with reference to their own published results. Moreover, outcomes where economic allocation is applied are typically quite different from those where biophysical criteria are used (for example, see Cederberg and Stadig 2003; Feitz *et al.* 2007; Pelletier and Tyedmers 2007). Consequently, we are puzzled by the logic that has been extended in defense of the economic allocation approach for several reasons, all of which stem from our appreciation of the inability of market signals to communicate environmentally meaningful information.

First, it is worth noting that physical and socio-economic (cause-oriented) causality are two very different things. It is unclear why the latter has been so widely accepted as a reasonable alternative to the former. An inherent physical property that determines the stoichiometry of co-product outputs constitutes a causal factor in a sense that bears little resemblance to the manner in which potential monetary gain might be thought of as a causal factor in economic activity. Certainly, the prospect of economic gain may provide partial motivation for an activity. However, while there is clearly a causal relationship between the existence of an activity and the resultant environmental impacts, this does not signal the existence of a causal relationship between the economic *values* that are attributed to co-produced economic goods and the environmental implications of their production.

While the literature is unclear as to what *purpose* the use of economic allocation is intended, we interpret that it may be intended to reflect a business-oriented perspective that couples the profit motive with eco-efficiency objectives, where the latter is understood in terms of resource/emissions intensity per unit economic value generated. This would explain the (to us) somewhat odd conclusion of Frischknect (2000) that "allocation in joint production is mainly performed for reasons of *competiveness* and not for reasons of finding the economic or environmental "truth*"*" (pp. 94, original emphasis). This implies that the analyst shares the objective of the firm, or assumes that the societal objective is the same. Indeed, this corresponds well with the notion of utility maximization in neoclassical economics, and sustainable development as unconstrained, albeit greener, development. From an ecological economic perspective, however, "...if we need an 80% reduction in material/energy throughput by 2050, growing efficiently does not address the problem. It merely makes us more efficiently unsustainable (Rees 2009)." We therefore stress the importance of *truth* over *competitiveness* as the basis for allocation.

Moreover, even if it is accepted that (1) causality is the necessary basis for choosing allocation criteria and (2) causality should be defined in a purely anthropocentric sense, surely there is a multitude of biophysical variables that are much more relevant than the profit motive upon which to base the attribution of causality. The root "cause" of most economic activity is provided by human need, real or otherwise, for the specific biophysical characteristics of products and services. For example, for much of humanity it is the need for a minimum quality and quantity of calories that is the causal force behind economic activities related to food production.

Second, even a cursory consideration of the nature of economic value points to powerful reasons why it should not be considered an adequate allocation criterion. For example, it is well known that economic values fluctuate with time and place in response to changes in supply, demand, regulation, subsidies, culture, and the like. One can thus easily imagine situations where the economic value of a given co-product may fluctuate between positive, neutral and negative values depending on market conditions and cultural context. Otherwise identical analyses using economic allocation would therefore indicate that the environmental implications of producing such a co-product would similarly fluctuate between positive, neutral and negative values. From a biophysical perspective, this is patently absurd. Moreover, even were this not the case, conducting time-intensive LCA research to inform decisions with relevance in such narrowly circumscribed spatial and temporal contexts seems to us a rather pointless endeavor.

These weaknesses are inherent to the nature of economic value, itself. To overcome the limitations of direct exchange (barter), money is used as a store of exchange value for economic goods and services. Money is thus the dominant (symbolic) currency of market economies (Daly and Farley 2004). However, due to the persistence of various forms of market failure, we know that money is a highly imperfect symbol for the flows of material/energy which underpin the goods for which it stands in proxy and, by consequence, of the ultimate environmental implications of economic activity. In other words, there is no logical relationship between culturally-defined economic exchange values and the relative shares of the flows of matter/energy/entropy attributable to the co-products of economic activity. Rather, if economic value actually provided an adequate proxy for the environmental implications of these flows then analyses such as LCA would be largely redundant. Clearly, this is not the case. Frischknect (2000) makes the interesting suggestion that one might assign monetary values to environmental costs derived from an LCA then aggregate these with private cost data as a basis for allocation in order to better reflect the "social costs" of the product system. Setting aside the challenges inherent in such a complex valuation exercise, this proposal clearly recognizes that current prices do not, in fact, reflect environmental burdens.

For the reasons described, consideration of recent LCA research employing economic allocation reveals, from the perspective of an ecological economist, some discordant results. For example, in an LCA of feed production for aquaculture, Papatryphon and colleagues (2004) note that their results suggest that fishmeal produced from the trimmings of fuel-intensive fisheries for human consumption generate lower impacts than fishmeal from more fuel-efficient reduction fisheries due to the use of economic allocation. Here, it would appear that such fishmeal is less emissions-intensive simply because human culture, in a particular space-time configuration, has deemed fisheries trimmings to be of lesser economic value than fish fillets. From this logic, one must assume that a relative rise in the value of trimmings would in some logical way result in an increase in the resource use and emissions intensities associated with their production. Or conversely, the unremunerated use of trimmings would result in the provision of a biophysically "free" resource despite the scale of inputs and impacts associated with its acquisition. Similarly, in an analysis of shrimp production, Mungkung (2005) used economic allocation based on relative prices to partition burdens in a fishery that yielded several tonnes of low-value fish used as an input to shrimp feed and a relatively trivial quantity of highly valued shrimp broodstock. As a result, the apparent environmental impacts of shrimp feed production largely disappeared. One wonders how remarkable would be the change in environmental fortunes of this fishery and of shrimp aquaculture should wild-caught broodstock no longer be required.

We also object to the widely cited opinion that physical allocation criteria are somehow arbitrary compared to economic allocation. Managing the environmental implications of our industrial economy requires management tools that adequately account for the material/energy and entropic dimensions of economic activity. Since physical transformations of matter and energy are the fulcrum for all economic activity, and it is the biophysical implications of these transformations that are of interest in LCA, we believe that physical properties of co-product streams actually provide the most logical basis for apportioning inputs and emissions in multi-output systems. Consistent with the Laws of Thermodynamics, waste emissions represent the inevitable entropic losses associated with transforming raw materials and high quality energy into desired outputs. Moreover, they are produced in quantities that can be related to physical properties inherent to both inputs and co-product outputs in a consistent manner, regardless of whether or not the amount of co-products produced can be independently varied.

A physical property such as mass is certainly a somewhat crude common denominator for these relationships. However, there are a range of other physical/chemical properties that might be chosen as allocation criteria, which simultaneously reflect both the biophysical nature of the process and the causative impetus provided by specific human needs. This is particularly true if the allocation criterion and functional unit (the physical quantity of the product/service system to which impact assessment results are related) are defined according to the same biophysical currency.

Here, it is necessary to establish that we advocate the same delineation of system boundaries (i.e. at the biosphere/technosphere interface) typically used in LCA. However, we stress that activities in the technosphere (i.e. human-mediated flows of resources and wastes) must be defined in biophysical terms rather than economic terms, which are unrepresentative in this context. This will require continued development of appropriate terms and criteria.

We take as an example the flows of material/energy inputs and emissions in human food production systems. Accepting that the root motivating force behind food production is human need for a minimum quantity and quality of food energy, the caloric energy content of food co-products, regardless of their subjective value to the consumer, provides an allocation criterion that is causal in both a biophysical and a social sense. For this reason, tracking its flow through multi-output food production systems produces information which simultaneously reflects the underlying thermodynamic nature of the process, and the efficiency with which a specific human need is met. By defining the functional unit in terms of a specified amount of food energy delivered, the analysis obtains a logical biophysical consistency throughout. It is exactly this consistency that is necessary in order for the information derived to be of value for environmental management. For example, in the case of processing soy beans to produce soy meal and oil, if we define our functional unit as a quantity of caloric energy potentially available for human nutrition and allocate the environmental burdens of soy bean production and processing according to the respective caloric content of the meal and oil co-products, our analytical outcomes will actually reflect the environmental implications of producing the functional unit. This is because energy content is an inherent property of the raw material which is apportioned between the co-product streams in a manner which speaks directly to the eco-efficiency with which the functional unit is provided (and a specific human need is satisfied). If the co-products are subsequently used as animal feed, then the amount required to produce an equivalent caloric output in the form of meat will effectively capture the entropic processes of feed conversion. Defining the functional unit based on specific properties (functions) of food products has been previously suggested in review papers by Marshall (2001) and Schau and Fet (2008), and employed by various researchers (see references therein). Schau and Fet also underscore the relevance of biophysical allocation criteria.

It must be noted, however, that it is both difficult and counterproductive to prescribe a single biophysical allocation decision rule. Rather, allocation criteria should be chosen and justified in the context of each analysis according to the overarching objective of managing the environmental dimensions of meeting specific human needs. Furthermore, the example we have provided is a simple case. Such decisions and their justifications will be much more complex where it is necessary to allocate burdens between co-products with divergent functions – for example, where soy processing co-products are used as meal for animal feeds and as oil for biodiesel.

A potential argument in support of economic allocation here is that it at least provides a consistent criterion that can be used along supply chains, regardless of sector or application. However, consistent signals are of little value if they consistently fail to reflect biophysical realities while inconsistently varying with changing circumstances. We thus maintain that from our perspective as ecological economists it is preferable in all cases to choose a biophysical allocation criterion over one (such as economic allocation) that introduces the distorting biases of the current economic system into the analysis. A potential fall-back criterion is exergy, which captures the relative potential energy of co-products and, at least, reflects the most elemental currency of economic activity and the "motivating" factor in the behaviours of all self-organizing systems. In all cases, employing a sensitivity analysis using a range of relevant biophysical characteristics would provide a richer suite of information reflecting the environmental ramifications of meeting the multiple dimensions of human needs.

We explicitly underscore and stand behind the desirability of exploiting this opportunity rather than treating it as a weakness of our approach. Whereas contemporary economic wisdom is that all values are commensurate and may be reduced to the price numeraire, we believe that our management decisions must be based on considerably more nuanced information. When we assess the impacts of meeting a variety of needs served by specific products using appropriate biophysical allocation criteria and corresponding functional units, then we arrive at a much richer understanding of the environmental dimensions of meeting the needs served by the product system than if we simply use one arbitrary allocation criterion.

A second justification that has been advanced in support of economic allocation is that it produces results that penalize valued products, thereby providing incentives to producers to reduce environmental impacts. For example, in the co-production of gold and copper, it might be argued that because economic allocation would attribute the majority of burdens to the gold, the mining company will have an incentive to reduce the impacts of gold production. Instead, were the mine manager held responsible for the sum total of environmental emissions, the necessity of also mitigating the environmental impacts of copper production would provide a similarly effective incentive than would attributing virtually all impacts to the gold based on the rather arbitrary fact that clever chimps prefer rare, shiny objects.

More importantly, as the allocation criterion is chosen in order to obtain the result whereby the most valued co-product will be the most penalized, an expressly normative element is introduced into the modeling exercise. In other words, the result is decided a priori to the analysis, effectively stripping the methodology of objectivity, rigor and, we would argue, legitimacy. This is very different from choosing a biophysical allocation criterion that facilitates quantifying the impacts of fulfilling a specified (function) human need in an empirically robust manner. Certainly, there is a place for a normative dimension in LCA. However, this normative element lies in the recognition of the utility of the methodology in serving sustainability objectives, which is operational at a level that legitimizes our research endeavours but in no way predetermines the outcomes of our modeling exercises.

A final potential argument for economic allocation is that in theory, economic value effectively reflects utility, and we should ultimately be concerned about managing for utility maximization. Setting aside the utility-masking effects of subsidies and related market distorting interventions and the moral baggage of the neoclassical utility model, it is internally inconsistent to appeal to the logic of using economic value as an allocation criterion based on the fact that it constitutes a common denominator for utility, and to simultaneously support the use of economic allocation based on the argument for causality because this approach systematically penalizes co-products in direct proportion to their supposed utility.

In sum, the use of economic allocation produces results that simply mirror existing, environmentally myopic market signals. The use of information derived from attributional LCA research of multi-output systems which employs economic allocation in an environmental management context is thus an exercise akin to rearranging deck chairs on the Titanic, as results generated do not provide a meaningful basis for redirecting our activities towards the sustainability objective. If the goal of LCA is to understand and manage the biophysical environmental implications of the material/energy flows and emissions associated with meeting human needs, introducing market information into the analysis via the allocation process is antithetical to that purpose. In contrast, the use of biophysical allocation criteria in attributional LCAs involving co-production situations, which allow us to express flows and emissions in terms of the functional properties that actually motivate their production, provides information relevant to both improving the eco-efficiency with which we meet human needs and, if we choose, to further managing the cumulative throughput of economic activity with reference to the limits of sustainable scale (see below).

6.4.3 Consequential LCA: Strengths And Limitations

According to Weidema (2003, pp. 9) “the fundamental rule to apply in all methodological choices in life cycle assessment is that the data used must reflect as far as possible the processes actually affected as a consequence of the decision that the specific life cycle assessment is intended to support.” Consequential LCA, which typically uses market-based system delimitation, is therefore intended to describe the environmental implications of *changes* in a product or service system including, importantly, those induced by associated market changes as a basis for decision making. This is accomplished by identifying the marginal economic process that may be affected by the change in the product/service system, predicting the extent of the affect using market models, and accounting for the associated changes in environmental inputs and emissions at the market level. Consequential LCA thus solves the co-product problem through system expansion. It is claimed that this approach side-steps the allocation problem (Weidema 2003; Weidema and Ekvall 2009) although, following the distinction presented by Tillman (2000) this might be interpreted as allocation based on "effect-oriented causality".

An advantage of the consequential approach is that it encourages managers to take a systemic perspective, and recognizes the ultimate interconnectedness of the global economy (Thrane *et al.* 2007). For example, Ekvall and Andrae (2006) examine the potential effects of a sectoral shift to lead-free solders using both attributional and consequential LCAs. In their consequential model, the discontinuance of lead use in solder production is assumed to result in a lower price for lead. Lead-acid battery production is then identified as the (marginal) market that will be affected, and (following standard, neoclassical theory) it is hypothesized that decreased lead prices will result in increased battery production and use. The authors go on to predict that these batteries will then be used entirely to store energy collected with solar panels in remote locations, thus offsetting the use of diesel fuel in generators and decreasing global warming emissions. Ekvall and Andrae further observe that, while the attributional study provides the obvious insight that the environmental burdens of lead use are eliminated from the life cycle of soldered products, the consequential approach reveals that this shift will (or perhaps more accurately, could) be partially off-set by increased use of lead in batteries, as well as the associated reduction in GHG emissions in the economy as a whole.

There is clearly much intuitive appeal in the consequential approach. Certainly, it would be of great value to environmental policy makers and managers to be able to consider the broader environmental implications of changes in product/service systems at the market level (Thrane *et al.* 2007) and take these into account in decision-making. However, there are two distinct challenges that compromise the efficacy of this methodological approach. The most obvious is the challenge of arriving at robust models of market change (Guinee *et al.* 1999). As demonstrated by Ekvall and Andrae's analysis, the consequential approach is certainly conducive to considering the potential environmental implications of almost any scenario a researcher can envision. Whether these scenarios accurately reflect real world conditions and can hence provide information relevant for environmental management is, however, another issue altogether. The difficulty of developing robust models of market change has perennially been considered the Achilles heel of the consequential approach (for example, see Guinee *et al.* 1999). Several severely confounding factors, including variable elasticities of supply and demand, rebound effects (Stern *et al.* 1996; Binswanger 2001), and various forms of market failure previously identified, may simply preclude the development of realistic models (Guinee *et al.* 1999). These factors and how they might best be accommodated have recently been discussed by Weidema and Ekvall (2009). We laud any and all efforts to develop robust predictive models. However, where these efforts fail (and we believe this will most often be the case), the results produced may be generally interesting but of little value in a management context.

Let us imagine, however, that these obstacles are surmountable and that the complexities of the global economic system can be reduced to simple models with powerful predictive ability. Under these conditions, consequential LCA using market-based system delimitation would absolutely provide valuable insights as to the global environmental repercussions of changes in product/service systems *within the current configuration of economic organization*. Such information would be conducive to optimizing eco-efficiency given the constraints of the status quo. As described previously, however, eco-efficiency alone is insufficient to achieving environmental sustainability so long as aggregate throughput continues to increase. In other words, while an idealized form of consequential LCA based on robust models of market change would, indeed, be valuable in the context of tweaking the current configuration of economic organization, we know that the current configuration is fundamentally unsustainable. Moreover, since this approach relies on market information and models that are themselves deeply flawed from a biophysical perspective, its efficacy in informing either effective eco-efficiency measures or sustainable scale considerations is severely hampered. As with economic allocation in attributional LCAs of multi-output systems, the outcomes can only inform exercises that are themselves at best akin to rearranging deck chairs on the Titanic.

6.4.4 LCA, Eco-efficiency And Scale

Life cycle assessment was developed and has largely been applied as an eco-efficiency tool. If life cycle models accurately reflect the biophysical flows of material, energy and waste associated with the economic activities they are meant to represent, LCA can be an excellent tool in this respect. Eco-efficiency, however, is an important but insufficient objective in the pursuit of environmentally sustainable economic organization (Krysiak 2006). Rather, the singular pursuit of eco-efficiency as an environmental policy and management objective is, by itself, yet another exercise in futility reminiscent of rearranging the Titanic's proverbial deck chairs. Instead, the ultimate objective must be to constrain the biophysical scale of economic activity such that the resource provisioning and waste assimilatory capacity of the biophysical environment can be maintained in perpetuity. This raises important questions regarding the potential role of LCA and comparable tools in contributing to this objective.

Achieving environmental sustainability will be greatly facilitated if our evaluations of the environmental dimensions of economic activity can be interpreted with reference to clear indicators of sustainable scale. In other words, whereas LCA has historically been used to quantify the environmental dimensions of discrete activities, it should also be used to estimate *cumulative* effects and to inform policy initiatives focused on scale considerations. This has indeed been undertaken, for example the large-scale modeling of food systems in the EU by Weidema and colleagues (2008), and by researchers using input-output LCA models (for example, see Weber and Mathews 2008). Similar to treating eco-efficiency concerns, doing so in a useful manner requires first that LCAs produce information which realistically reflects the biophysical dimensions of economic activities. As a starting point, we believe this requires the exclusion of market information where the use of such information produces results which effectively mirror existing market signals rather than provide a window onto the underlying biophysical realities. This will also require greater attention to the development and use of robust normalization data sets across impact categories, and clear priorities regarding the shares of resources and wastes sinks that are to be allocated to specific sectors of economic activity.

6.5 Conclusions

It is hardly debatable that the cumulative resource demands and waste emissions of industrial society are increasingly in conflict with the integrity of ecosystems and the stability of biogeochemical cycles across scales. Nor is it particularly contentious to claim that our current approach to environmental management, with its focus on more albeit greener development, is failing to reverse this trend. As suggested by the ecological economic perspective, resolving this dilemma requires foremost that economic organization be restructured with reference to the dictates of environmentally sustainable scale – the achievement of which will be much facilitated by eco-efficiency measures

LCA may provide the most useful framework currently available for understanding the environmental implications of the material and energy flows which underpin all economic activity, and for managing them with respect to both eco-efficiency and scale considerations. Its utility, however, is ultimately a function of the extent to which life cycle models actually represent these flows and their repercussions in a manner which accurately reflects the environmental dimensions of meeting the human needs which provide the causal impetus for economic activity.

As we have argued, from our perspective as practicing ecological economists, the pervasive current practice of introducing market information into life cycle models, whether through economic allocation or market-based system delimitation, severely undermines the utility of the research outcomes to managing for sustainability objectives. Market signals generated within our current economic system are inherently flawed due to the inability of the price-determining neoclassical paradigm to account for the central role of ecosystem goods and services and the limits of sustainable scale. As a result, the implicit and explicit use of price-based information in LCA introduces distorting biases that fundamentally misrepresent the environmental dimensions of economic activity from a biophysical perspective. We therefore propose that LCA models explicitly exclude market information that introduces these distorting biases, and rely instead on best-fit biophysical parameters in all cases. For attributional LCAs examining multi-output systems, this means rejecting economic value as the preferred allocation criterion. Instead, we advocate the use of biophysical criteria that simultaneously reflect both inherent physical properties and social functions of the co-product streams and hence directly represent the environmental implications of meeting specific needs through economic activity. We recognize that this might not always be straightforward. For consequential LCA using market-based system delimitation, this implies that unless and until market signals fully reflect the true environmental dimensions of economic activity (in which case such analyses would arguably be redundant), and researchers are able to model these changes in a robust manner, its efficacy will perennially be limited to tweaking an inherently unsustainable economic system.

We further conclude that LCA can and should be used both to continuously seek more eco-efficient (understood in biophysical rather than monetary terms) means of production and consumption, as well as inform our management of the cumulative impacts of industrial metabolism with reference to well-defined limits for sustainable scale. Such an evolution in environmental management is not only feasible, but prerequisite to the long-term viability of a global industrial society.

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CHAPTER 7: OF LAWS AND LIMITS: AN ECOLOGICAL ECONOMIC PERSPECTIVE ON REDRESSING THE FAILURE OF CONTEMPORARY GLOBAL ENVIRONMENTAL GOVERNANCE

7.1 Publication Information

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7.2 Abstract

The persistent failures of international environmental governance initiatives to halt the degradation of the global commons are directly linked to the implicit worldview and assumptions fueling the proliferation of industrial society. These include an instrumental conception on non-human nature, technological optimism, and an expansionary economics premised on the axiomatic necessity of unconstrained growth. Permeating contemporary environmental governance regimes, it is argued that these premises are fundamentally incompatible with the requirements of environmental sustainability. Proceeding from the perspective of ecological economics, it is further argued that achieving environmental sustainability in industrial society requires foremost that we restructure and constrain the scale of economic activities relative to global biocapacity. It is concluded that a scale-based approach to governing the environmental commons, operationalized by a strong world environment organization, offers at least a partial solution to this conundrum.

7.3 Introduction

Contemporary industrial society consumes a continuous and accelerating throughput of material and energy resources (Ayres and Kneese 1994; Ayres and Ayres 2002; Fischer-Kowalski and Haberl 1997; Haberl 2006; Krausmann *et al.* 2008; Haberl *et al.* 2009). These resources are transformed, through the application of various technologies, into the diverse economic goods and services that fuel the metabolic processes of social production and reproduction, as well as a wide range of waste products. Thus, industrial society hinges on access to a material base that is both generous with regards to resource supplies and forgiving in terms of waste assimilation.

In large part, the genesis of industrial society can be attributed to a specific confluence of material and cultural phenomena: the advent of revolutionary industrial technologies (in particular, atomized factory-based mass production); European colonization (which opened up seemingly inexhaustible resource streams); and the birth of the fossil fuel economy (founded on a one-time, incredibly concentrated reserve of fossil solar energy). Of particular impact was the configuration of social relations and cultural mores that emerged during the 17th-18th centuries, which served to legitimize and mobilize the necessary materialist orientations to exploit these conditions. All of these substantially amplified human capacity to access, transform and accumulate resources (Hall and Klitgaard 2006). In the modern era, the continued growth and rapid proliferation of the industrial social model is much facilitated by the processes of neo-liberal economic globalization, which effectively seeks to mobilize global resources as fodder for industrial metabolism (Sklair 2002). However, the wide-spread and pernicious environmental decline associated with the rise of industrialism and its present forms raises pertinent questions with respect to its long-term viability as a model for global civilization.

Indeed, the emergence of international environmental politics as a major concern of the post-war era marks the waterline in the short history of industrial society where the sheer magnitude and cumulative effects of industrial activities have extended their environmental consequences from local to global scales (Held *et al.* 1999). The growing litany of pressing global environmental concerns, including acid rain, ozone depletion, biodiversity loss, climate change and reactive nitrogen accumulation, indicates that the metabolic throughput of industrial society has, for the first time in human history, begun to stretch the carrying capacity of the global biosphere.

Since the United Nations Scientific Conference on the Conservation and Utilization of Resources (UNSCCUR) was convened in 1949 as the first international effort to address the need for coordinated management of the global environmental commons, environmental diplomacy efforts have proliferated. At present, more than 30 UN agencies and programs are active in environmental governance, numerous high-level conferences have been convened to address a wide range of environmental issues, and in excess of 500 multilateral environmental agreements (MEAs) have been brokered in recent decades (Najam *et al.* 2006). Yet the deterioration of the global environmental commons has not abated appreciably but, rather, appears to be accelerating (MEA 2005). This contradiction underscores the necessity of re-evaluating the norms that underpin industrial society (Pelletier *in press*) and the legal structures/mechanisms that mediate its interactions with non-human nature. Specifically, it is worthwhile questioning the extent to which current governance regimes perpetuate and reinforce the environmentally destructive tendencies of industrial society, and to entertain possible alternatives. The ecological economic perspective on the necessity of constraining the scale of economic activity relative to biocapacity (Martinez-Alier 1993; Daly 1996; Costanza *et al.* 1997), which is seen as the first principle of sustainability, provides a promising departure point.

7.4 Discussion

7.4.1 Industrialism As A Social Model

Numerous researchers have contributed to a significant body of scholarship interpreting the philosophical roots of the dominant worldview espoused by industrial society, and of the modernist paradigm generally (Bermann 1981; Merchant 1990; Wilber 1998; Holder 2000; Capra 2001). Most of these point to the European Age of Reason and Enlightenment during the 17th and 18th centuries, and cite the contributions of intellectuals such as Descartes, Locke, Hobbes, Hume, Smith and Newton. Above all, this period was characterized by a commitment to the idea of progress in human material well-being and civility. At the same time, an emergent subject-object dualism contributed to the rise of a scientific empiricism based on the conceptual separation of human and non-human nature. In combination with Newtonian mechanics, which envisioned the natural world as a vast clock-works that could be understood in terms of its constituent elements, this reductionist perspective paved the way for a general belief in the power of humanity to create, improve, and reshape our environment through science and technology. Thus, the modernist world view reduced non-human nature to an object of exploitation to be manipulated and transformed in the interests of individual material enrichment and collective well-being (Bermann 1981; Merchant 1990).

Since laws and legal systems serve to institutionalize the values and worldview of the society in which they arise, we would anticipate that our systems of governance serve to perpetuate these same assumptions. Indeed, proceeding from Locke's prescription that a just and efficient social order requires enclosure of the commons and state-guaranteed property rights (Locke 1690), the legal tradition we have inherited very much concerns itself with the protection of private property and the right to accumulation. Similarly, if we consider the kind of economic system that western law legitimizes and maintains, we see that it is above all an expansionary system premised on an instrumental conception of non-human nature, technological optimism, and a belief in the possibility of unlimited economic growth (Daly and Farley 2004). It is further a system which reduces the calculus of well-being to individualist, instrumental rationality in place of societal, values-based discourse (Vatn 2005) and communitarian priorities (Pelletier *in press*).

Our environmental laws similarly reflect these assumptions (M'Gonigle 1986) – for example, through privileging managerial environmentalism (which requires expert scientific knowledge to manage nature as resources for human use) (Holder 2000); espousing the technological optimism that environmental crises can be resolved through the development of novel technologies; or in the contemporary conception of sustainable development, which constitutes an uneasy (if rarely challenged) marriage of the promotion of environmental integrity and greener, yet similarly unconstrained, economic growth. A pressing question is the extent to which these assumptions are actually compatible with the objective of environmental sustainability.

According to Kutting (2001), understanding the effectiveness of environmental agreements necessitates a social and critical approach that emphasizes the wider historical contexts in which environmental problems are produced, how they are defined, and the policy preferences that circumscribe the realm of politically feasible solutions entertained. In this light, both actors and policies are rooted in particular social and structural origins that strongly influence and delimit outcomes. Moreover, particular governance regimes invariably serve certain social groups over others, with agenda setting and potential outcomes determined by the most powerful actors and their vested interests (Paterson 1995; Luke 2005). Broader questions regarding the role of existing political, social and economic orders in generating environmental crises are generally precluded (Paterson 1995). An examination of a cross-section of international environmental governance tools is telling in this respect.

7.4.2 The Built-In Failure Of Contemporary Environmental Governance Regimes

Entering into force in 1979, the Long-Range Transboundary Air Pollution Convention (LRTAP) was a product of the 1972 UN Conference on the Human Environment, and represented the first multilateral environmental agreement on air pollution (LRTAP 1979). LRTAP was developed primarily to mitigate potential transboundary harms related to acid precipitation associated with sulphur and nitrous oxide emissions from fossil fuel combustion. Generally, LRTAP is considered to be an institutionally effective regime (Gehring 1994; Sprinz and Vaahtoranta 1994). However, as argued by Kutting (2001), the environmental goals related to this convention have always been subordinate to political and economic necessities. Thus, its environmental effectiveness is questionable.

The predominate approach to managing acid precipitation through LRTAP has been to reduce sulphur and nitrous oxide emissions via the development and application of technologies to sequester these compounds or their progenitors prior to or immediately proceeding fossil fuel combustion (Kutting 2001). LRTAP has served to coordinate this process by facilitating information sharing, monitoring of regional sectoral emissions, and negotiating emission reduction targets. However, this approach is problematic on several counts. First, emission reductions are only effective if they are sufficient to achieve environmental improvement (Kutting 2001). Second, the Convention targets specific sectors at the expense of ignoring other critical contributors such as the transport sector, which is responsible for a large share of nitrous oxide emissions (Kutting 2001). Here, economic considerations dictate problem-solving foci. Third, although the sequestration of acidifying products may ameliorate concerns of acid precipitation, the problem is simply shifted, becoming one of disposal. Finally, the actions stemming from this convention are inevitably limited to treating the symptoms of the much larger issue at stake – the root cause of acid rain, which is an expanding fossil fuel economy (Kutting 2001).

The short-comings of the LRTAP Convention speak more broadly to the limits of technological optimism and a managerial environmentalism constrained by political expediency. The implicit assumptions of industrial society which permeate such regimes provide intrinsic advantage to specific actors and ideologies. Economic reliance on fossil fuels in a system premised on growth means that questions of constraint are not entertained. Instead, politically feasible reduction targets achieved through problem shifting end-of-pipe technologies are favoured over precautionary solutions, fundamental changes in production systems, and hard limits on total allowable emissions (Kutting 2001).

The Kyoto Protocol provides a similar example of these same premises at work in, and ultimately limiting the potential of, a major international environmental governance regime. Again, the focus is on politically expedient rather than environmentally relevant reduction targets and technological solutions that do little to reduce fossil energy dependency. Even more insidious, however, is the fact that the Protocol actually serves to perpetuate expansionary and energy-intensive development through the Clean Development Mechanism (CDM).

The CDM, defined in Article 12 of the Kyoto Protocol, "provides for Annex I Parties to implement project activities that reduce emissions of non-Annex I Parties, in return for certified emission reductions (CERs). The CERs generated by such project activities can be used by Annex I Parties to help meet their emissions targets under the Kyoto Protocol" (UNFCCC 1997). Article 12 further stipulates that such projects are intended to assist developing countries in achieving sustainable development, in the interest of contributing to the ultimate objective of the UN Framework Convention on Climate Change. The CDM is thus intended to generate investment in developing countries, and promote the transfer of "environmentally-friendly" technologies (UNFCC 1997).

To date, however, it would appear that the CDM has been used primarily to reduce the costs of complying with Kyoto targets through a focus on projects that deliver high volumes of cheap credits, with little attention paid to their long-term contribution to greenhouse gas abatement. These are typically projects that capture or destroy gases with high global warming potentials at existing facilities in developing countries rather than investments in renewable energy technologies (Pearson 2007; Ellis *et al.* 2004). More important still is the unspoken assumption in the CDM process that facilitating greener industrialism in the developing world is, in fact, a positive social objective. A more cynical, or perhaps simply realistic perspective, is that the CDM actually serves to reproduce the very same model of industrial development that now plagues us with a suite of global environmental crises. Through the subsidization of (albeit less greenhouse gas intensive) industrial infrastructure, the development trajectories of non-Annex I countries are co-opted into western-style industrial development, thus precluding their potential for charting alternative, more ecologically-benign development pathways that are not dependent on unsustainable levels of material and energy through-put.

Perhaps most deserving of closer scrutiny is the concept of sustainable development itself. Prior to the 1972 UN Conference on the Human Environment, international environmental governance discourse had focused largely on environmental conservation through scientific management. However, the interests of the global south (in particular, resource-rich developing countries) gained increasing prominence in the years leading up to the Stockholm conference, and were accommodated through the emerging consensus that environmental problems were best solved through economic development (Irwin 2001). The focus of international environmental politics thus shifted from management to development, as reflected in the naming of subsequent UN conferences as "Conference on Environment and Development" and finally "Summit on Sustainable Development," essentially side-lining environmental concerns in favour of economic priorities (Irwin 2001). The term "sustainable development" was first catapulted to prominence in the 1987 Brundtland Report and later institutionalized in Agenda 21 and the Rio Declaration (UNDESA 1992).

Managing potential conflicts with organized social forces has perennially been an important role of international institutions in the liberal global order (Murphy 1995). Thus, environmental politics have been as much about formulating a consensus to secure the conditions for maintaining economic development and suppressing political challenges as about the need for environmental protection (Kutting 2001). Nowhere is this more evident than in the institutionalization of sustainable development as the organizing principle of contemporary global environmental diplomacy.

It is no surprise that the marriage of environmental problem solving and economic growth through "sustainable development" has been so readily embraced by government and private sector interests alike. Rather than posing a threat, such an approach to environmental diplomacy ensures the preservation of existing economic and political structures (Williams 2002, Luke 2005). Economic growth remains the central organizing principle, with environmental considerations reduced to a greening of the status quo through waste reduction, pollution prevention, and efficiency measures (Luke 2005). Assuming that humans are united in their want for ever more material goods and social services, and that answering these wants in an ecologically efficient manner constitutes the central objective for global society, sustainable development repositions the industrial model as the sole rational and desirable form of global social organization (Luke 2005). Thus the same tired premises of technological optimism, and profit maximization through unconstrained growth gain renewed legitimacy as contemporary environmental politics manifest a novel incarnation of a historical project rooted in the origins of modernism and Enlightenment thought (Marcuse 1964; Luke 2005).

7.4.3 Ecological Economics: A New Paradigm For Governing The Commons

The term economics derives from the Greek words oikos (house) and nomos (custom or law), and hence translates loosely as "rules of the house(hold)”. Since economies are characterized both by the physical flows of material and energy that underpin the provision of economic goods and services and the market interactions by which they are distributed through society (Daly and Farley 2004), the idea of the household does indeed provide a useful analogy. Here, the biophysical component might be equated with the actual structure of the house, while market interactions and relations represent the rules that circumscribe and direct the activities of those within. In this light, it is readily apparent that the natural environment provides the implicate order for all economic activity.

As a field of inquiry, economics is concerned with allocation of resources amongst competing activities, and how and to whom the costs and benefits accrue. It is therefore somewhat odd to note that the contemporary discipline of economics, which we call “neoclassical economics,” is largely a social science enterprise – focusing entirely on the political economic aspect of exchange (the rules) at the expense of considering the underlying physical inputs and outputs (the structure) (Hall and Klitgaard 2006). Instead, premised on the ideal of *Homo economicus* – a vision of humans as rational, self-interested, and utility-maximizing economic actors whose cumulative choices (given the availability of perfect information and a suite of other questionable assumptions) collectively discipline an efficient marketplace – neoclassical economics effectively divorces the conception of economic activity from its biophysical environmental context (Daly and Farley 2004). For this reason, market prices typically reflect human preferences and other influences, but do not capture the suite of environmental costs associated with their provision.

In contrast to contemporary neoclassical economics, ecological economics is expressly concerned with understanding the material/energetic aspect of economic activity (Costanza *et al.* 1997; Daly and Farley 2004; Hall and Klitgaard 2006). To this end, ecological economics attempts to reframe economic activities in environmentally relevant currencies – for example, by representing economic flows in terms of material and energy intensities, and associated emissions of wastes such as greenhouse gases, ozone depleting compounds or acid precipitants. The purpose is to map the relationships between specific kinds and quantities of economic activity, and the capacity of the biophysical environment to furnish the requisite material/energy resources, absorb the resultant wastes, and respond to the attendant changes over variable spatial and temporal horizons (Martinez-Alier 1993; Daly 1996; Costanza *et al.* 1997).

The premises of ecological economics are actually quite simple, and may be reduced to three central biophysical principles which govern all self-organizing systems. The first is a recognition of the limits imposed by the Laws of Thermodynamics. According to the First Law, energy can neither be created nor destroyed, but only changed from one form to another. Thus, the amount of energy in any closed system (indeed, in the universe as a whole) is fixed. Furthermore, all such transformations inevitably degrade concentrated, usable forms of energy into unusable, high entropy forms (Second Law). Closed systems therefore tend towards maximum entropy, or thermodynamic equilibrium.

The second principle is that *every* economic activity has a material/energy correlate. These are the biophysical flows of resources that underpin economic goods and services. Following the Laws of Thermodynamics, then, all economic activities deplete the available stock of usable material/energy and result in the production of high entropy waste (Ayres and Kneese 1969; Daly and Farley 2004). All else being equal, increasing economic activity equals increasing entropy. Of course, the resource and emissions intensity per unit economic good or service provided will vary widely, and there exist substantial opportunities for efficiency gains. However, excessive faith has been placed in the potential “decoupling” of growth and throughput to mitigate environmental concerns whilst allowing for continued growth. To date, empirical evidence does not support this position (for example, see Binswanger 2001), and even the vaunted dematerialization of the "information economy" has proven illusive (for example, see Williams *et al.* 2002). For this reason, it is imperative to remain cognizant that any relative gains through efficiency measures will be outweighed by absolute throughput so long as growth is unconstrained (Binswanger 2001).

The third principle is that, as a closed system save for discrete inputs of solar energy from the sun and gravitational energy (in the form of tides) from the moon, the earth has an inherently finite capacity to supply energy and material resources as fodder for economic activity and to absorb the associated entropic wastes produced (Georgescu-Roegen 1971; Odum 1971; Smil 1991; Cleveland and Ruth 1997). Certainly, the total flux of solar energy is enormous relative to current human energy needs. However, its distribution is spatially diffuse. While technologies for capturing this energy continue to improve (for example, Kubiszewski and Cleveland [2009] report a current energy return on investment ratio from solar cells of 6.56), as of 2004, installed photovoltaic capacity accounted for less than 1% of energy supplies (Kubiszewski and Cleveland 2009). Material resources and waste sink capacities are, in contrast, inherently limited.

Simply put, the implication of the ecological economic perspective is that achieving environmental sustainability in human societies necessarily requires strict attention to these limitations. Sustainable scale is the first condition of sustainability (Daly and Farley 2004). Effective governance of the global environmental commons must thus proceed from a recognition of the scale of resource extraction and waste production that natural systems can accommodate in perpetuity (i.e. without degrading the integrity of host ecosystems and biogeochemical cycles). The ecological economic perspective thus presents a profound challenge to the organizational premises of industrial society and its systems of governance. A society premised on unconstrained growth but dependent on a finite biophysical base is inevitably doomed to undermine the very foundations on which it is built. Moreover, such a society will likely face mounting conflicts associated with scarce resources and environmental decline.

The rapid rise and proliferation of industrial society was, in large part, made possible by the abundance of available resources relative to population, consumption levels and technology. However, the inevitably entropic nature of the economic process has degraded natural resources and polluted the environment to a commensurate extent, creating the twin challenges of dwindling resource supplies and rising environmental costs. In a post-peak oil world, the ramifications of energy insecurity are only now beginning to be felt, and will certainly present profound implications in the coming decades (Campbell 2002). With respect to the exploitation of biotic and other renewable resources, we are similarly faced with constraints. Human appropriation of global terrestrial net primary productivity (NPP), a measure of biotic resource availability, has been estimated at 23% (Haberl *et al.* 2007), and an estimate of 8% has been advanced for the appropriation of marine NPP by fisheries (Pauly and Christensen 1995). As human populations continue to grow and levels of consumption increase, this appropriation places increasing strain on ecosystems and biodiversity.

Much of the historical debate between technological optimists and proponents of the eco-scarcity thesis has focused on resources. Eco-scarcity arguments, from Malthus (1798) onwards, posit that population growth will inevitably outstrip resource availability, hence leading to resource depletion, environmental degradation, and population collapse. Such arguments, although true in an absolute sense, have been roundly criticized in various contexts for simplifying what are often political problems of resource distribution rather than abundance (Robbins 2006). In contrast, technological optimists have long held that human ingenuity will always lead to novel substitutes for increasingly scarce resources, and that free markets will ensure that, as specific resources become depleted, rising prices will prevent their over-exploitation (Myers and Simon 1994). Besides the facts that the latter argument has proven historically questionable (for example, see Basagio 1994) and that the depletion of biotic resources (the majority of which are not marketable goods) does not necessarily follow a linear trajectory (Vatn and Bromley 1994), both of these arguments miss an important point. Many of the most critical limits that we are encountering at present are not limits of resource availability. Rather, they are limits of waste sink capacity.

In fact, the history of environmental politics in the post-war era might be best interpreted as a series of reactions and attempted accommodations as industrial society has successively generated sufficient high entropy wastes of particular kinds to overshoot the limits of sustainable scale on a variety of fronts. Generally, these limits represent the stability domains of specific biogeochemical cycles – for example, ecosystem acidification resulting from excessive precipitation of sulphur and nitrous oxides (which represent an entropic by-product of fossil fuel combustion); the depletion of stratospheric ozone due to the accumulation of chlorofluorocarbons and related compounds (entropic wastes of various industrial processes, most notably refrigeration); or increased radiative forcing as a result of excessive greenhouse gas emissions (the entropic by-products of a host of activities, including fossil fuel combustion, ruminant metabolism, and synthetic nitrogen fertilizer production and use in industrial agriculture, or deforestation). All of these problems are essentially issues of excessive increases in high-entropy waste, and can be resolved only by constraining the scale of contributing activities.

Moreover, contrary to techno-optimist assumptions, such entropic problems (by their very nature) cannot be simply reversed by technological means, since the energy and material throughput required would produce still greater amounts of high entropy waste. Similarly, unless the scale of alternative technologies is constrained, their application will result in problem shifting, thus contributing to new crises of excessive waste. Certainly, great potential exists for dramatically improving the ecological efficiency of numerous industrial activities. However, so long as growth is unconstrained, efficiency measures will, at best, buy time. These challenges are further compounded by growing scientific evidence that individual threats to ecological resources and media are often interconnected in an ultimately integrated biosphere where they synergize, and create emergent properties and positive feedback effects with unpredictable consequences (Millennium Ecosystem Assessment 2005).

It is therefore useful to consider what the insights of ecological economics imply for the previously examined environmental governance regimes, and for environmental governance generally. Both LRTAP and Kyoto are doomed to fail in their specific mandates unless they are revised to constrain emissions with respect to the limits of sustainable scale. This requires elucidating the scale of emissions that natural systems can accommodate in perpetuity (regionally with respect to LRTAP and globally for Kyoto), and restructuring economic activities accordingly. It also demands that proposed technological solutions, such as sulphur and nitrous oxide sequestration, deep-well carbon injection, or alternative energy developments be carefully scrutinized with respect to problem-shifting and long-term sink capacities. Technological "solutions" will invariably constitute a double-edged sword.

The implications for sustainable development are equally profound. The conventional notion of sustainability sees sustainable development as a balancing of equally important social, economic and environmental considerations. This implies that each sphere is independent, and whole onto itself. In contrast, the ecological economic perspective reveals that, as the implicate order for human activity, environmental sustainability is prerequisite to sustainability in any other sphere. The first consideration of sustainability must therefore be to constrain human activities with respective to the absolute limits imposed by the finite nature of our biospheric environmental context. The ultimate solutions to the environmental crises generated by industrial economic development will therefore not be found in more status quo industrial development, or even greener industrial development. Rather, they will be found in alternative modes of human organization that transcend the pathological premises of a growth-fixated industrial society.

7.4.4 Operationalizing Limits-Based Governance: The Argument For A World Environment Organization

Any serious discussion of alternative approaches to global environmental governance must necessarily address the issue of operationalization. What might be the most appropriate and/or effective organizational structure to oversee a scale-based approach to governing the global environmental commons? To answer this question, it may be constructive to begin by illustrating why the status quo of global environmental governance is inadequate to this task. This requires scrutiny of the predominantly UN-administered structure of international environmental governance that has evolved during the post-war era.

The United Nations (UN) was established in 1945 to promote cooperation in international law, international security, economic development, social progress and human rights issues. At the time, environmental concerns had not achieved significant international profile. For this reason the UN Charter, while explicitly committing to the economic and social advancement of all peoples, contains no reference to the environment whatsoever.

The increasing prominence of environmental concerns on the world stage in the decades following its establishment prompted various bodies within the UN system to administer their own environmental programs, as well as the convening of decadal UN conferences to facilitate international dialogue and cooperation in responding to emerging challenges of global environmental degradation (Najam *et al.* 2006). One of the outcomes of the 1972 UN Conference on the Human Environment was the establishment of the United Nations Environment Programme (UNEP), with a mandate to coordinate the development of environmental policy consensus by reviewing the status of the global environment and bringing emerging issues to the attention of governments and the international community. However, UNEP was not granted authority of its own to develop international environmental policy instruments. Moreover, with its low status within the UN bureaucracy and its detachment from many of the existing multilateral environmental agreements, it has generally been perceived as unable to meet the pressing needs of global environmental governance (Charnovitz 2005).

In the absence of a strong core, contemporary environmental governance is hence orchestrated by a diverse range of UN bodies and UN-sponsored conferences, and proceeds via the progressive development of multilateral environmental treaties based on the collective bargaining of sovereign states on a single-issue basis. No strong mechanism exists to identify the most serious gaps in environmental management, nor to determine where new environmental investments are most appropriate and coordinate effective responses (Charnovitz 2005). As a result, environmental politics are reactive rather than proactive – perennially playing catch-up with an ever-accelerating rate of environmental change and the acute crises that manifest. This ad-hoc approach is unlikely to provide adequate recourse to the increasingly complex, challenging, and pressing demands of global environmental governance.

7.4.5 Why The Status Quo Is Insufficient

First, relying on the good will of sovereign states represented by politicians whose terms of engagement are dictated by short-term political expediency can hardly be expected to result in sound environmental agreements. In the words of historian JR McNeil (2000, pp. 337), "the overarching priority of economic growth was easily the most important idea of the twentieth century." Certainly, this is evidenced, whether in political debate or media reporting, by the holy status of measures of economic performance such as Gross Domestic Product. As a result, few politicians are prepared to voluntarily embrace restrictions that will significantly constrain their country's capacity for economic growth and, by association, their own political viability. An observation of the track record of the United States in cooperating with global environmental governance regimes is telling in this respect. It does not help that the paramountcy of state sovereignty is built into the UN system and enshrined in virtually all multilateral environmental governance regimes, legitimizing the right of nation states to opt-out of MEAs in favor of domestic priorities (Najam *et al.* 2006). Moreover, so long as domestic priorities are framed in the individualist, instrumentally rational terms of conventional economics, the side-lining of global societal values and governance discourse will be systematically deprioritized.

The subordination of global environmental policy to economic concerns is further evidenced in the power disparities between environmental and trade regimes. With economic globalization have come powerful institutions of global economic governance, most notably the World Trade Organization (WTO), the International Monetary Fund (IMF) and the World Bank. In the absence of comparable global environmental institutions, the result is an ever-growing gap between the authority that economic and environmental concerns demand on the world stage (Kirton 2005). In fact, environmental treaty negotiations are now regularly monitored to ensure that they do not contravene WTO rules, creating an adversarial stance that has "chilled" environmental policy making across levels of governance (McCarthy and Prudham 2004). It is deeply ironic that, in the enabling of a strong WTO, governments acknowledge the fact of global economic interdependence and the necessity of effective global economic governance while simultaneously failing to draw a parallel conclusion about environmental interdependence and governance (Charnovitz 2005).

The second evident weakness of interstate diplomacy in the environmental arena is that the soft law multilateral environmental agreements that result from the collective bargaining of nation states often cater to the lowest common denominator. In the interest of securing consensus and initiating processes, collective decisions are unduly influenced by the self-interested demands of single states. The resultant MEAs also typically lack effective enforcement and compliance mechanisms. Moreover, the expectation that soft law agreements and principles will eventually be transformed into customary international law (Palmer 1992; Birnie and Boyle 2002) presents a somewhat vacuous hope if the agreements and principles themselves are lacking in merit. Clearly, there is a pressing need for environmental governance structures that can both produce environmentally effective tools and ensure compliance.

The third, and perhaps most compelling, argument against the efficacy of current global environmental governance structures relates to the sheer rate and magnitude of environmental change that we face and the rapidity and authority of the response that is required. On both counts, these supersede the capacity of status quo environmental governance measures. At current levels of production and consumption, industrial society is already overshooting multiple limits of sustainable scale (Rockstrom *et al.* 2009). Yet, as economic globalization increases the metabolic throughput of existing industrialized societies and facilitates the growth of industrialism in developing countries, both the relative and absolute proportions of consumption and waste generation are changing quickly (Haberl 2006). Per capita consumption is increasing, and the global population may double before leveling out towards the end of the century (FAO 2006). The massive increase in resource throughput and entropic waste production that will result during the coming decades may be orders of magnitude higher than anything historically experienced. Coupled with the sluggish nature of interstate diplomacy, the weaknesses of the resultant environmental governance regimes, and the lack of a strong, coordinating body, the prospects for environmental integrity and, by default, human societies, are bleak. An alternative approach to global environmental governance is long overdue and, indeed, may be necessary to prevent widespread conflict between classes, nation states and regions which will likely accompany continued environmental decline.

7.4.5 Alternative Approaches To Global Environmental Governance

Numerous suggestions for alternative environmental governance regimes have been advanced. Among the more modest proposals are those that call for increased coordination in international environmental governance through a clustering of existing MEAs on an issue area basis (von Moltke 2005). Slightly more ambitious are the suggestions of upgrading UNEP to the status of a UN agency to act as an umbrella coordinator of existing and future MEAs (Biermann 2001). More radical are those that propose a strong, centralized environmental organization with broad authority to legislate and enforce global environmental law (Esty 1994a, b; Gupta 2005). Such proposals almost exclusively envision an environmental governance regime situated within the United Nations hierarchy.

It is clear that simply clustering existing environmental agreements, whether independently or under the umbrella coordination of a stronger UNEP, would be inadequate to achieving an effective scale-based approach to environmental governance, or even offer substantial improvement over the status quo. Certainly, such institutional rearranging might result in more cost-effective and efficient execution of existing agreements (Najam *et al.* 2006). However, since most (if not all) existing MEAs were not formulated with attention to absolute scale limits, and embody the same pathological premises previously discussed in relation to LRTAP, the Kyoto Protocol, and the concept of sustainable development generally, simply shuffling them into novel organizational hierarchies would be, at best, a symbolic act. Although improvements may be possible on a case-by-case basis, this is unlikely to achieve significant progress under status-quo negotiating processes. Particularly salient is that such modifications would be doomed to fail so long as state sovereignty is guaranteed. Moreover, following the Principle of Subsidiarity, governance institutions should be commensurate with the scale of issues to be governed (Daly and Farley 2004). Existing MEAs may be international, but they are not global. The sum of their parts will not result in an effective global governance apparatus. A decidedly more radical approach must be considered. Of particular interest is the extent to which a strong, centralized World Environment Organization (WEO) might better address the need for a global ecological political economy.

The earliest proposals for a global environmental governance body are attributed to George Kennan (1970), a US foreign policy strategist who in the early 1970's advanced the idea of an "International Environmental Agency" comprised of representatives from a limited number of developed countries. In response to the ensuing debate, the international community established the United Nations Environment Programme in 1972. However, as discussed previously, UNEP has fallen far short of this ideal. Since that time, the debate over the need for an international environmental governance body and the appropriate structure for such a body has continued (Biermann and Bauer 2005). In 1992, Lord Geoffrey Palmer, then Prime Minister of New Zealand, renewed calls in the United Nations for the creation of an international environmental agency with real power and authority to legislate international environmental law within the UN system (Palmer 1992). More recently France, Germany, Brazil, Singapore and South Africa led an unsuccessful push for a world environment organization at the 2002 World Summit for Sustainable Development (Chirac 2003). In addition, numerous researchers have contributed proposed models for a WEO (Biermann and Bauer 2005).

A salient critique of all such models is that, without a modification of existing decision-making procedures, a WEO would be neither strong nor effective (Najam 2005; Oberthur and Gehring 2005). The formulation of an organizational structure for a WEO must therefore pay close attention to assuring adequate decision-making powers. In large part, current environmental diplomacy is hamstrung by the narrow interests of sovereign states, and the guarantee of sovereignty enshrined in the UN system and written into many MEAs. Since the most serious contemporary environmental problems are trans-border and/or global, their resolution requires global-level decision making that can effectively trump the opposition of recalcitrant states. As other authors have noted, we already have at our disposal working models (represented by specific aspects of existing regimes) for a potential decision-making architecture of a WEO. For example, in both the Montreal Protocol and the Global Environment Facility, decisions are passed by a simple majority of both developing and developed countries (Biermann 2005). Such an approach avoids the trend towards lowest-common-denominator agreements where consensus is required, and overcomes issues of undue influence by either developed or developing countries. Articles 21 and 22 of the World Health Organization constitution make majority decisions binding on all members (WHO 2008). Certainly, the selective-exit option for environmental governance regimes must be firmly closed to prevent the largest polluters from shirking on global responsibilities, and constrain the potential for free-riding, generally. In addition, the decision-making powers of several regimes such as the Montreal Protocol and the Convention on Trade in Endangered Species (CITES), are far-reaching and include the delegation of decision-making authority to competent bodies (DeSombre and Kauffman 1996; Biermann 1997; Biermann 2005). This consideration may be of particular importance to ensuring that decisions are made according to the best available information rather than based on political expediency.

The problem of free-riding is particularly acute for global environmental governance since the countries refusing to cooperate in environmental regimes cannot be excluded from the benefits of common property resources such as a stable climate or intact ozone layer. It is therefore necessary that a World Environment Organization has broad powers to both legislate and enforce compliance of limits-based constraints on economic activities that contribute to global-scale environmental degradation. In other words, in keeping with the ecological economic perspective, the regulation of economic activity for other objectives should be subservient to scale-oriented environmental regulation in the hierarchy of global environmental governance. However, environmental governance could certainly be facilitated through developing direct linkages with existing economic governance regimes. In particular, the WTO provides an excellent model for enforcing compliance of international agreements through trade-based sanctions and penalties (Campbell 2004; Charnovitz 2005). Making WTO membership contingent on membership in a WEO, and compliance with WEO regulations enforceable through trade sanctions and penalties would provide a powerful means of ensuring effective global environmental governance.

A second challenge facing a WEO employing a scale-based approach to environmental governance is that of equity. Following Daly and Farley (2004), once appropriate scale has been established, the subsequent priority is to ensure that distribution corresponds to a shared conception of distributive justice. Only then can resources be allocated with respect to societal objectives. In other words, in a world of limited resources and waste sinks, how should it be determined who gets what share of the pie within the constraints of sustainable scale? Certainly, this is manifest in the current struggle to define a model to replace the Kyoto Protocol when it expires in 2012.

A rich discussion of the ethical dimensions of global resource distribution is beyond the scope of the current analysis. It should be noted, however, that the currently prevalent neoclassical economic model focuses on individual, instrumentally rational decision making with respect to market-mediated preferences for rival/excludable goods in situations where transaction costs are low. In contrast, ecological economics and the governance institutions it informs must also grapple with community values in allocating (usually non-marketable) public goods where transaction costs are high (Norton 2005; Vatn 2005; Pelletier *in press*). This requires attention to concerns regarding intra- and intergenerational distributive justice for both human and non-human natural communities (Pelletier *in press*). Strict reliance on the market mechanism is therefore insufficient.

Within this context, however, there certainly may be situations where the market can be leveraged towards achieving specific objectives. For example, in some situations, one potentially feasible option that could drive developed countries towards rapid emissions reductions while simultaneously facilitating the transfer of funds and capacities necessary for developing countries to chart environmentally viable development pathways would be to apportion tradable per capita quotas, adjusted to reflect historical contributions, to all nation states. Kyoto made partial headway towards such an approach through the Clean Development Mechanism. However, the overarching requirement here would be that the total quotas reflect a scale of greenhouse gas emissions deemed to be well within the buffering capacity of the global carbon cycle – in other words, absolute as opposed to relative targets. Applying this same approach where possible across issue areas would help prevent problem shifting, and contribute to more radical reformulations of potential sustainable development trajectories. This approach would also provide incentive for developing countries to participate in a WEO, whilst tempering the power asymmetries that have long-plagued international environmental diplomacy.

The critical difference between this approach and previous uses of economic incentives to achieve environmentally desirable outcomes is that it would proceed from the dictates of biophysical limits to sustainable economic activity. Within this context, a combination of market and non-market instruments would be necessary to allocate rights to resources. The resultant constraining effect on key activities such as fossil fuel use would simultaneously mitigate numerous environmental issues such as climate change, acid precipitation, reactive nitrogen accumulation and, potentially, numerous other as-of-yet unrecognized problems. Clearly, however, the capacity of market mechanisms to mediate the pursuit of global societal objectives is limited and should hence be considered only one among a suite of potential tools.

Critics of the idea of a strong world environment organization frequently raise two central concerns. The first is that the environment is far too complex to be managed by one organization (Biermann and Bauer 2005). In contrast, it might be argued that the current system of environmental governance is far too unwieldy and uncoordinated to manage the complexity of human interactions with the environment. A strong WEO, with a scale-based governance mandate, would be much more effective than the status quo in proactively managing human/nature interactions, since most environmental problems are fundamentally the result of the inappropriate scale of human activities. The second concern is that incremental improvements in current governance will have to be adequate because the idea of a strong WEO is grossly utopian (Oberthur and Gehring 2005). Once again, such an argument must be turned firmly on its head. What is grossly utopian is the expectation that humanity can proceed on its current trajectory of global industrialization, with a system of global environmental governance that is hopelessly inadequate to the task. The language of necessity must replace that of political expediency.

Certainly, a WEO would not solve all of the problems of human/nature interactions in industrial society. As has been argued previously, the root of these problems is inherent in the very fabric of our worldview and embodied in our social institutions. In addition, the complexity of environmental systems, our lack of knowledge regarding thresholds, non-linearities and synergies, and our tradition of reactive instead of predictive and proactive management underscores the weaknesses of contemporary managerial environmentalism – much of which would be replicated in a scale-based governance approach. Moreover, the ecological economic approach to governance may itself be fraught with challenges of social legitimacy. It should also be noted that population, which is a key determining factor in anthropogenic environmental impact, has not been considered here, although this could potentially also be an appropriate domain for WEO intervention. What is certain is that a strong WEO, with a scale-based mandate and appropriate mechanisms for ensuring effective coordination, regulation and compliance, could provide the necessary means of reining in the juggernaut of industrialism and, in the least, provide us with the grace necessary to contemplate alternative modes of human organization.

7.5 Conclusions

The majority of species that have contributed to the long march of biological evolution are no longer with us. Similarly, the history of humanity is littered with examples of civilizations that arose within the context of specific social and environmental circumstances, flourished briefly, and then disappeared (Diamond 2004). In short, societies have been variably successful in perpetuating specific configurations of social relations and human/nature interactions within the constraints of particular social and environmental conditions. A civilization's viability over the long-term is, in large part, related to either the stability of these conditions, or its capacity to adapt as conditions change.

The Enlightenment period was a time of remarkable change for western society. Out of this tumult coalesced a new vision of humanity's rightful place in the world: a vision premised on the paramountcy of the individual pursuit of material well-being; a belief in human capacity to control and reshape, through the application of science and technology, an objectified natural environment over which we enjoyed unquestioned dominion; and the certainty that unlimited material progress was desirable, possible, and even necessary for the well-being of society as a whole. But these same premises that fueled the rise of industrialism, while tenable in a context of limited population size, nascent technological prowess and seemingly inexhaustible resources have now become pathological in the context of exponential population growth, powerful and pervasive technologies and a level of material and energy consumption and waste production that exceeds the capacities of our inherently finite biophysical environment to support. As we begin to encounter one biophysical limit to the scale of industrial activity after another in the form of emergent environmental crises, we are forced to recognize that industrial society, in its present manifestation, is no longer a viable mode of social organization for a globalizing society. Moreover, since these same premises imbue our social institutions, including the legal mechanisms we have advanced to mediate human/nature relations, we must rethink our entire approach to global environmental governance.

The insights of ecological economics, founded on a recognition of the implications of the Laws of Thermodynamics for human organization, point towards a partial recourse to the pathologies of industrial society and clear direction for a more effective form of environmental governance. Although this perspective does not overturn the spectrum of problematic assumptions inherent in the modernist enterprise, such as the glorification of materialism, and the conception of non-human nature as mere means to human ends, it does effectively constrain their most environmentally pernicious potentials by challenging the concept of unconstrained growth. Moreover, it makes explicit the recognition that sustainability is a global, community concern that transcends the capacity of market-mediated instrumental rationality to provide. It also reveals that industrial society can only be assured of long-term viability if we restructure our economic activities with respect to the absolute biophysical limits to sustainability inherent in a finite environment. This requires legal mechanisms explicitly intended to partition limited resources among the mutually constituting human and non-human natural communities in which we have our existence – both within and between generations – according to a shared conception of distributive justice and with foremost attention to sustainable scale.

The operationalization of this limits-based approach to global environmental governance further requires radical restructuring of the current environmental governance apparatus. This could potentially manifest as a strong World Environment Organization with the authority to legislate and enforce compliance of limits on the scale of material and energy throughput as well as specific forms of waste production. If properly formulated, such an approach to governance could simultaneously temper the most environmentally destructive tendencies of industrial society while facilitating a more equitable distribution of global resources and truly sustainable development opportunities for society as a whole.

As reasoning beings, we are uniquely situated to understand and respond to the changing nature of our social and environmental conditions, and hence take responsibility for our own evolution, both as individuals and as a society. As moral beings we are provided with a compass that points us towards the fulfillment of those ends that make us human. Industrial society is but one of myriad potential forms of social organization, and we can choose – both in the immediate and long-terms – those aspects of our current civilization that we wish to retain and those that are outmoded. The unraveling of the biotic fabric represented by the growing environmental crises of the modern era informs us that industrial society must reinvent itself. To advocate an ecological political economy founded on scale-based environmental governance is therefore by no means a utopian vision, nor is it even an unrealistic vision. Rather it is a pragmatic vision of the evolutionary path that industrial society must necessarily follow – not towards a certain future, but in order to secure the possibility of choosing amongst worlds that might be.

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CHAPTER 8: INDUSTRIAL SOCIETY, FOOD PRODUCTION AND ANTHROPOGENIC ENVIRONMENTAL CHANGE

8.1 Industrial Society And Global Environmental Degradation

The modern era has seen the emergence of a growing list of pressing global environmental concerns. This includes increased radiative forcing of the atmosphere due to large-scale greenhouse gas emissions (Hughes 2000; Robertson *et al.* 2000; Levitus *et al.* 2001; Walther *et al.* 2002); depletion of the stratospheric ozone layer associated with emissions of CFCs and other chemicals (Crutzen 1992; Madronich *et al.* 1995); acid precipitation linked to the combustion of fossil fuels (Likens *et al.* 1996; Bouwman *et al.* 2002); the biomagnification of persistent organic pollutants across trophic levels and their migration through ecosystems (Vallack *et al.* 1998); the doubling of biologically available reactive nitrogen in terrestrial ecosystems since 1960, and subsequent nitrogen cascades through ecosystems (Vitousek *et al.* 1997; Smil 1999; Galloway *et al.* 2003, 2004, 2008); the serial depletion of global fish stocks (Pauly *et al.* 1998); biodiversity loss at orders of magnitude higher than historical extinction rates (Nee and May 1997; Vitousek *et al.* 1997; Pimm and Raven 2000); and human appropriation of global terrestrial biological resources equivalent to an estimated 24-40% of total net primary productivity (Vitousek *et al.* 1986, Haberl 2006, Haberl *et al.* 2007). Although skeptics persist (for example, see Myers and Simon 1994), few would question that the economic metabolism of industrial society has become sufficient to upset the balance of planetary biogeochemical cycles and ecosystem integrity at multiple scales.

What is increasingly clear is that a central imperative of global governance must be the management of anthropogenic perturbations of these critical biogeochemical cycles and the maintenance of ecosystem integrity. This requires foremost that the aggregate material/energy flows and emissions associated with economic activity be constrained relative to carrying capacity (Daly 1991; Tilman *et al.* 2001; Daly and Farley 2004; Fujimori and Matsuoka 2007). Instrumental to this task are careful measures of impact potential per unit economic good or service produced to facilitate eco-efficiency improvements and the identification and promotion of least-environmental cost activities.

Given the direct linkages between ecosystem integrity and human well-being, efforts to quantify the scale of anthropogenic influence on global ecosystems, and to understand how these change over time, are of increasing relevance. The first widely publicized attempt to model human impacts on global ecosystems and to project these into the future was the World 3 model published by Meadows *et al.* in 1972. The World 3 model was a computer simulation produced and used by a Club of Rome study. World 3 simulated interactions between population, industrial growth, food production, and ecosystem limits. The model suggested that serious global ecological problems would become evident by 2030-40, leading to collapse by 2070. In the 1992 book “Beyond the Limits” as well as a 2004 update, the authors report a revised version of the model which reconfirms earlier predictions. Critics have pointed to several weaknesses in the model, including that it does not allow for substitutes and that it aggregates physically disparate processes in simplistic equations (Smil 2003).

Behrens *et al.* (2007) analyzed global material flows over time (1980-2002) to determine cumulative scale and trends in resource extraction between regions driven by global economic integration. Although overall material intensity per unit economic value decreased by 25%, suggesting a decoupling of resource use from economic growth, absolute annual extraction increased by 30%, demonstrating that increasing scale continues to outweigh both efficiency (technology) and structural (increased contribution of the service sector to GDP) effects. It is unclear whether the authors distinguished between real and virtual GDP. This is an important distinction from an ecological economic perspective, since any real growth must inevitably be underpinned by material/energy flows. Inclusion of virtual GDP would greatly reduce the apparent material consumption of the global economy. Regardless, this analysis provides an important illustration of the global distribution of environmental pressures associated with raw material extraction. An interesting extension, not undertaken by the authors, would be to reanalyze these flows from a consumption-based perspective of environmental responsibility.

Various attempts have also been made to quantify the scale of human appropriation of renewable resources relative to biocapacity. For example, Vitousek *et al.* (1986) estimated the total terrestrial net primary productivity (the net amount of solar energy converted to plant organic matter through photosynthesis) appropriated for human consumption at 40%. Recently, more refined estimations produced by Haberl and colleagues (2007) suggest a figure closer to 24% of global terrestrial NPP, and predict that this is likely to surpass 50% by 2050. In a similar fashion, Pauly and Christensen (1995) calculated that shellfish and fisheries production appropriated 8% of global aquatic primary production. It is difficult to predict the precise implications of increasing NPP appropriation. However, as pointed out by Imhoff *et al.* (2004), this is a remarkable level of appropriation for a species representing only 0.5% of planetary heterotroph biomass. It also has notable consequences for energy flows within food webs, the biodiversity that ecosystems can support, the composition of the atmosphere, and the provision of important ecosystem services (Imhoff *et al.* 2004).

Also of note are the global models of anthropogenic greenhouse gas emissions produced by the Intergovernmental Panel on Climate Change (IPCC 2007). These models quantify current emissions and scenario-model likely emissions under different development trajectories in order to predict the reduction targets necessary to maintain atmospheric concentrations of key greenhouse gases within specified bounds. These models are among the few robust measures of sustainable scale that have been advanced to date. As such, they may be very useful as a reference point for normalizing emissions in specific sectors. This subsequently serves as a basis for societal prioritization in the economic restructuring that will be necessary to constrain anthropogenic emissions.

Fujimori and Matsuoka (2007) used material flow analysis to calculate total anthropogenic mobilization of carbon, nitrogen and phosphorus flows in 2001. It was estimated that 13,395 Tg of carbon, 248 Tg of nitrogen, and 29.2 Tg of phosphorus were appropriated for human use. Although not speaking directly to environmental effects, these estimates also serve as useful normalization points and benchmarks.

Most recently, the World Energy Outlook 2008 was published by the International Energy Agency (IEA 2008). According to its Reference Scenario (no new government policies), world primary energy demand is projected to increase 45%, growing 1.6% per year between 2006 and 2030. Based on current trends, it is further estimated that energy-related CO_2 emissions will increase 45 % from 2006 to 2030. Achieving atmospheric concentrations of 550 ppm of CO_2-equivalent would require emissions to rise to no more than 33 Gt in 2030 for a mean temperature rise of 3 degrees. The challenge of limiting concentrations to 350 ppm – with an attendant 2^o C temperature rise – is significant, requiring that world energy-related CO_2 emissions decrease substantially from 2020 onwards, falling well below 26 Gt in 2030. According to the report, this would require that all major emitters undertake concerted actions. Moreover, OECD countries alone would be unable to chart a global 450-ppm trajectory, even if their emissions were to fall to zero (IEA 2008).

8.2 Food Production: Consuming The Planet?

Given the scope of anthropogenic environmental change and the attendant threats to human welfare and ecosystem integrity as a whole, economic restructuring founded on coupled eco-efficiency/sustainable scale considerations and societal sustainability objectives is imperative. This requires careful analysis of production efficiencies within and between sectors relative to sustainable scale targets. Such information will facilitate the identification of key leverage points for regulatory and market intervention, and provide hard estimates of potential gains. It will also provide a basis for explicit recognition of the intra/intergenerational and interspecies distributive implications of current and potential future economic configurations. Such considerations must be central to operationalizing a shared vision of societal sustainability objectives.

Thomas Malthus was the first to call attention to the relationship between food production and carrying capacity (Malthus 1798). According to Malthus, since food production increases arithmetically whereas population has the potential to increase exponentially, overshoot and collapse are inevitable. Although Malthus's dire predictions failed to materialize in nineteenth century England, his theories have nonetheless provided impetus for an intense interest in the role of food production in environmental change and the capacity of the biosphere to support the consumption patterns of increasing populations.

Over the last half century, the application of new technologies in agriculture has spurred a dramatic increase in world food production, more than doubling the volume of cereals produced and precipitating a price decline exceeding seventy percent in real terms (Wolf 1986, Tilman *et al.* 2001). For the poor, who devote proportionately more of their incomes to food than the wealthy, this transformation has been of profound consequence. Yet the so-called "Green Revolution" has not come without costs. Agriculture has been implicated in a broad range of direct, ecological impacts. At the same time, the finite nature of fossil fuel reserves as well as the environmental impacts of extracting and consuming the fossil energy that underpins intensive agriculture is of increasing concern (Pimentel and Pimentel 1996; Pimentel *et al.* 2005, 2008).

Food systems have also been identified as a major contributor to global-scale environmental change. For example, it is estimated that food systems contribute 30% to anthropogenic greenhouse gas emissions in the EU (EU 2006). Due to enhanced biological nitrogen fixation in agriculture and the production and use of nitrogen fertilizers, food production is also the primary source of reactive nitrogen mobilization, accounting for roughly 80% of anthropogenic fixation (Socolow 1999, Galloway *et al.* 2004, 2008). Moreover, it is a key driver of biotic resource appropriation (Vitousek *et al.* 1986; Imhoff *et al.* 2004; Haberl *et al.* 2007) and consumes significant amounts of energy (Pimentel and Pimentel 1996; Pimentel *et al.* 2005). Given that total food production volumes are anticipated to double by 2050 (FAO 2006) to meet the demand of a growing and increasingly affluent population, how to meet these demands without severely compromising ecological integrity across scales constitutes a defining challenge for contemporary society (Pelletier *et al.* 2008). This challenge is significantly increased by the rising prevalence of meat-based diets (Steinfeld *et al.* 2006).

The expansion of industrial animal husbandry has engendered both substantial productivity gains and environmental consequences. A recent report from the UNFAO implicates the livestock sector as among the leading contributors to a spectrum of critical environmental problems (Steinfeld *et al.* 2006). Occupying 70% of cultivated land and 30% of total land surface area, the livestock sector is the leading land user and cause of deforestation. By association, animal husbandry is also thought to be a central driver in biodiversity loss, land degradation, pollution, climate change, overfishing, sedimentation of coastal areas, and spread of invasive species (Steinfeld *et al.* 2006). Specifically, it has been estimated that the sector accounts for 8% of global freshwater use, 9% of CO_2 emissions, 37% of methane emissions, 65% of nitrous oxide emissions, and 64% of ammonia emissions. In total, livestock production alone is responsible for 18% of anthropogenic greenhouse gas emissions (Steinfeld *et al.* 2006).

In recent decades, considerable research effort has been invested in elucidating the material and energy dependencies and the attendant macroscale environmental implications associated with diverse food production systems. Odum's pioneering work on the energetics of global food systems (Odum 1967) spawned a wealth of research regarding energy use in food production, much of which was lead by American researcher David Pimentel (as summarized in Pimentel and Pimentel 1996). More recent work on food system energetics includes analyses of beef (Heitschmidt *et al.* 1996), conventional and organic dairy (Refsgaard *et al.* 1998), bread (Gronroos *et al.* 2006) and poultry production systems (Castellini *et al.* 2006).

Ecological footprint analyses have similarly been used as an indicator of biophysical sustainability in food systems, and have variously been applied to tomato, dairy and wine production (Wada 1993; Thomassen and de Boer 2005; Niccolucci *et al.* 2008), farms and cropland (van der Werf *et al.* 2007; Cuadra and Bjorklund 2008; Liu *et al.* 2008), several aquaculture products (Larsen *et al.* 1994; Kautsky *et al.* 1997; Tyedmers 2000), and to quantify the resource appropriation associated with different dietary patterns (White 2000; Gerbens-Leenes and Nonhebel 2002; Deutsch and Folke 2005).

For example, Gerbens-Leenes and Nonhebel (2002) compared the land requirements associated with existing and hypothetical diets and per capita consumption in Europe. It was found that a hypothetical wheat-based diet required six times less land than the existing meat-based diet, and that future changes in consumption patterns will be more important than population growth in determining total land requirements for food production. White (2000) used ecological footprinting to examine relationships between scale and distribution issues in global food consumption, concluding that differences in diet across populations lead to inequality in the distribution of environmental impact. In particular, it was shown that meat-intensive diets in industrialized countries results in inequality in the use of global environmental services.

Life cycle assessment has been the most widely used framework for studying environmental performance in food systems from a supply chain perspective in recent decades. The vast majority of these studies have treated single product systems or made comparisons between production technologies. Published studies have variously investigate oil seed crops (Schmidt 2007; Pelletier *et al.* 2008; Dalgaard *et al.* 2008), dairy systems (Cederberg and Mattsson 2000; Hogass-Eide 2002; Casey and Holden 2005; Olesen *et al.* 2006; Thomassen and De Boer 2008; Arsenault *et al.* 2009); beef production (Nunez *et al.* 2005; Ogino *et al.* 2004, 2007; Casey and Holden 2006), pork production (Nunez *et al.* 2005; Erikkson *et al.* 2005; Basset-Mens and van der Werf 2005) and poultry production (De Boer *et al.* 2000; Mollenhorst *et al.* 2006; Ellingsen and Aanondsen 2006; Williams *et al.* 2006; Pelletier 2008). Several studies of fisheries and aquaculture production systems have also been reported (Zeigler *et al.* 2003; Papatryphon *et al.* 2004; Hospido and Tyedmers 2005; Thrane 2006; Aubin *et al.* 2006; Ellingsen and Aanondsen 2006; Mungkung *et al.* 2006; Pelletier and Tyedmers 2007; Gronroos *et al.* 2006; Ayer and Tyedmers 2008; Pelletier *et al.* 2009; Pelletier and Tyedmers 2010)

At an aggregate level, LCA has been used to investigate the environmental implications of different dietary choices, with a particular emphasis on plant versus meat-based diets (Carlsson-Kanyama 1998, 2004; Reijinders and Soret 2003; Zhu and van Ierland 2004;

Eshel and Martin 2006; Garnett 2007). What is clear from all of these studies is that the impacts of food production vary widely along different dimensions of environmental performance. In general, animal husbandry products are much less eco-efficient than crop production systems per unit of nutrition provided, with large differences between animal husbandry systems. This is unsurprising given the inefficiencies inherent to biological feed conversion, and the range of conversion efficiencies characteristic of each species and husbandry system (Pelletier and Tyedmers 2007).

In comparison, relatively few researchers have attempted to scenario model the projected impacts of food production systems over time. In a widely cited study published in 2001, Tilman and colleagues (2001) forecasted the potential non-climate related global environmental impacts of agriculture over 20-50 years by extrapolation from past global trends, population growth and GDP. Because the historical trends employed were influenced by technological developments, shifting consumer preference and regulatory change, the modeled trajectories implicitly assumed the prevalence of similar influences into the future. It was estimated that continuance along current trajectories would result in the further conversion of 10^9 hectares of ecosystem for agricultural purposes. It was also predicted that global N fertilization would increase 1.6 fold by 2020 and 2.7 fold by 2050, at which time annual addition of N to terrestrial systems would be 236 x 10^6 T per year compared to 140 x 10^6 T per year from all natural sources. Combined with increasing P emissions, this would result in a 2.4-2.7 fold increase in eutrophication. These changes, the authors suggested, would result in ecosystem simplification, reduced provision of ecosystem goods and services, and biodiversity loss (Tilman *et al.* 2001).

With growing populations and increasing affluence, demand for food products is accelerating rapidly. Between 1970 and 2000, global daily per capita caloric intake increased from 2411 to 2789 calories and is anticipated to further increase to 3150 calories per capita by 2050 (FAO 2006). Moreover, changing consumptive preferences are resulting in diets higher in vegetable oils, fish and livestock products (Delgado *et al.* 1999). At present, livestock products contribute in excess of 17% of food energy and 33% of dietary protein intake globally. According to FAO projections, these proportions will increase to 30% and 50% respectively by 2050 (FAO 2006), effectively doubling meat production from 1999/2001 levels of 229 million tonnes to 465 million tonnes in 2050. In other words, environmental impacts must be halved per unit production just to maintain currently unsustainable damage levels (Steinfeld *et al.* 2006).

To meet burgeoning demand, these sectors are in a state of rapid expansion, largely met by a move towards intensive husbandry systems (Steinfeld *et al.* 2006). What is less clear are the potential environmental impacts of meeting projected demands, the extent to which these might be mitigated by alternative production strategies and consumption patterns, and the implications for sustainability objectives. The subsequent four chapters contribute to resolving these questions.

In Chapter 9, scenario models are used to conservatively estimate a subset of the potential global environmental costs of livestock production in 2050 relative to year 2000 levels and estimates of sustainability boundary conditions for human activities as a whole. Specifically, aggregate greenhouse gas emissions, reactive nitrogen mobilization, and biomass appropriation are assessed because the livestock sector is major contributor to each of these issues.

Global GHG emissions due to human activities have grown rapidly since the industrial revolution, increasing 70% between 1970 and 2004 (IPCC 2007). On current trajectories, it is estimated that the associated anthropogenic climate change may increase global mean temperatures by 3^{o} by 2100 (IPCC 2007). Given that a rise of 2°C above pre-industrial levels may result in ‘dangerous climate change,’ with serious negative impacts to ecosystems and human welfare, this issue has moved to the fore of global environmental governance discourse (IPCC 2007; Garnett 2008; Allison *et al.* 2009).

To date, no full cradle-to-plate estimates of global food system greenhouse gas emissions are available (Garnett 2008). However, IPCC (2007) estimates the contribution from agriculture at 10-12%, not accounting for land conversion effects. If the latter is included, one study estimates agriculture’s contribution at 17-32% of anthropogenic emissions

(Bellarby *et al.* 2008). Estimates of full supply chain emissions are available for the EU-25, which suggest that the food system contributes 31% to total emissions (EU 2006). For animal husbandry, livestock production is thought to be responsible for 18% of anthropogenic greenhouse gas emissions – a share greater than that of transport (Steinfeld *et al.* 2006). The projected doubling of livestock production by 2050 (FAO 2006) suggests that mitigating greenhouse gas emissions in this sector must be a research and policy priority such that changes in this sector do not undermine climate change prevention efforts.

Nitrogen is one of the key elements required for life, as well as the most abundant element in the earth's atmosphere. Atmospheric N, however, exists in a stable form (N_2) inaccessible to most life forms until fixed in a reactive form (N-) via combination with carbon, hydrogen or oxygen. The supply of reactive nitrogen plays a pivotal role in controlling the productivity, carbon storage, and species composition of ecosystems (Vitousek *et al.* 1997).

Ice core records suggest that reactive nitrogen levels fluctuated very little during the 2000 years prior to the industrial revolution, when a significant upward climb began (Socolow 1999). Over the past century, annual anthropogenic reactive nitrogen emissions have increased to the extent that human activities now contribute more fixed N to terrestrial ecosystems than do all natural sources combined (MEA 2005). Background levels have effectively doubled since 1970 (Vitousek *et al.* 1997; Socolow 1999; MEA 2005; Galloway *et al.* 2008).

Alteration of the nitrogen cycle has numerous consequences, including: increased radiative forcing of the atmosphere due to elevated nitrous oxide levels (Vitousek *et al.* 1997); photochemical smog and acid deposition associated with increased ammonia and nitric oxide flux (Vitousek *et al.* 1997); productivity increases leading to ecosystem simplification and biodiversity loss (Vitousek *et al.* 1997); and eutrophication of water bodies (Socolow 1999; Galloway *et al.* 2003, 2004, 2008). Moreover, reactive nitrogen is known to cascade through ecosystems (Galloway *et al.* 2003), sequentially contributing to these impacts as it cycles from one form to another.

According to Socolow (1999), the food system plays a central role in disrupting the nitrogen cycle through industrial fertilizer production and biological nitrogen fixation via the cultivation of leguminous crops. These activities are increasing rapidly. Roughly 50% of total synthetic nitrogen fertilizer application has occurred within the last 15 to 20 years (INI 2006). Of this, it is estimated that only 10-20% was actually consumed by humans, 95% of which was subsequently lost to the environment (INI 2004). Of total reactive nitrogen emissions, a much smaller share is contributed by fossil fuel combustion (Galloway *et al.* 2003).

Concerted research into the potential impacts of reactive nitrogen accumulation in ecosystems is only now beginning. Yet the critical role of reactive nitrogen in a spectrum of environmental problems at local, regional and global scales, coupled with its central importance in food production, suggests that resolving this issue may prove to be the most challenging dilemma that industrial society has yet encountered. The livestock sector is the single largest contributor to reactive nitrogen mobilization (Steinfeld *et al.* 2006).

Global estimates of biotic resource use have been reported by several researchers (Vitousek *et al.* 1986; Haberl *et al.*, 2006, 2007; Krausmann *et al.* 2008). At present, it is estimated that humans appropriate 24% of potential net primary productivity, 53% of which is attributable to harvest, 40% to productivity changes related to land use, and 7% to fires of anthropogenic origin (Haberl *et al.* 2007). In other words, the human food system currently consumes in the region of 12% of total potential net primary production. Moreover, Haberl *et al.* (2007) predict that humans may appropriate as much as 50% of NPP by 2050. More recently, Krausmann and colleagues suggested that 58% of directly used human-appropriated biomass was utilized by the livestock sector in 2000. Given the inefficiencies inherent to biological feed conversion, the projected expansion of animal husbandry will likely figure large in future anthropogenic biomass consumption, with attendant consequences for ecosystem integrity (Haberl *et al.* 2004). Of particular import will be the extent to which animal husbandry systems rely on animal products as feed inputs.

In Chapters 10-12, life cycle assessment is used to evaluate an important subset of the environmental dimensions of livestock production in the major animal husbandry sectors in the United States from the perspective of eco-efficiency. Here, four measures of environmental performance are applied – cumulative energy demand, greenhouse gas emissions, eutrophying emissions, and ecological footprint – which are thought to constitute an important subset of indicators of resource use and emissions intensities in livestock production at a variety of scales. Three additional efficiency measures, based on the concept of biophysical "returns on investment" are also applied.

Cumulative energy use provides a strong indication of the extent to which current production technologies augment or supplant ecosystem services in order to maximize productivity. It also offers a measure of the non-renewable energy intensity of livestock production, which speaks to an important element of food security concerns. Given the finite nature of these resources, as well as the environmental (and social) costs of their production and consumption, maximizing efficiencies and minimizing the scale of aggregate use is clearly desirable.

The quantification of greenhouse gas emissions vis-à-vis global climate change concerns is similarly of obvious interest. This measure speaks to the relationships between geographically and temporally bounded economic activities and the perturbation of a global-scale biogeochemical cycle.

In contrast, eutrophying emissions are relevant at several scales, as they may impact on local water bodies, drive regional eutrophication concerns (for example, the hypoxic zone in the Gulf of Mexico linked to nutrient run-off from agriculture within the Mississippi drainage), as well as contribute to global scale perturbations of the nitrogen and phosphorus cycles.

Finally, the ecological footprint metric allows for a direct assessment of the ultimate dependence of these production systems on finite bioproductive land to furnish an important subset of resource inputs and to assimilate a subset of the resultant wastes. Livestock production is the dominant user of land globally (Steinfeld *et al.* 2006). In concert, these measures thus provide a robust and complementary basis for assessing important efficiency considerations within and between livestock production systems.

Clearly, the reality of finite resources and waste sink capacities point to the desirability of managing food systems for efficiency objectives. While the previously described resource use and emissions considerations speak to important dimensions of ecological efficiency, our capacity to make informed policy and management decisions in the interest of eco-efficiency will certainly be improved by the consistent application of a broad suite of robust indices. Towards this end, I also apply three additional efficiency metrics for livestock production.

As noted by ecologist Charles Elton, energy is the currency of the economy of nature (Worster 1994). Ecological economics recognizes that the human economy is a subsystem of the biosphere, hence energy is similarly, by default, the fundamental currency of human economies. Since energy is available in finite quantities and because thermodynamic principles dictate that all energy transformations result in increased high entropy waste, managing human activities for energy efficiency is necessary from both a resource and waste perspective. What is unique about current human economies is that, whereas most species are directly or indirectly dependent on flows of solar energy through trophic webs, industrial society is, at present, also heavily dependent on flows of fossil solar energy in the form of non-renewable petrochemical resources. Hence, our energy efficiency concerns and management strategies must take into account both biotic and abiotic energy flows.

In particular, modern food systems are heavily dependent on fossil energy inputs in the form of fuels for farm machinery, fertilizer and pesticide production, and irrigation. For example, Pimentel *et al.* (2008) estimate that the annual diet of the average American is underpinned by approximately 2,000 l/year of oil equivalents, which accounts for roughly 19% of total energy use in the USA. Agricultural production, food processing and packaging account for 14%, while transportation and preparation consume 5%. The fossil energy required to produce the animal products consumed in the average American diet are estimated to account for 50% of the total energy inputs (Pimentel *et al.* 2008).

Following the energy shocks of the 1970's, a publication titled "Eating Oil: Energy Use in Food Production" called attention to the fossil fuel dependency of global food systems (Green 1978). More recently, Crews and People (2004) questioned the life history strategy of a species that has become fundamentally dependent on a non-renewable resource for its food supply. Mears (2007) estimates that energy inputs to agricultural crop production worldwide, most of which are non-renewable, account for less than 3% of global energy use. However, if the energy costs of processing and transporting feedstuffs, as well as energy inputs and conversion efficiencies in animal husbandry were included, the estimate of total energy use in global food production would be much higher. The doubling of world food prices in 2007, largely attributable to rising energy costs and the use of farmland for biofuel production in the United States (IMF 2007), suggests that energy is set to once again become a central consideration in food policy. Tracking fossil energy use in global food systems is therefore valuable to both efficiency and scale considerations, as well as for evaluating distributive effects.

Taking into account biotic energy use efficiencies requires tracking flows of organic carbon, which represents the transferable product of photosynthesis, through human economies. Since organic carbon underpins almost every trophic web, effectively serving as a vehicle for the transfer of solar energy in a medium accessible to heterotrophs, it should be considered the primary currency of the economy of nature. It is similarly essential to climate stability and the provision of diverse ecosystem goods and services (Imhoff *et al.* 2004). In other words, as with fossil energy efficiency, tracking and managing biotic energy efficiency is similarly essential.

The concept of "return on investment" is a well-known term in financial management. In recent decades, energy economists and others have adapted this concept to the study of energy returns on investment (EROI) – in other words, the quantity of energy delivered by a production technology relative to the quantity of energy invested (for example, see Cleveland 2005). Others have further adapted the concept to quantify ratios of food energy output relative to industrial energy inputs, which provides an important measure of the non-renewable resource intensity of food production technologies (for example, see Troell *et al.* 2004). Here, I apply three measures of energy resource returns on investment which are intended to inform efficiency measures from three distinct perspectives. The first is human-edible food energy return on industrial energy investment, which provides an anthropocentric perspective on the non-renewable resource efficiencies of competing food production technologies. The second is human-edible energy returns on human edible energy investment – in other words, an anthropocentric perspective on renewable resource efficiencies. The third is gross chemical energy returns on gross chemical energy investment. This metric represents an ecological perspective on the efficiency with which the human species appropriates the fundamental currency of the economy of nature towards the end of satisfying our need for food energy.

In concert, the purpose of these modeling endeavors reported in Chapters 9-12 is to provide the information necessary to manage for sustainable scale and efficiency objectives in livestock production, and in the global food economy more generally.

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CHAPTER 9: WHAT'S AT STEAK? GLOBAL ENVIRONMENTAL COSTS OF LIVESTOCK PRODUCTION 2000 – 2050

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9.1 Publication Information

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9.2 Introduction

Global food systems play a pivotal role in anthropogenic environmental change (Tilman *et al.* 2001; MEA 2005; Krausmann *et al.* 2008; Weidema *et al.* 2008). In particular, the livestock sector is a key contributor to a range of critical environmental problems (Millennium Ecosystem Assessment 2005; Steinfeld *et al.* 2006). Substantial projected growth in this sector from 2000 – 2050 (FAO 2006, World Bank 2008) due to increasing population and per capita demand will greatly exacerbate these challenges. Environmental boundary conditions are biophysical limits which define a "safe operating space" for economic activities at a global scale (Rockstrom *et al.* 2009), including greenhouse gas emissions, reactive nitrogen mobilization, and biomass appropriation. Here, we conservatively estimate the aggregate direct contributions to these issues associated with producing edible livestock products in 2050 under four scenarios based on projected and alternative production and consumption patterns. Results are compared with known contributions of the livestock sector as of 2000, and with estimated sustainability boundary conditions for human activities as a whole. We show that, in 2050, the livestock sector alone will either occupy the majority of or considerably overshoot humanity's safe operating space in each of these domains. We suggest that contributions of livestock production to global environmental change relative to sustainability boundary conditions indicate that reining in growth of this sector should be a policy priority.

9.3 Estimating Livestock's Global Environmental Costs

As of 2000, the livestock sector contributed 14% of anthropogenic greenhouse gas emissions (18% taking into account land use, land use change and forestry) (Steinfeld *et al.* 2006), 63% of reactive nitrogen mobilization (calculated from Steinfeld *et al.* 2006), and consumed 37% of human-appropriated biomass globally (Krausmann *et al.* 2008), which has significant consequences for energy flows within food webs and the biodiversity that ecosystems can support (Imhoff *et al.* 2004). Using conservative models, we estimate that production of livestock in 2050 at levels projected by the United Nations Food and Agriculture Organization (FAO Projections) will increase direct livestock-related GHG emissions from meat, milk and egg production by 39%, biomass appropriation (NPP) by 21% and reactive nitrogen mobilization by 36% above year 2000 levels (Figure 9.1). However, there is a wide range in resource and emissions intensity between different livestock products. Accordingly, under the same conditions, substituting poultry for all marginal beef production above year 2000 levels would reduce these impacts by a modest 5-13%. Similarly, human protein needs may be satisfied in numerous ways, and with differing contributions from livestock products. The range of impacts associated with achieving USDA-recommended per capita protein consumption levels when derived either in entirety from meat/eggs and dairy, or from soy protein, for global populations in 2050, spans almost two orders of magnitude (Figure 9.1). This underscores the considerable role of dietary patterns in determining environmental outcomes.

As of 2000, we estimate that the livestock sector alone occupied 52% of humanity's "safe operating space" for anthropogenic greenhouse gas emissions and that direct appropriation of biomass accounted for 72% of sustainable human appropriated net primary productivity (Figure 9.1). The sustainability boundary condition for reactive nitrogen mobilization was exceeded by 117%. We further estimate that, by 2050, meeting projected demands for edible livestock products will increase these shares to 70%, 88% and 294% of humanity's "safe operating space" (Figure 9.1). We also suggest that if the livestock sector is to grow as forecasted but maintain its current proportional share of contributions to these issue areas and human activities are to be constrained to respect sustainability boundary conditions, this will require reducing greenhouse gas emissions per unit livestock protein produced to 13% of year 2000 levels, biomass appropriation to 25% and reactive nitrogen mobilization to 14%.

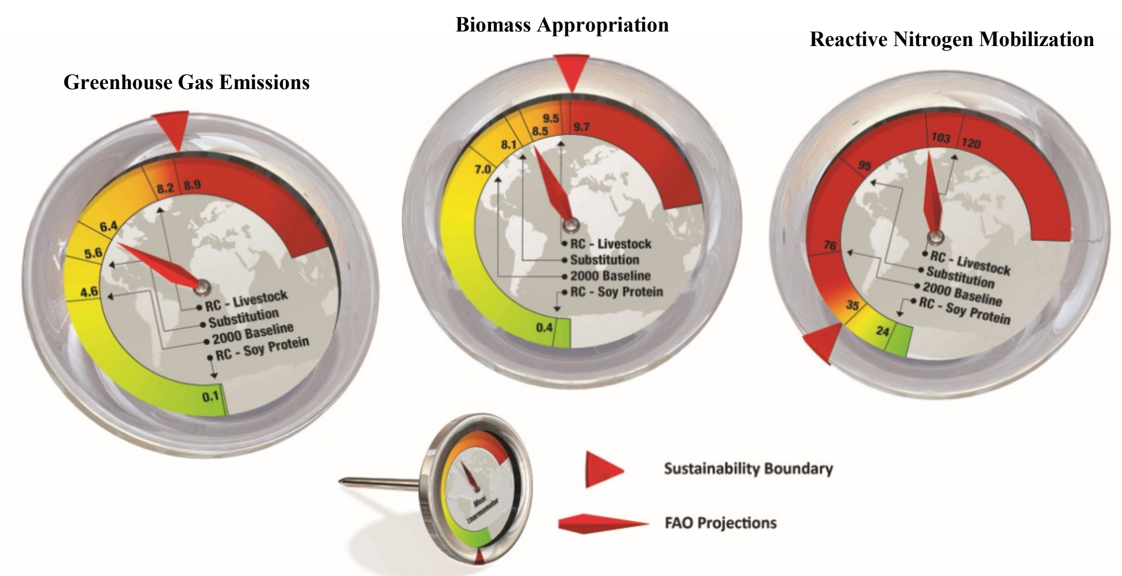

Figure 9.1. Estimated greenhouse gas emissions (Gt CO_2-e) , biomass appropriation (Gt C), and reactive nitrogen mobilization (Mt N_r) associated with the global livestock sector in 2000 versus 2050 under FAO production estimates (FAO Projections) as well as three alternative scenarios (Substitution; Recommended Consumption (RC) – Livestock; and RC Soy Protein) relative to sustainability boundary conditions for human activities in aggregate.

Modeling the future is fraught with uncertainties. We have thus erred on the side of caution in developing what are almost certainly highly conservative forecasts of the potential future environmental impacts of livestock production. While providing only a coarse-grained indication of the role that the livestock sector is likely to play, we nonetheless posit a profound disconnect between the anticipated scale of environmental impacts associated with projected livestock production levels and even the most optimistic mitigation strategies relative to estimated sustainable biocapacity. As such, these observations merit serious consideration in food policy and environmental governance discourse.

9.4 Implications For Food And Environmental Policy

Due to the biological inefficiencies inherent to feed conversion, animal products have been described as the electricity of food (Smil 1999). Certainly, substantial efficiency gains in recent decades and the range of existing regional efficiencies of livestock production suggest additional opportunities for improvements both at the level of feed crop production and animal husbandry (Steinfeld *et al.* 2006; Capper *et al.* 2009; Spiertz and Ewert 2009) – all of which should be vigorously pursued (Beddington 2010). However, from our analysis (which includes generous assumptions regarding efficiency gains over time) it is clear that such objectives are unlikely to be met by technological means alone. Instead, all feasible options for reducing impacts in this sector must be considered (Eriksen *et al.* 2009) if its relative and absolute contributions to achieving sustainability objectives are to be met.

This would include policies aimed at a shift in production away from ruminants (McAlpine *et al.* 2009) and towards lower impact species such as poultry (McMichael *et al.* 2007; Sustainable Development Commission 2009) through targeted taxes, subsidies and regulation. Well-managed fisheries and aquaculture might also stand to displace a share of terrestrial animal protein production, with careful attention to tradeoffs (Pelletier *et al.* 2009). Across the board reductions in per capita consumption of livestock products should also be a policy priority. To meet sustainability boundary conditions whilst maintaining the year 2000 proportional contribution to total anthropogenic impacts for GHG emissions, biomass appropriation, and reactive nitrogen mobilization implies reductions in anticipated per capita consumption in 2050 to 19%, 42%, and 21% of projected levels respectively. Such reductions may be particularly feasible and advantageous in developed countries where consumption is currently twice USDA-recommended levels. Certainly, a redistribution of livestock consumption from food surplus to food deficit regions would have coupled health and environmental benefits

(McMichael *et al.* 2007; Stehfest *et al.* 2009). However, curbing growth in anticipated consumption in developing countries, where the majority of increase in production is projected to occur (Steinfeld *et al.* 2009), will also be critical. Satisfying nutritional requirements through largely plant-based diets must be emphasized, while remaining sensitive to the developmental status and aspirations of the less advantaged, as well as the environmental implications of specific plant protein production strategies – for example, soybean agriculture in South America (Nepstad *et al.* 2006; Soares-Filho *et al.* 2006; McAlpine *et al.* 2009).

We stress that the estimates reported here are likely conservative, and that improved understanding of sustainable boundary conditions may continue to shift thresholds downwards. Moreover, increased competition for limited resources including energy for fertilizers, pesticides and fuels, arable land for crops destined for direct human consumption, and political pressures for expanded biofuel production will require difficult trade-offs (Spiertz and Ewert 2009). Given the limited consideration of the livestock sector in environmental governance regimes to date and the scale of the issues to be addressed, mobilizing the necessary political will to implement such policies is a daunting but necessary prospect. As the human species runs the final course of rapid population growth before beginning to level off mid-century, and food systems expand at commensurate pace, reining in the global livestock sector should be considered a key leverage point for averting irreversible ecological change and moving humanity towards a safe and sustainable operating space.

9.5 Methods Summary

We coupled published estimates of aggregate greenhouse gas emissions, biomass appropriation, and reactive nitrogen mobilization for the global livestock sector in 2000 with United Nations Food and Agriculture Organization (FAO) projections for production of edible livestock products from 2000-2050 (FAO 2006). We conservatively assumed that all predicted increases in livestock production will occur in intensive, arable crop-based as opposed to extensive fodder-based animal husbandry system (Keyzer *et al.*

2005; Steinfeld *et al.* 2006; Keyser) and that impacts per unit production above year 2000 levels are equivalent to those reported for the most efficient intensive animal husbandry sectors. This generously implies an average global decrease in impacts per unit livestock protein produced of 35% from year 2000 levels by 2050. We subsequently predicted changes in the scale of absolute impacts over time associated with projected production levels (FAO Projections) and under three additional scenarios where we assumed that (1) all marginal production of beef is substituted for poultry (Substitution); (2) per capita consumption of protein from meat/legume sources matches United States Department of Agriculture (USDA) Food Pyramid recommendations and is satisfied entirely by livestock products at projected production ratios (RC – Livestock) and; (3) USDA Food Pyramid-recommended per capita protein consumption is satisfied entirely by soy protein (RC – Soy Protein). We then contrasted anticipated 2050 impact levels between scenarios relative to year 2000 levels (for a full methods description, see Supporting Information).

We further estimated the distance to threshold for each of these livestock production scenarios relative to sustainability boundary conditions of: 8.9 Gt of total anthropogenic CO_2-e/year for GHG emissions (necessary to stabilize atmospheric CO_2 at 350 ppm) (Allison *et al.* 2009); 35 Mt of N_r removed from the atmosphere per year (Rockstrom *et al.* 2009); and an annual human appropriation of net primary productivity (HANPP) rate of 9.7 Gt of carbon (Bishop *et al.* 2009).

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9.7 Supporting Information

9.7.1 Materials And Methods

Our overarching purpose is to: (a) compare and contrast conservative estimates of GHG emissions, biomass appropriation, and reactive nitrogen mobilization associated with projected production levels of edible livestock products in 2050 (FAO Projections scenario) relative to reported 2000 production levels and associated impacts for the livestock sector as a whole; (b) evaluate the mitigation potential of alternative consumption patterns where technological or consumption variables are manipulated; and (c) assess the scale of impacts associated with these scenarios relative to both current reported impacts and estimates of sustainable, global-scale boundary conditions for the impact categories of concern.

We draw on previously published global-scale estimates of greenhouse gas emissions (CO_2-e), biomass appropriation (C), and reactive nitrogen mobilization (N_r) associated with the global livestock sector in 2000 as our baseline. It should be noted that, although the livestock sector also furnishes important non-edible product/service streams such as wool, leather and draft-power, we attribute all current and future impacts to the production of edible products. We also do not consider second order effects potentially associated with the scenarios considered, such as land use change or market-mediated effects.

We adopt estimates of production of edible livestock products in 2000 and 2050 from UNFAO data (FAO 2006). To account for likely continued increases in efficiency in the livestock sector, we make the conservative assumption that all additional production above year 2000 levels to meet anticipated consumption levels of edible livestock products in 2050 will achieve productive efficiencies comparable to those of the most efficient contemporary, arable-crop based livestock systems in industrialized countries based on literature values. This is consistent with predicted constraints to the expansion of pasture lands due to increased competition for land for arable crops, and a transition towards intensive, grain-fed husbandry systems (Keyzer *et al* 2005; Steinfeld *et al.* 2006). Hence our estimates of marginal (post-2000 level) impacts are based on contemporary industrial production efficiencies whereas our estimates of aggregate impacts in 2050 are the sum of reported year 2000 (baseline) and estimated year 2050 (marginal) impacts. We model impacts for live-weight production as equivalent to consumption on the generous assumption that all processing co-products not directly consumed by humans are cycled back into the livestock feed production stream or otherwise diverted to further economic use and are thus allocated a commensurate share of production-related impacts. Current, anticipated, and marginal livestock production volumes (i.e. in addition to year 2000 production) to meet projected 2050 demands are detailed in Table S9.1.

Table S9.1. Estimated global production volumes of meat, eggs and dairy for 2000 and 2050 as reported by FAO (2006).

Production (1000 tonnes)	**Ruminant Meat**	**Pork**	**Poultry**	**Eggs**	**Dairy**
2000	70,715	90,666	68,331	51,194	577,494
2050	127,740	139,344	196,643	135,023	1,048,358
Marginal	57,025	48,678	128,312	83,829	470,864

9.7.2 Calculating GHG Emissions For The FAO Projections Scenario

Steinfeld *et al.* (2006) estimate that in 2000 the livestock sector was responsible for 13.9% of anthropogenic greenhouse gas emissions (4.6 out of 33 Gt CO_2-e), not including Land Use, Land Use Change and Forestry (LULUCF), and 18% when LULUCF is included. We adopt the lower estimate as our baseline for greenhouse gas emissions from the livestock sector in 2000, and do not attempt to account for LULUCF associated with projected livestock production in 2050. We assume that all marginal production of livestock products associated with anticipated consumption levels in 2050 will achieve greenhouse gas intensities per unit production comparable to those currently observed in the most efficient intensive livestock production systems in industrialized countries. Table S9.2 reports the range of literature values for greenhouse gas emissions per unit of livestock product produced in contemporary production systems in industrialized countries.

Table S9.2. Reported greenhouse gas intensities (in kg of CO_2-e/kg of product) for meat, egg and dairy production in industrialized countries.

Livestock Product	Region	Estimate (kg CO_2-e/kg)	Source
Beef[1]	Ireland	13[2]	Casey and Holden (2006)
	Sweden	13.5[3]	Carlsson-Kanyama (2004)
	France	14.3-18.3[4]	Veysset *et al.* (2009)
	United States	14.8-19.2[5]	Pelletier *et al.* (*in press*)
	United States	15.5 +/- 2.3[6]	Phetteplace *et al.* (2001)
	UK	15.8-25.8[7]	Williams *et al.* (2006)
	Sweden	22.3[8]	Cederberg and Stadig (2003)
	Sweden	29[7]	Cederberg *et al.* (2009)
	Europe	30[7]	Weidema *et al.* (2008)
Pork	United States	2.38-3.35[9]	Pelletier *et al.* (*in review*)
	France	2.30-3.97[10]	Basset-Mens and van der Werf (2005)
	Sweden	3.5[7]	Cederberg *et al.* (2009)
	Sweden	3.6-4.4[11]	Stern *et al.* (2005)
	Sweden	4.2[12]	Carlsson-Kanyama (2004)
	UK	6.4[13]	Williams *et al.* (2006)

Livestock Product	Region	Estimate (kg CO_2-e/kg)	Source
	Europe	11[7]	Weidema *et al.* (2008)
Poultry	Sweden	1.4[14]	Carlsson-Kanyama (2004)
	United States	1.4[15]	Pelletier (2008)
	Argentina	2.1[16]	Bennet *et al.* (2006)
	Sweden	2.1[7]	Cederberg *et al.* (2009)
	UK	4.6[17]	Williams *et al.* (2006)
	Europe	3.6[7]	Weidema *et al.* (2009)
Eggs	Netherlands	3.9-4.6[18]	Mollenhorst *et al.* (2006)
	UK	5.5[19]	Williams *et al.* (2006)
Dairy	Canada	1.0[20]	Arsenault *et al.* (2008)
	Sweden	1.0-1.1[21]	Cederberg and Mattsson (2000)
	UK	1.1 [22]	Williams *et al.* (2006)

(1) We conservatively assume that beef production is representative of ruminant production generally. (2) Live-weight/year in conventional production systems (3) Ready-to-eat (4) Live-weight (5) Live-weight grain- and grass-finished (6) Live-weight (7) Carcass-weight (8) Bone-free meat, organic, pastured (9) Live-weight, commodity production (10) Live-weight, conventional and organic (11) Bone-free, fat-free meat (12) Ready-to-eat pork (13) Carcass-weight (14) Ready-to-eat (15) Live-weight (16) Live-weight (17) Carcass-weight (18) In shell, battery and free-run (19) In shell (20) Raw milk, farm-gate (21) Raw milk (22) Raw milk

Based on this range of reported estimates, we adopt values for ruminant, pork and poultry meat which approximate those reported for contemporary US production systems by Pelletier (2008), and Pelletier *et al.* (*in review/press* a,b). These represent the conservative end of the range of reported emissions, and were generated using a consistent methodology. We assume that beef is representative of ruminant meat production, generally. Only two literature sources report values for eggs, both of which are considerably higher than the estimate for broiler production reported by Pelletier (2008). However, Williams *et al.* (2006) reports values for both poultry and eggs, with greenhouse gas emissions for egg production roughly 20% higher than for poultry production. We hence scale the estimate for US poultry production by 20% to arrive at our estimate for the greenhouse gas intensity of egg production. The values reported for milk production are remarkably consistent between literature sources. Our assumed GHG intensities per unit production of edible livestock products for all production above year 2000 levels are detailed in Table S9.3.

Table S9.3. Selected greenhouse gas intensities (in kg CO_2-e/kg) for modeling marginal impacts of livestock production in 2050.

	Beef	Pork	Poultry	Eggs	Milk
CO_2-e/unit production	15[1]	3[1]	1.5[1]	1.8[2]	1[3]

(1) Live-weight production

(2) Whole eggs, in shell

(3) Raw, unprocessed whole milk

9.7.3 Calculating Biomass Appropriation For The FAO Projections Scenario

We take dry matter biomass appropriation as equivalent to the appropriation of Net Primary Productivity (NPP_{act}) following Krausmann *et al.* (2008). We account only for directly consumed fractions, hence our estimates of biomass appropriation are quite conservative.

According to Krausmann *et al.* (2008), human terrestrial biomass appropriation was 18.7 billion tonnes of dry matter, which is equivalent to roughly 16% of global terrestrial NPP, in 2000. Of the directly used portion (12.1 billion tonnes), 58% was directed through the livestock system. This is equivalent to 1155 kg of dry matter (DM) of livestock feed per capita in 2000. At a population of 6.071 billion, this equals a total feed throughput of 7.012 billion tonnes DM directly used to produce livestock products in 2000.

We estimate additional feed demands for anticipated increases in livestock production by multiplying marginal production above year 2000 levels by feed efficiency factors as currently achieved in the most efficient industrialized, intensive animal husbandry systems (Table S9.4). These estimates are challenging, since feed efficiency will be influenced by numerous factors including diet composition – particularly for ruminants, where dry matter per unit production will vary widely with feed quality. It is important to also take into account feed use by breeding stock.

Table S9.4. Estimated feed conversion efficiencies for industrialized, intensive animal husbandry systems and marginal edible livestock product production in 2050.

	Ruminant Meat	Pork	Poultry	Eggs	Dairy
MarginalProduction[1]	57,025	48,678	128,312	83,829	470,864
Feed Efficiency[2]	10	2.5	2	2.4	.8

(1) Thousand tonnes

(2) Dry matter/unit production

9.7.4 Calculating Reactive Nitrogen Mobilization For The FAO Projections Scenario

Steinfeld *et al.* (2006) estimate that roughly 25% of global N fertilizer produced was applied to feed crops for animal husbandry in 2000, amounting to 20 million tonnes of N_r mobilized through the Haber-Bosch process. Applying the IPCC default emission factor of 1.25%, it is further estimated that biological nitrogen fixation associated with the growth of leguminous fodder consumed by livestock resulted in .7 million tonnes of N_2O emissions (Steinfeld *et al.* 2006). From this, it can be estimated that Nr mobilization associated with legumes consumed by livestock in 2000 amounted to roughly 56 million tonnes. In total, then, we estimate that in 2000 the livestock sector was directly responsible for mobilizing 76 million tonnes of atmospheric N as Nr into terrestrial ecosystems.

Steinfeld *et al.* (2006) also suggest that competition for agricultural land precludes opportunities for further expansion of pasture lands globally. We thus assume that current fodder crop production will remain constant such that all increases in livestock production above year 2000 levels will be predicated on the production of additional feed crops. Feed crop production for livestock in 2000, which was responsible for the mobilization of 20 million tonnes of Nr (Steinfeld *et al.* 2006), was estimated at 965 million tonnes (Steinfeld *et al.* 2006) or 900 million tonnes of dry matter (Krausmann *et al.* 2008).

Accordingly, we indirectly quantify future Nr mobilization for feed crop production by estimating dry matter requirements in 2050 relative to year 2000 levels and subsequently scaling the estimate of year 2000 Nr mobilization. To account for potential increases in nitrogen use efficiency, which would reduce Nr mobilization, we assume that nitrogen efficiency in feed crop production will improve from the current average of 50% to the theoretical optimum average of 70% by 2050 (Smil 1999; Steinfeld *et al.* 2006).

9.7.5 Substitution Scenario

Our substitution scenario explores the implications of replacing poultry (the most efficient) with ruminant (the least efficient) meat production. Towards this end, we simply replace all projected marginal beef production (i.e. above year 2000 levels) with poultry production, along with associated differences in GHG emissions, biomass appropriation, and reactive nitrogen mobilization, for 2050 (Table S9.5).

Table S9.5. Anticipated marginal greenhouse gas emissions and DM demands associated with livestock production in 2050 under and a Substitution scenario where all ruminant production above year 2000 levels is assumed to be substituted by poultry production.

	Ruminant Meat	**Pork**	**Poultry**	**Eggs**	**Dairy**	**Total**
Marginal Production[1]	0	48,678	185,337	83,829	470,864	
Feed Efficiency	10	2.5	2	2.4	.8	
Marginal CO_2-e		146,034	278,006	150,892	470,864	1,815,633
Marginal DM	0	121,695	370,674	201,190	376,691	1,070,250

(1) Thousand tonnes

9.7.6 Recommended Consumption Scenarios

Here, we explore the implications of global consumption of edible livestock products in 2050 using two variants of recommended consumption levels. For our first recommended consumption scenario, we use demographics-adjusted estimates of per capita annual consumption of meat-equivalent units of meat/eggs/legumes and whole milk-equivalent units of milk based on USDA (2009a) age group recommendations and UN median population forecasts for 2050. Ruminant, pork and poultry meat are considered equivalent. 1.9 units of eggs are considered equivalent to 1 unit of meat (USDA 2009a). We assume that relative consumption of livestock product types from each group corresponds to projected production ratios, adjusted for equivalency (Table S9.6). For this "100% Livestock" scenario (RC-Livestock), the 58.9 kg per capita meat-equivalent consumption of meat/eggs/legumes recommended by the USDA is assumed to be satisfied by ruminant meat, pork, poultry and eggs at projected production ratios. A recommended per capita consumption of whole milk equivalents of 225 kg is similarly applied following USDA guidelines.

For our second recommended consumption scenario, we assume that all protein provided under the RC-Livestock scenario is derived instead via direct consumption of soy beans (RC-Soy Protein). We derive estimates of protein content of meat, eggs, dairy and soy from the USDA nutrient database (USDA 2009b) (Table S9.7). For impacts of soy production, we assume: a GHG intensity of 0.2 tonnes CO_2-e/tonne (conservative end of range of literature values); a direct biomass appropriation of 0.9 tonnes/tonne DM (Sauvant *et al.* 2004); and an N_r mobilization rate of 50 kg/tonne of soy beans produced (based on the nitrogen content of soy beans (Sauvant *et al.* 2004) and crop above/below ground residues (IPPC 2006)). Estimated production volumes under these scenarios are detailed in Table S9.6.

Table S9.6. Production volumes of meat, eggs, dairy and soy beans under the RC-Livestock and RC-Soy Protein% Soy Protein recommended consumption scenarios.

Production (1000 tonnes)	**Ruminant Meat**	**Pork**	**Poultry**	**Eggs**	**Dairy**	**Soy Beans**
Production in 2050	127,740	139,344	196,643	135,023	1,048,358	0
RC-Livestock	125,613	137,024	193,369	132,775	2,276,760	0
RC-Soy Protein	0	0	0	0	0	475,986

Table S9.7. Protein content of raw livestock products and soy beans.

	Ruminant	**Pork**	**Poultry**	**Eggs**	**Dairy**	**Soy Beans**
Protein Content[1]	19%	17%	18%	11%	3%	36%

(1) All protein content values are for raw products, and are derived from USDA (2009b). Values for meats are averages.

9.7.7 Estimating Sustainable Boundary Conditions

In a recent paper from the Resilience Alliance published in Nature, Rockstrom and colleagues (2009) describe the concept of "boundary conditions" for the sustainable scale of human activities. These boundary conditions represent best estimates of the scale of annual anthropogenic impacts which the biosphere can sustain in perpetuity for particular impact areas. Specifically, they describe boundary conditions for anthropogenic contributions to: climate change; ocean acidification; stratospheric ozone depletion; the nitrogen and phosphorous cycles; freshwater use; land use; biodiversity loss; atmospheric aerosol loading; and chemical pollution.

Steinfeld *et al.* (2006) suggest that the livestock sector is a leading contributor to many serious environmental problems, whether at local, regional or global scales. In light of projected growth in the livestock sector from 2000 – 2050, it is therefore of great interest to ascertain the potential role of the livestock sector in contributing to the capacity of humanity to respect these boundary conditions. Here, we specifically examine the environmental profile of the livestock sector relative to boundary conditions for GHG emissions, biomass appropriation (which, as pointed out by Imhoff *et al.* (2004), has significant consequences for energy flows within food webs and the biodiversity that ecosystems can support) and reactive nitrogen mobilization. We draw our estimates of boundary conditions from recent, peer-reviewed literature (Table S9.8).

Rockstrom *et al.* (2009) define the sustainable boundary condition for anthropogenic greenhouse gas concentrations as 350 ppm, which is the level at which anticipated increases in mean surface temperatures will likely not exceed 2 degrees C. This target is supported by recent work by Malte *et al.* (2009). The review of climate science by Allison *et al.* (2009) suggests that per capita emissions will need to be well below 1 tonne per year to achieve this target, hence we calculate a conservative boundary condition of 8.9 Gt of CO_2-e per year in 2050. Steinfeld *et al.* (2006) estimate annual anthropogenic GHG emissions (not including LUCLUF) of 33 Gt for 2000, with the livestock sector contributing 4.6 Gt.

Rockstrom *et al.* (2009) suggest that the sustainable boundary condition for anthropogenic mobilization of atmospheric N_r is 35 million tonnes. This is a small fraction of their estimate of 121 million tonnes of anthropogenic reactive nitrogen mobilization for 2000, of which we estimate 76 million tonnes was attributable to the livestock sector.

Bishop *et al.* (2009) estimate that humans can sustainably appropriate 9.72 billion tonnes of net primary production annually. Krausmann *et al.* (2008) estimate that in 2000, humanity collectively appropriated 18.7 billion tonnes of NPP as biomass, 39% of which was used directly to produce livestock products.

Table S9.8. Estimates of sustainable planetary boundary conditions for anthropogenic GHG emissions, NPP appropriation, and reactive nitrogen mobilization relative to year 2000 levels and estimated contributions from the livestock sector in 2000.

Boundary Condition	**Proposed Boundary**	**Status in 2000**	**Contribution from Livestock Sector in 2000**
Greenhouse Gas Emissions	8.9 Gt CO_2-e	34 Gt CO_2-e	4.6 Gt CO_2-e per year (not including LULUCF)
Biomass Appropriation	9.72 Gt C	18.7 Gt C	7.0 Gt C
Reactive Nitrogen Mobilization	35 Mt of atmospheric nitrogen	121 Mt of atmospheric nitrogen	76 Mt of atmospheric nitrogen

9.7.8 Results

GHG Emissions

By multiplying anticipated marginal production by our conservative estimates of GHG intensity per unit production, we estimate an additional 1.82 Gt of CO_2-e emissions from producing edible livestock products by 2050, which represents an increase of roughly 40% above current livestock-related emissions (Table S9.9), not including LULUCF.

Table S9.9. Estimated marginal greenhouse gas emissions (CO_2-e) associated with the production of meat, eggs and dairy in 2050 above reported year 2000 levels.

Production 1000 Tonnes	**Ruminant Meat**	**Pork**	**Poultry**	**Eggs**	**Dairy**	**Total**
Marginal Production	57,025	48,678	128,312	83,829	470,864	
CO_2-e/unit production	15	3	1.5	1.8	1	
Marginal CO_2-e	855,375	146,034	192,468	150,892	470,864	1,815,633

Biomass Appropriation

We estimate additional feed demands for anticipated increases in livestock production by multiplying marginal production above year 2000 levels by feed efficiency factors as currently achieved in the most efficient industrialized, intensive animal husbandry systems (Table S9.10). We estimate that an additional 1.53 billion tonnes of DM will be required annually for the production of edible livestock products by 2050, which represents an increase of 21.8% above year 2000 biomass appropriation levels for the livestock sector as a whole.

Table S9.10. Estimated marginal biomass appropriation (C, measured as DM) associated with the production of meat, eggs and dairy products in 2050 relative to year 2000 production levels.

	Ruminant Meat	**Pork**	**Poultry**	**Eggs**	**Dairy**	**Total**
Marginal Production[1]	57,025	48,678	128,312	83,829	470,864	
Feed Efficiency[2]	10	2.5	2	2.4	.8	
Marginal DM	570,250	121,695	256,624	201,190	376,691	1,526,450

(1) Thousand tonnes

(2) DM/unit production

Reactive Nitrogen Mobilization

Applying the estimates for marginal DM production of 1,526,450 Mt required to support livestock production above year 2000 levels in 2050 (Table S9.5), this requires an increase of 70% in feed crop production for livestock. At status quo average nitrogen use efficiencies of 50% in feed crop production (Smil 1999; Steinfeld *et al.* 2006), this represents additional inputs of 33.8 million tonnes of Haber Bosch Nr. Making the optimistic assumption that nitrogen efficiency in crop production can be improved to the theoretical optimum average of 70% (Smil 1999; Steinfeld *et al.* 2006)) over this time frame, this will nonetheless require additional Haber-Bosch Nr mobilization of 27 million tonnes. This represents an increase of 36% above year 2000 levels.

Scenario Results

Under the Substitution scenario, anticipated marginal CO_2-e emissions would rise by 22% and biomass appropriation would increase by 15% relative to year 2000 levels. Anticipated Nr mobilization would increase by 25% relative to year 2000 levels. However, relative to the FAO Projections scenario, substituting poultry for marginal ruminant production would reduce anticipated GHG emissions by only 13%, biomass appropriation by 5%, and reactive nitrogen mobilization by 8%.

For the RC-Livestock scenario, aggregate recommended meat and egg consumption is 98% of projected consumption levels, suggesting that anticipated livestock production levels in 2050 will be sufficient to satisfy USDA dietary recommendations for meat consumption for the entire human population provided it is equitably distributed. In contrast, recommended dairy consumption is 217% higher than the projected supply in 2050 and 394% higher than year 2000 supplies (Table S9.8).

For the RC-Soy Protein scenario, we estimate a total protein equivalency of 164,875 thousand tonnes, which is satisfied by 457,986 thousand tonnes of soy beans. This is 284% higher than global year 2000 production volumes (FAOstat 2009), but very comparable to projected production levels in 2050 (FAO 2006) (Table S9.8).

Largely due to the much higher demand for dairy products, GHG emissions are 28% higher for the RC-Livestock scenario relative to projected year 2050 impact levels. Biomass appropriation is 12% higher and N_r mobilization is 17% higher. For the RC-Soy Protein scenario, GHG emissions are 1.5% of anticipated 2050 levels, and biomass appropriation is 5%. Reactive nitrogen mobilization is 23% (Table S9.11).

Table S9.11. Comparative GHG emissions, biomass appropriation, and reactive nitrogen mobilization associated with projected livestock production in 2050 under FAO Projections and three alternative scenarios relative to year 2000 (baseline) levels.

Scenario	CO_2-e (Gt)	Biomass (Gt)	N_r (Mt)
Year 2000 Baseline	4.6	7.0	76
FAO Projections	6.4	8.5	103
Substitution	5.6	8.1	95
RC-Livestock	8.2	9.5	120
RC-Soy Protein	0.1	0.4	24

Assessment Of Results Relative To Sustainable Boundary Conditions

Based on current estimated GHG emissions for the livestock sector and additional emissions to meet projected demands for livestock products in 2050 assuming contemporary industrialized country production norms for marginal production, we estimate that the livestock sector alone will contribute 72% of sustainable boundary condition GHG emissions at that time. This is over five times the relative contribution (13.9%) of the livestock sector to total anthropogenic emissions in 2000.

Bishop *et al.* (2009) estimate that humanity can sustainably appropriate 9.72 billion tonnes of net primary production annually. By our conservative estimates, in 2050 the livestock sector alone will directly require 8.54 billion tonnes of NPP as dry matter biomass. Taking into account indirect appropriation, this suggests that the livestock sector will exceed the sustainable scale of anthropogenic biomass use under FAO Projections.

Even at feed conversion efficiencies currently characteristic of intensive animal husbandry operations in industrialized countries for all marginal livestock production in 2050 compared to 2000, and an increase in crop N use efficiency to the theoretical maximum for all marginal production, by 2050 we estimate that annual N_r mobilization for the livestock sector will amount to close to 100 million tonnes under FAO Projections

– exceeding the sustainability boundary condition for anthropogenic Nr mobilization by a factor of almost three.

Replacing marginal ruminant production with poultry production (Substitution scenario) would reduce aggregate impacts 5-13%. Producing sufficient livestock products to satisfy per capita consumption recommendations as per USDA guidelines would slightly reduce impacts for meat products, but aggregate impacts would increase 12-28% relative to the FAO Projections scenario due to a much higher production of milk. Providing equivalent soy protein to that associated with RC-Livestock scenario (RC-Soy Protein) would result in major reductions in GHG emissions (98%), biomass appropriation (94%) and reactive nitrogen mobilization (32%). Under this scenario, satisfying human protein needs would contribute 1.1% to boundary condition GHG emissions, 1.1% to boundary condition biomass appropriation and 69% of boundary condition reactive nitrogen mobilization.

Notes On Efficiency Assumptions

Based on year 2000 production levels for edible livestock products, the aggregate edible protein content of these products, and reported baseline impacts for the livestock sector, we estimated average impacts/unit protein production for 2000. We then compared these estimates to our calculations of impacts per unit protein production in 2050 to evaluate the scale of efficiency gains implied by our models. For greenhouse gas emissions, our calculations imply a 31% efficiency gain. For biomass, the implied gain is 40% and, for reactive nitrogen, 34% (Table S9.12).

Table S9.12. Estimated GHG emissions, biomass appropriation, and reactive nitrogen mobilization per kg of edible protein produced by the livestock in 2000 and 2050, and implied efficiency gains.

	Protein Production (1000 tonnes)	**GHG intensity (kg CO_2e-/kg)**	**Biomass appropriation (kg C/kg)**	**Reactive nitrogen (kg N_r/kg)**
2000	64,105	71.8	109.2	1.2
2050	129,658	49.4	65.6	0.8
Implied Efficiency Gain		31%	40%	34%

We further estimate necessary efficiency gains in order for the livestock sector to maintain its year 2000 proportional contribution to total anthropogenic greenhouse gas emissions, biomass appropriation and reactive nitrogen mobilization when total human contributions are constrained to match the proposed boundary conditions. For year 2000 production levels, this would have necessitated that the livestock sector produce 27% of the estimated greenhouse gases emitted, appropriate 51% of biomass used, and mobilize 29% of the reactive nitrogen. In face of projected growth in the sector this will require that livestock production be 19% as greenhouse gas intensive per unit production than is anticipated under the FAO Projections scenario, 42% as biomass intensive, and mobilize only 21% of the reactive nitrogen per unit production (Table S9.13).

Table S9.13. Necessary efficiency gains in livestock sector to maintain year 2000 proportional contributions to greenhouse gas emissions, biomass appropriation and reactive nitrogen mobilization when anthropogenic activities are constrained to respect sustainability boundary conditions.

	GHG Emissions ($Gt\ CO_2$-e)	**Biomass Appropriation (Gt C)**	**Reactive Nitrogen (Mt N_r)**
Boundary Condition	8.9	9.7	35
Anthropogenic Contributions (2000)	34	18.7	121
Livestock Contributions (2000)	4.6	7.0	76
% Livestock of Total	14%	37%	63%
Livestock Relative to Boundary[1]	1.2	3.6	22
Necessary Efficiency Gain for 2000	283%	94%	245%

	GHG Emissions ($Gt\ CO_2$-e)	Biomass Appropriation (Gt C)	Reactive Nitrogen (Mt N_r)
Projected Livestock Contributions (2050)	6.4	8.5	103
Necessary Efficiency Gain (2050)	433%	136%	368%

(1) i.e. the proportional share of boundary condition GHG emissions, biomass appropriation and reactive nitrogen mobilization based on 2000 levels.

Given the magnitude of necessary efficiency gains, it would appear highly unlikely that technological improvements alone will be sufficient to achieve the objective of maintaining the proportional contribution of the livestock sector to cumulative anthropogenic contributions to these issues if cumulative contributions are constrained relative to sustainable boundary conditions. One alternative is to reduce the production and consumption of livestock products. Assuming the efficiency gains implied in our FAO Projections scenario, we estimate that the necessary reduction in terms of greenhouse gas emissions would be to 19% of projected 2050 production levels. For biomass appropriation, the necessary reduction would be to 42% of projected 2050 production levels. For reactive nitrogen mobilization, the necessary reduction would be to 21% of projected 2050 production levels.

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CHAPTER 10: ENVIRONMENTAL PERFORMANCE IN THE US BROILER POULTRY SECTOR: LIFE CYCLE ENERGY USE, ECOLOGICAL FOOTPRINT, AND GREENHOUSE GAS AND EUTROPHYING EMISSIONS

10.1 Publication Information

This paper is a modified version of a manuscript, reworked here for methodological consistency with subsequent analyses in Chapters 11 and 12, which was originally published as:

Pelletier N. 2008. Environmental performance in the US broiler poultry sector: Life cycle energy use and greenhouse gas, ozone depleting, acidifying and eutrophying emissions. Agricultural Systems 98:67-73. It is solely the work of the lead author.

10.2 Abstract

Most published research of the environmental impacts of broiler poultry production is limited to assessments of on-farm gaseous and nutrient emissions. Here, ISO-compliant Life Cycle Assessment was used to predict a subset of the broader, macroscale environmental impacts of the material and energy inputs and emissions along the US broiler supply chain. It was found that feed provision accounts for the majority of supply chain impacts, although litter management is also important to greenhouse gas and eutrophying emissions. In-barn energy use, largely related to heating and ventilation, contributes only a small share of these impacts. These results underscore the fallacy of "landless farming" and the importance of full supply-chain environmental management for improving sustainability in the US poultry industry.

10.3 Introduction

The US broiler poultry sector is both the largest and most efficient in the world, as well as the single largest meat producing industry in the United States. In 2007, close to 9 billion broilers were raised in the US, yielding 22.4 million live-weight tonnes with a farm-gate value of 21.5 billion dollars (USDA 2008a). At an annual per capita consumption of 39 kg, broiler chicken was also the most widely consumed meat, accounting for 38% of US meat consumption (USDA 2008b).

Although the low economic feed conversion ratios (ratio of total dry weight feed inputs to live weight poultry produced) achieved in the broiler sector make poultry production among the most efficient means of producing terrestrial animal protein (Flachowsky 2002), the sheer scale of this industry necessitates close attention to a range of potential environmental impacts. Numerous researchers have previously evaluated various aspects of environmental performance in broiler production, in particular point-source eutrophying emissions from poultry litter management (Pope 1991, Reynells 1991, Sims and Wolf 1994, De Boer *et al.* 2000), and gaseous emissions from poultry houses (Wathes *et al.* 1997, Chambers and Smith 1998, De Boer et al 2000, Ullman *et al.* 2004, Wheeler *et al.* 2006). However, the environmental performance of broiler production from a supply chain perspective has received relatively little attention. In particular, the broader, macroscale implications of the material and energetic intensity of US broiler production have been largely unaddressed.

Life Cycle Assessment (LCA) is an ISO-standardized environmental accounting framework used to inventory the material and energy inputs and emissions associated with each stage of a product life cycle, from resource extraction through processing, use and disposal, and to express these in terms of their quantitative contributions to a specified suite of environmental impact categories (Guinee *et al.* 2001). Such analyses facilitate the identification of life cycle stages that contribute disproportionately to specific areas of environmental concern, as well as comparisons of environmental performance between competing production technologies. This study used LCA to quantify the cradle-to-farm gate energy use (cumulative, human-mediated energy including renewable and non-renewable sources), ecological footprint, and greenhouse gas and eutrophying emissions associated with contemporary broiler poultry production in the United States.

The US broiler sector is dominated by fewer than 50 vertically integrated agribusiness firms, the top four of which are responsible for almost 50% of production (USDA 2002, Sambidi *et al.* 2003). These firms own or control breeder flocks, hatcheries, broiler flocks, feed mills, slaughter and processing plants, and transportation/distribution centers (Sambidi *et al.* 2003). As such, this industry structure may facilitate rapid and coordinated response to existing and emerging environmental issues. In particular, mounting concern regarding human-induced climate change may provoke regulatory measures and consumer pressure may also prove to be a powerful factor in influencing improved accountability for greenhouse gas emissions and other environmental impacts along the broiler supply chain. The purpose of this research is therefore to assist the broiler industry in targeting effective supply chain management for environmental performance, as well as inform appropriate regulatory initiatives.

10.4 Methods

10.4.1 Goal And Scope Definition

ISO-compliant Life Cycle Assessment methodology (Guinee *et al.* 2001) was used to estimate the average energy use, ecological footprint, and greenhouse gas and eutrophying emissions associated with the contemporary production of broiler poultry in the continental United States. The functional unit for this cradle-to-farm gate analysis was one live weight tonne of broiler poultry. The system boundaries for the analysis encompassed all direct inputs and emissions associated with the production of poultry feed ingredients (including fuel use for field operations and crop drying, the production of fertilizers/soil amendments, seed and pesticides, as well as nitrous oxide and ammonia emissions from fertilizers and crop residues for agricultural crops; crop processing; poultry processing and rendering to produce poultry fat and by-product meal; and reduction fisheries and fishmeal reduction plants), feed milling, hatchery chick production, on-farm energy use, litter management, and all associated transportation stages (Figure 10.1). Production, processing and transportation infrastructure (which previous LCA analyses of food systems have shown to make trivial contributions to supply chain impacts – for example, see Ayer and Tyedmers (2009)), feed additives (amino acids, pigments, etc.), maintenance of breeder flocks, hatchery wastes, and disposal of mortalities were not included in the analysis.

Co-product Allocation

In subsystems with multiple outputs it was necessary to partition environmental burdens between co-product streams. All co-product streams were allocated burdens based on gross chemical energy content. For example, in the case of soy bean processing, where one tonne of soy beans yields 200 kg of soy oil and 800 kg of soy meal with respective gross chemical energy contents of 39.3 and 17.1 MJ/kg (Sauvant *et al.* 2002), burdens for soy meal production were allocated according to the formula:

800 kg*17.1 MJ/kg / (800 kg*17.1 MJ/kg + 200 kg*39.3 MJ/kg) = 64%.

Soy meal was therefore allocated 64% of the burdens associated with soy bean production, transport and processing, while soy oil was allocated 36%.

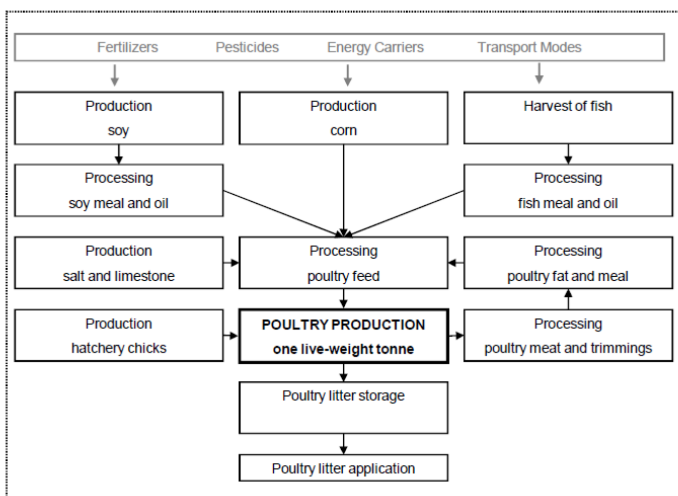

Figure 10.1. System boundaries for a life cycle assessment of US broiler poultry production (background processes such as fertilizers, pesticides, energy carriers and transport modes derived from the EcoInvent 2008 database).

10.4.2 Life Cycle Inventory

Product life cycle assessments require collecting data for all relevant material and energy inputs and emissions associated with each stage of the life cycle considered. Data for the cradle-to-farm gate life cycle of US broiler poultry were derived primarily from contacts in industry and academia, as well as peer-reviewed literature and government statistical publications (Table 10.1). Data for background systems including transportation, fertilizer and energy production were derived from the EcoInvent (2008) database and modified as appropriate to correspond to regional norms.

Feed Input Production And Processing

Inventory data for soy and corn feed inputs to poultry feed are derived from US National Agricultural Statistics Service (NASS) publications and peer-reviewed literature. Yields are based on 5-year averages for 2003-2007 calculated from NASS data. Fertilizer mixes correspond to average US consumption as reported by NASS. Application rates of pesticides and fertilizers used in soy and corn production are based on 2005 NASS data for the US. Energy inputs are based on US averages for 2001 (detailed inventory data provided in Table S10.1). Field-level ammonia, nitrous oxide, nitric oxide, nitrate and carbon dioxide (from urea fertilizers) emissions are calculated following IPCC (2006) Tier 1 protocols using relevant default emission factors. A 2.9% surplus phosphate emission rate is assumed following Dalgaard *et al.* (2008). All fertilizers and pesticides are assumed to be transported 1000 km by truck, and all seed inputs 100 km by truck. Soy beans are assumed to be transported 100 km by truck to processors. Corn is assumed to be transported 30 km to railheads. Feed inputs are assumed to be transported 1000 km to broiler production regions. Processing of crops and poultry processing materials follows Pelletier *et al.* (2009), adjusted to reflect the US electricity mix (Table S10.4). Production of fishmeal, which is assumed to be menhaden meal produced in the US, also follows Pelletier *et al.* (2009). Feed milling is based on average data collected from several US feed mills. Mineral input models are derived from the EcoInvent (2008) database.

In-barn Energy Use

Direct in-barn energy use associated with climate control, lighting, feed distribution and cleaning follows data collected from commercial scale broiler research facilities in Arkansas (Tabler, pers. comm.).

Manure Management

Manure mass is estimated following Chamblee and Todd (2002). Manure is handled as a solid, and is scraped and stockpiled in windrows within 300 m of the barn following each production cycle. All manure is assumed to be applied to fields within 5 km of farms. Energy use for transportation and application of manure follows EcoInvent (2008) database processes.

Nitrogen and phosphorus emission rates are calculated using a nutrient balance based on feed composition and assuming an N retention rate of 58% and a P retention rate of 50% following Kratz *et al.* (2004). Nitrogen excretion estimates are used to calculate direct nitrous oxide, ammonia and nitric oxide emissions from manure management and indirect nitrous oxide emissions from nitrate leaching following IPCC (2006) protocols and relevant Tier I and Tier II emission factors at time of deposition, storage and application. Methane emissions from enteric fermentation and manure management are calculated following IPCC (2006) Tier I protocols.

Table 10.1. Life cycle inventory data sources for US broiler poultry production.

Inventory Data	**Source**
Crops used in Poultry Feed	
- average yields and fertilizer/energy/crop protection inputs	NASS (2004, 2006)
- field-level nitrous oxide emissions	calculated as per IPCC (2006)
- inputs to crop processing	Pelletier *et al.* (2009)
Fishmeal used in Poultry Feed	
- inputs to fisheries and reduction plants for US menhaden meal	Pelletier *et al.* (2009)
Poultry Fat used in Poultry Feed	
- poultry processing and reduction of by-products for fat and meal	calculated as per Pelletier *et al.* (2009)
Feed Formulation (70% US corn, 20% US soy meal, 2.5% poultry	generic US industry-representative

Inventory Data	Source
by-product meal, 2.5% poultry fat, 2.5% US menhaden fish meal, 2.5% salt and limestone)	formulation based on consultations with feed mills and academic experts
Economic Feed Formulation (FCR) of 1.9	as above
Energy Use in Poultry Feed Mills[1]	Anon.
Energy Use in Hatcheries[2]	Anon.
Broiler Poultry Farm Energy Inputs[3]	Tabler (pers. comm. 2007)
Gaseous Emissions from Litter Management[4]	IPCC (2006)
Broiler Cycle 48 days, Slaughter Weight 2.26 kg	Tabler (pers. comm. 2007)
Poultry Litter	
- produced at rate of 650 kg/tonne of poultry produced	conservative estimate based on literature review[5]

1) Data for energy use in poultry feed mills represents average data collected from three companies (137 MJ of electricity and 294 MJ of natural gas/tonne of feed produced).

2) Data for energy use in hatcheries represents average data collected from three hatcheries employing a combination of single and multi-stage incubators (.18 MJ of electricity and .23 MJ of natural gas/chick produced).

3) Data for poultry farm energy inputs supplied by Tabler (pers. comm.. 2007) for a commercial-scale research facility at the University of Arkansas operating according to industry norms (64.8 kWh of electricity and 80.7 l of LPG/tonne of poultry produced). This facility is a four-house farm employing a range of industry-representative technologies, and setting 83,475 birds per cycle with 6 cycles per year and 5% mortality.

4) Data for gaseous emissions calculated following IPPC (2006) Tier 1 protocols and default emission factors.

5) For a range of reported values, see Chamblee and Todd (2002)

10.4.3 Life Cycle Impact Assessment

The impact categories considered in this study were cumulative energy use (MJ), ecological footprint (m^2 productive ecosystem), greenhouse gas emissions (CO_2 equiv.), and eutrophying emissions (PO_4 equiv). Energy use (MJ) is quantified following the Cumulative Energy Demand method (Frischnect *et al.* 2001), which accounts for conversion efficiencies and the quality of energy inputs. The ecological footprint, which quantifies the area of productive global ecosystem required to furnish the material and energy resources and sequester the greenhouse gas emissions associated with a product or service (in m^2 of productive ecosystem) is calculated following the EcoInvent 2.0 method (EcoInvent 2008). This method is modified to include methane and nitrous oxide emissions. Calculating the marine component of the ecological footprint which arises due to the use of fishmeal in poultry feeds follows the methods reported by Pelletier and Tyedmers (2007).Greenhouse gas emissions (CO_2- equiv.) (assuming a 100-year time horizon) and eutrophying emissions (PO_4^{-}equiv.) are quantified following the CML 2001 method (Guinee *et al.*, 2001). The SimaPro 7.1 life cycle impact assessment software package from PRe Consultants (PRe 2008) was used.

10.4.4 Life Cycle Interpretation

The total contributions to each impact category associated with the cradle-to-farm gate production of one live-weight kg of broiler poultry were evaluated, and the relative contributions of hatchery chick production, feed production, in-barn energy use, and litter management were calculated. The total impacts associated with producing one kg of feed and the relative contributions of individual feed components to total impacts are also determined.

In addition, the energy return on investment (EROI) ratios in the US broiler poultry production system are assessed in order to estimate the ecological efficiency with which this industry provides valued outputs from a variety of perspectives. This includes: (a) the amount of human-edible food energy produced relative to the total industrial energy inputs required (an anthropocentric perspective on non-renewable resource use efficiency); (b) the amount of human-edible food energy produced relative to the amount of human-edible food energy consumed by the poultry (an anthropocentric perspective on renewable resource use efficiency); and (c) the amount of gross chemical energy produced relative to the gross energy consumption (an ecological perspective on renewable resource use efficiency). Finally, the total estimated energy use, ecological footprint, and greenhouse gas and eutrophying emissions associated with broiler production in the United States in 2007 are calculated.

10.5 Results

Feed provision contributes the majority of the predicted life cycle cradle-to-farm gate impacts associated with the US broiler poultry supply chain (Table 10.2). Specifically, feed provision accounts for 80% of overall supply chain energy use, 92% of the ecological footprint, 76% of greenhouse gas emissions, and 44% of eutrophying emissions. Litter management is responsible for a larger fraction of eutrophying emissions (56%). In barn energy use, largely related to heating and ventilation contributes, on average 8% across impact categories. The production of hatchery chicks contributes negligibly to all impact categories.

Table 10.2. Cradle-to-farm gate life cycle energy use and global warming, ozone depleting, acidifying and eutrophying emissions per kg of live-weight broiler poultry produced in the United States.

	Poultry Feed	Hatchery Chicks	In-barn Energy Use	Litter Management	Total
Energy Use (MJ)	13.2	0.4	2.9	<0.1	16.5
GHG Em. (kg CO_2 equiv.)	1.3	<0.1	0.2	0.2	1.7
Eutrophication (g PO_4 equiv.)	8.7	<0.1	<0.1	11.1	19.9
Ecological Footprint (m^2)	13.5	0.1	0.5	0.6	14.7

Given that feed production dominates contributions to most impact categories considered, it is worthwhile examining these background production systems in greater detail. Corn, with an inclusion rate of 70% by mass in the modeled feed, contributes on average 31% of the impacts per kg of broiler feed produced, while soy meal (which comprises 20% of the feed) contributes 11%. Fertilizer production is the major contributor to energy use in crop production, while field-level nitrous oxide emissions and nitrate/phosphate leaching are responsible for the greatest shares of greenhouse gas and eutrophying emissions respectively. In combination, poultry fat, poultry by-product meal and fishmeal (which together comprise 7.5% of the feed) contribute on average 53% across impact categories (Table 10.3). Feed milling contributes only 5%.

Table 10.3. Cradle-to-feed mill gate life cycle energy use, ecological footprint, greenhouse gas and eutrophying emissions per kg of feed produced for the US broiler poultry industry.

Input (fraction of total by mass)	**Energy Use (MJ)**	**GHG Em. (kg CO_2-e/kg)**	**Eutroph. Em. (g PO_4-e/kg)**	**Ecological Footprint (m^2)**
Corn (70%)	1.87	0.23	1.02	2.77
Soybean Meal (20%)	0.70	0.06	0.25	1.48
Poultry By-product Meal (2.5%)	1.18	0.12	1.17	0.87
Poultry Fat (2.5%)	2.09	0.21	2.07	1.54
Fish Meal (2.5%)	0.23	0.01	<0.01	0.42
Salt and Limestone (2.5%)	0.05	<0.01	<0.01	0.01
Feed Milling	0.78	0.04	<0.01	0.13
TOTAL	6.90	0.68	4.59	7.10

On an equivalent-mass basis, corn and soy meal generate a fraction of the impacts associated with poultry fat, poultry by-product meal and fishmeal (Table 10.4). More specifically, fishmeal production generates from 1.4-2.6 times the impacts associated with either of the crop ingredients, largely due to fuel inputs for fishing and the energy inputs and emissions associated with the reduction of fish into fishmeal and oil. The impacts of poultry by-product meal, the production of which relies on background poultry and crop production and processing, are 5-32 times those of crop production. Poultry fat, the highest impact ingredient, generates 8-57 times the impacts of crop production.

Table 10.4. Cradle-to-feed mill gate life cycle energy use, ecological footprint, and greenhouse gas and eutrophying emissions per kg of broiler feed ingredients produced in the United States.

Input	**Energy Use (MJ)**	**GHG Em. (kg CO_2-e/kg)**	**Eutroph. Em. (g PO_4-e/kg)**	**Ecological Footprint (m^2)**
Corn	2.68	0.33	1.46	3.96
Soybean Meal	3.48	0.30	1.23	7.42
Poultry By-product Meal	47.3	4.79	46.7	34.9
Poultry Fat	83.5	8.47	82.7	61.7
Fish Meal	9.12	0.55	2.88	10.3
Salt and Limestone	2.81	0.15	0.17	0.55

Human edible energy returns on industrial energy investment were 15.7%. Human edible energy returns on edible energy investment ratios were 11%. Gross chemical energy returns relative to gross chemical energy consumed by the poultry were 17.5% (Table 10.5).

Table 10.5. Energy return on investment ratios for US broiler poultry production as (a) human edible caloric energy return on industrial energy investment (b) human edible caloric energy return on human edible caloric energy investment and (c) gross chemical energy return on gross chemical energy investment.

Industrial Energy[1]	**Human Edible Energy**[1]	**Gross Chemical Energy**[2]
15.7%	11.0%	17.5%

1) Assumes 56% yield of boneless meat per live-weight kg produced and an energy density of 4.63 MJ/kg of raw, boneless poultry.

2) Assumes a whole-animal energy density of 4.63 MJ/kg.

Based on the modeled system and national production of 22.4 million live-weight tonnes for 2007, the US broiler sector was underpinned by an estimated 296 billion MJ of energy (the equivalent of 8.3 billion litres of crude oil). This is greater than the annual energy consumption in 2007 for 133 of the 225 regions/countries for which data is reported by the US Energy Information Administration (EIA 2009). Similarly, the sector generated 38.1 million tonnes of CO_2 equiv. greenhouse gas emissions, which is more than the national emissions of 10 out of 39 Annex 1 countries for which data was reported to the UNFCC for 2007 (UNFCC 2009), including Norway. Total eutrophying emissions amounted to almost 450 thousand tonnes of PO_4-equiv. The ecological footprint for the sector was 330 thousand km^2, which is equal to 4% of the surface area of the continental United States.

10.6 Discussion

Most published research regarding the potential environmental impacts of broiler production focuses on farm-specific emissions from poultry houses or litter management (Pope 1991, Reynells 1991, Sims and Wolf 1994, Wathes *et al.* 1997, Chambers and Smith 1998, De Boer et al 2000, Ullman *et al.* 2004, Wheeler *et al.* 2006). However, on-farm broiler production is but one step in a complex series of interlinked industrial activities that together comprise the broiler supply chain. Understanding the environmental impacts of broiler production must therefore involve careful consideration of each step in this process. In particular, broiler production is fundamentally dependent on concentrate feed derived from crop and other feed input production systems, processing and transportation links typically far removed from the farm itself. Indeed, from a life cycle perspective, it would appear that upstream feed production processes are responsible for the bulk of macroscale, environmental impacts associated with material and energy inputs and emissions along the broiler supply chain. This insight is consistent with previous analyses of other animal husbandry and aquaculture production systems (Carlsson-Kanyama 1998, Cederberg and Mattson 2000, Basset-Mens and van der Werf 2005, Ellingsen and Aanondson 2006, Pelletier and Tyedmers 2007, Thomassen *et al.* 2008, 2009 a,b,c).

In an analysis of pork production in Sweden, Carlsson-Kanyama (1998) found that the production of concentrate feed contributed 53% of greenhouse gas emissions, and 70% of energy use. Basset-Mens and van der Werf (2005) also found that feed production was an environmental hotspot in pork production. Life cycle assessment research of milk production has similarly indicated that feed production is the major contributor to most impact categories (Cederberg and Mattson 2000, Thomassen *et al.* 2008). For example, in a recent comparison of conventional and organic milk production in the Netherlands, Thomassen *et al.* (2008) found that the production of concentrate feeds for conventional dairy farms produced the highest contributions to all impact categories considered. In a similar analysis in Sweden, Cederberg and Mattson (2000) also identified concentrate feed production, and the use of synthetic fertilizer in feed crop production in particular, as a central factor in the environmental performance of conventional dairy farming. The contribution to eutrophying impacts associated with field-level nitrogen and phosphorous leaching was particularly noteworthy. Casey and Holden (2005) reported that, although methane emissions from enteric fermentation in Irish dairy production accounted for 49% of greenhouse gas emissions, the off-farm production of concentrate feeds and on-farm grass cultivation contributed 34% to total emissions. In contrast, on-farm manure management contributed only 11% to emissions, and on-farm energy use only 5%. An Irish study of suckler-beef production yielded similar results (Casey and Holden 2006). Ellingsen and Aanondsen (2006) compared the life cycle impacts of chicken, farmed salmon and wild-caught cod production in Norway and found that feed production dominated supply chain impacts for both chicken and salmon farming. Several analyses of aquaculture production systems have yielded comparable results, with feed production typically accounting for the majority of resource use and emissions-related environmental impacts (Pelletier and Tyedmers 2007, Pelletier *et al.* 2009). Clearly, understanding and ameliorating the environmental impacts of feed provision is central to improving the environmental sustainability of broiler production as a whole.

For the crop-derived feed components (soy meal and corn), impacts were predominantly associated with the agricultural rather than transportation or processing phases. More specifically, field-level nitrous oxide and nitrate emissions dominated greenhouse gas and eutrophying emissions, while the production of fertilizers (in particular, nitrogen fertilizer) was also important for greenhouse gas emissions and the dominant factor in energy use. These findings complement a broad body of published work (for example, see Williams *et al.*, 2006, Hoeppner *et al.* 2006, Pimentel *et al.* 2005). Although not analyzed in this study, the use of organic feed ingredients, which are typically less energy and emissions intensive due to the disallowance of synthetic fertilizers in their production (Pelletier *et al.* 2008), may potentially offer a viable means of much reducing the life cycle impacts of broiler production.

On an equivalent mass basis, crop-derived feed ingredients generate a fraction of the impacts associated with fishmeal and, in turn, correspondingly less than poultry fat and poultry by-product meal. For this reason, even though used in relatively small quantities (7.5% inclusion rate), the animal-derived material included in feed production contributes a disproportionate share of environmental burdens in the broiler supply chain. These findings are consistent with research of feed production for salmon aquaculture (Ellingsen and Aanondsen 2006, Pelletier and Tyedmers 2007, Pelletier *et al.* 2009).

This is not simply an artifact of accounting methodology. In this analysis, environmental burdens were partitioned between co-products of multi-output systems on the basis of gross chemical energy content (see Ayer *et al.* 2007). For example, in the production of soy meal and oil from soy beans, each co-product was allocated a share of the upstream impacts of soy production according to gross energy content. Poultry meat and the processing trimmings used to produce poultry fat and poultry by-product meal were similarly allocated burdens according to the same principle. For this reason, poultry fat and by-product meal derived from poultry processing trimmings carry a share of the total impacts associated with broiler production (including crop production, processing, transportation and on-farm inputs and emissions), as well as some of the burdens of processing and reduction. Since poultry production requires 1.9 tonnes of feed per tonne of poultry produced, and large volumes of poultry processing trimmings are required to produce poultry fat and poultry by-product meal (1 tonne of trimmings yield 260 kg of by-product meal and 100 kg of poultry fat), the environmental burdens associated with producing these materials is considerable.

While other allocation criteria may be used (for example mass-based allocation, or allocation based on the relative economic value of co-product streams), it was thought that gross energy content best reflects the actual flows of biologically valuable material and associated environmental impacts (see Pelletier *et al.* 2007). Moreover, using economic allocation would result in even higher attribution of impacts to broiler meat were the analysis carried through to processed products since, at processing, the primary product (meat) would be assigned a much larger share of the impacts than the economically less valuable processing trimmings. Such an approach would also tend to produce results that simply reflect the distortions inherent in market prices (which externalize many of the environmental impacts of products and services) rather than logical, biophysical relationships.

Modern broiler production and other intensive animal husbandry operations are often referred to as "landless farming." However, as demonstrated by this analysis, understanding the full environmental implications of animal protein production requires a systemic perspective that includes all aspects of the supply chain and, in particular, activities related to the provision of feed. Indeed, the estimation that feed crop production for intensive animal husbandry accounts for 33% of total arable land use (Steinfeld *et al.* 2006) demonstrates well the fundamental and considerable dependence of so-called landless farming on human activities and ecosystem goods and services far removed from the location of the farm itself.

Amongst animal protein production systems, broiler poultry production is considered relatively efficient (Flachowsky 2002). Compared to similar evaluations for feedlot-finished beef production in the upper Midwestern United States (Pelletier *et al. in press*), the US broiler poultry system produces EROI ratios 2.6-8.8 times higher than beef. Compared to commodity pork production in this same region (Pelletier *et al. in review*), poultry produces lower human-edible returns on industrial energy investment (15.7% versus 26.7%), but higher returns for both human-edible energy returns on investment (11% versus 4.2%) and gross chemical energy returns on investment (17.5% versus 13.1%). On the other hand, some fuel-efficient fisheries and low trophic-level aquaculture technologies may perform better still (Troell *et al.* 2004). Moreover, vegetable protein production generates a fraction of the impacts associated with terrestrial animal protein production. After all, the inefficiencies inherent in biological feed conversion dictate that any animal husbandry system will necessarily consume more energy (in crops) than it generates (in animal products) (Goodland 1997, CAST 1999, Delgado *et al.* 1999, Zhu and Ierland 2004, Pelletier and Tyedmers 2007), as well as compound the associated impacts. What emerges from these considerations are key societal questions regarding the optimal means of satisfying human wants and needs in a sustainable manner, and to what extent animal protein can/should contribute. Regardless of relative efficiencies compared to other meat producing industries, the sheer scale of the US broiler poultry sector means that associated impacts are considerable, with energy and emission intensities exceeding those of numerous countries. The promotion of plant protein consumption over animal protein is therefore an obvious suggestion for reducing the environmental impacts of food production generally (Goodland 1997, Carlsson-Kanyama 1998, Zhu and Ierland 2004).

Although efforts have been made to achieve the most complete analysis possible, the generic nature of the broiler production system modeled as well as limitations in data availability inevitably result in a simplified representation of the US broiler supply chain. For example, since broiler poultry farms are located throughout the US (with high concentrations in certain states) it was not feasible to arrive at a concrete representation of the average distance traveled by feed ingredients (which also derive from various states and travel variable distances from farms to distribution centers to processors, etc.). Rather, average distances and transport modes were assumed based on distances between major feed ingredient production regions and major poultry production regions. It should also be noted that the values reported here are likely conservative as the cumulative contributions of seemingly minor omissions may be non-trivial. Moreover, the environmental performance of specific supply chains will almost certainly differ according to the particulars of management, efficiency and logistics. However, the general insights derived from this analysis are relevant across broiler supply chains. As such, although addressing only a limited number of the broad range of the potential environmental impacts of broiler production (and animal protein production, generally), this information can and should be used in complement with existing management tools by industry and regulators to improve supply chain environmental management for the impact categories considered.

10.7 Acknowledgements

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10.9 Supporting Information

Table S10.1. Inputs and outputs per tonne of crop produced as feed inputs for broiler production in the United States.

INPUTS	**Corn**	**Soy**
Fertilizer (kg)		
N	17.1	2.4
P2O5	7.2	7.5
K2O	10.4	11.8
Energy		
Diesel (l)	6.4	14.2
Gas (l)	1.8	4.5
LPG (l)	4.9	1.4
Elect. (kWh)	11.4	7.2
Herb/Pesticides[1] (kg)	0.3	0.5
Seed (kg)	2.6	45
OUTPUTS		
Nitrous Oxide (kg)	0.5	0.3
Ammonia (kg)	2.5	2.8
Nitric Oxide (kg)	0.3	<0.1
Carbon Dioxide[2] (kg)	3.3	0.5
Nitrate (kg)	2.5	0
Phosphate (kg)	<0.1	0
Yield (tonnes)	9.1	2.7

(1) Active ingredients.

(2) From lime and urea fraction of N fertilizer as per IPCC (2006).

CHAPTER 11: LIFE CYCLE ASSESSMENT OF HIGH- AND LOW-PROFITABILITY COMMODITY AND NICHE SWINE PRODUCTION SYSTEMS IN THE UPPER MIDWESTERN UNITED STATES

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11.1 Publication Information

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11.2 Abstract

We used ISO-compliant life cycle assessment to evaluate the comparative environmental performance of high- and low-profitability commodity and niche swine production systems in the Upper Midwestern United States. Specifically, we evaluated the contributions of feed production, in-barn energy use, manure management, and weaner production to farm-gate life cycle energy use, ecological footprint, and greenhouse gas (GHG) and eutrophying emissions per animal produced and per live-weight kg. We found that commodity systems generally outperform niche systems for these criteria, but that significant overlap occurs in the range of impacts characteristic of high- and low-profitability production between systems. Given the non-optimized status of current niche relative to commodity production, we suggest that optimizing niche systems through improvements in feed and sow herd efficiency holds significant environmental performance improvement potential. Drivers of impacts differed between commodity and niche systems. Feed production was the key consideration in both, but proportionally

more important in niche production due to lower feed use efficiencies. Liquid manure management in commodity production strongly influenced GHG emissions, whereas solid manure management increased eutrophication potential due to outdoor storage in niche production. We further observe an interesting but highly imperfect relationship between economic and environmental performance measures, where profitability tracks well with resource (in particular, feed) throughput, but only indirectly with emissions intensity.

11.3 Introduction

The role of food systems in anthropogenic environmental change is a subject of concern for policy-makers and regulators, the food industry, environmental advocacy groups, and the general public alike. The cumulative contributions of current consumption patterns to resource depletion and waste emission issues (including acid precipitation, ozone depletion, eutrophication, and climate change) are attracting increasing attention. So, too, are the comparative resource and waste emission intensities of alternative food products and food production strategies. Clearly, ensuring the long-term sustainability of local, regional and global food systems requires attention to both scale and efficiency issues. However, appropriate mitigation strategies must be context-sensitive, and be formulated with attention to trade-offs between potentially conflicting social, economic and environmental objectives.

Life cycle assessment is an ISO-standardized biophysical accounting framework used to (1) characterize the material/energy flows underpinning specific activities and (2) quantify their contributions to resource depletion and emissions-related environmental concerns. This framework is not well-suited to the consideration of proximate ecological concerns nor the myriad socio-economic concerns associated with the management of human activities. It does, however, provide a relatively nuanced means of quantifying the requisite resource provisioning and waste assimilatory services upon which they depend, and has a strong history as an eco-efficiency tool.

Pork is currently the most widely consumed meat product, accounting for 37% of meat consumption in developed countries and close to 40% worldwide (FAO 2006). The United States is the second largest pork producer globally following China, and the second largest exporter after Denmark (FAOstat 2009). Within the US, a substantial share of pork is produced in the Upper Midwest. Iowa is the leading producer state, accounting for 28% of live-weight production in 2007 (NASS 2009).

The majority of contemporary pork production in the Upper Midwest comes from large, high-volume farms operating climate-controlled, slatted floor barns and producing over 50,000 pigs annually. A much smaller share of production is contributed by "niche" producers using alternative rearing systems. Although each niche is unique, farmers participating in these production chains often follow a series of guidelines and protocols as described by Honeyman *et al.* (2006). In general producers provide bedding and follow phase-specific space allowance guidelines. The use of farrowing crates, gestation stalls, sub-therapeutic antibiotics, and rendered animal protein is often prohibited. Such operations typically produce, on average, 500-600 pigs per year. Both researchers and consumers have expressed interest in considerations of environmental performance, animal welfare, food safety, and ownership structure associated with these different pork production strategies (Honeyman *et al.* 2006).

A number of European researchers have previously applied LCA to evaluate the environmental performance of alternative pork production systems (Zhu and Ireland 2004; Bassett-Mens and van der Werf 2005; Ericksson *et al.* 2005; Stern *et al.* 2005; Williams *et al.* 2006; Dalgaard *et al.* 2007). More recently, Lammers and colleagues (2009 a,b,c) report the direct and embodied energy and greenhouse gas emissions associated with Iowa pork supply chains using conventional versus hoop production facilities. No full LCAs of US pork production technologies have been reported to date.

As pointed out by Lammers and colleagues (2009c), US pork production systems are in many ways distinct from European systems, with differences in feeds, housing, and management strategies. Moreover, survey data of costs, profitability and animal performance in the US industry suggest considerable variability within and between US swine production strategies. The objective of the current analysis is therefore to complement and expand upon previous work in the US and elsewhere through the application of ISO-compliant life cycle assessment methods to characterize the cradle-to-farm gate cumulative energy demand, ecological footprint, and global warming and eutrophying emissions associated with commodity and niche pork production systems in the Upper Midwestern United States. In light of intra- and inter-production strategy differences, we further focus on elucidating the comparative performance of low- (LP) and high-profitability (HP) production systems. In concert with existing information and performance indicators, this work should contribute to furthering our understanding of a sub-set of sustainability considerations in pork production in this region, as well as the industry as a whole. It should also provide insights into the relationships between profitability and environmental performance in pork production.

11.4 Methods

Our analysis employs ISO-compliant life cycle assessment methods (ISO 2006) to characterize the cradle-to-farm gate flows of material and energy inputs, outputs, and a sub-set of the resultant waste emissions associated with conventional and niche swine production strategies in the Upper Midwestern United States. Towards this end, we consider and compare a production system representing contemporary high-volume commodity production norms with a deep-bedded niche swine production system as described by Honeyman (2005). Specifically, we quantify the cumulative energy use (MJ), ecological footprint (m^2 of productive ecosystem), greenhouse gas (kg CO_2-equiv.) and eutrophying (g PO_4-equiv.) emissions for weaner production and finishing pigs, and per kg of total live-weight pig production in each system.

11.4.1 Life Cycle Inventory Methods, Data And Assumptions

To account for the variability in economic and biological performance (and attendant environmental consequences) achievable within specific production strategies, we characterize production norms for both the most and least profitable (per unit live-weight production) 20% of commodity and niche farms for which we were able to access standardized data. Our estimates of animal performance for niche operations are based on data compiled by Stender and colleagues for the Iowa Pork Industry Council (IPIC 2009) for 41 Iowa swine farms surveyed in 2006. Our data for animal performance in conventional production systems is derived from 2006 survey data for 39 farms in southern Minnesota as reported in the Farm Financial Database (FINBIN 2009). Where possible, we build on the research of Iowa commodity pork production systems operating conventional and hoop facilities as reported by Lammers and colleagues (2009 a,b,c) – particularly with respect to swine diets, and farm-level material and energy use. Remaining data gaps are populated from published literature sources and recommendations provided by swine researchers.

The system boundaries for our analysis encompass the production, processing and transportation of all material and energy inputs to swine feeds, direct in-barn resource flows, and supply chain gaseous and nutrient emissions associated with the live-weight production of slaughter-ready swine at the farm gate. We do not consider the production and maintenance of capital goods, labour inputs or chemotherapeutants (Figure 11.1).

Commodity Production System

The commodity production system we model is representative of approximately 50% of high-volume swine production in the Upper Midwestern United States. Our model assumes a 2400-sow unit producing 40-50,000 weaned pigs annually. Three distinct husbandry phases, each with characteristic diets, housing, and material/energy inputs, are considered. These are a sow gestation phase, a sow farrowing/lactation phase, and a finishing phase. Breeding and gestation occurs in individual stalls, and farrowing/lactation in individual farrowing crates. We assume that weaned pigs are produced and finished on site, where they move directly from weaning to finishing. Finishing pigs are housed in pens in barns with a maximum capacity of 1,200 pigs. Climate-controlled barns with slatted floors are used for all phases (as described in Lammers *et al.* 2009a**)**. It is further assumed that feeds are milled on site.

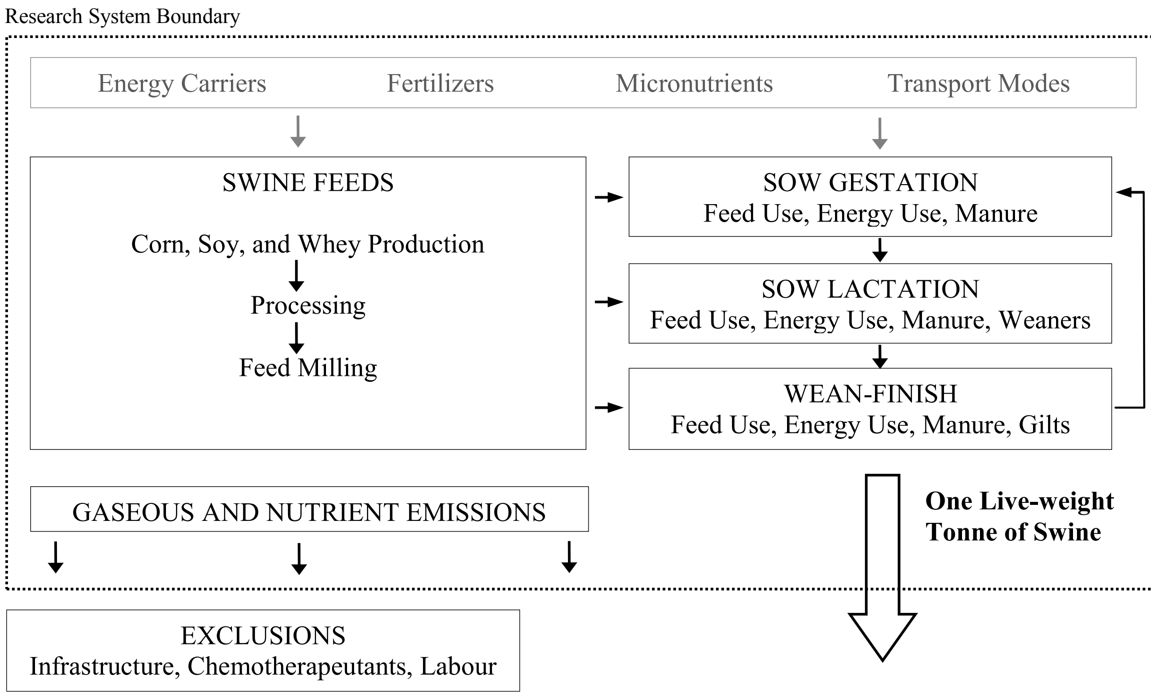

Figure 11.1. System boundaries for a cradle-to-farm gate LCA of live-weight swine production in high- and low-profitability commodity and niche production systems in the Upper Midwestern United States (background system data for energy carriers, fertilizers, micronutrients and transport modes derived from the EcoInvent 2008 database, modified as appropriate to conform to regional conditions).

Niche Production System

The niche production system we model is characteristic of a very small fraction of swine production in the US Upper Midwest. Our model for niche production similarly comprises a sow gestation phase, a sow lactation phase and a finishing phase. Gestation occurs in bedded group pens with individual feed stalls. Farrowing/lactation take places in bedded, insulated buildings, with heaters and heat lamps for winter climate control. We assume a 60 sow unit producing roughly 600-650 weaned pigs per year. Finishing pigs are housed in deep-bedded hoop barns with a maximum capacity of 200 pigs.

Manure Management

In the commodity system, manure is handled as slurry in a central collection pit beneath the finishing barn. Manure volume is calculated based on estimated daily manure excretion rates (ISU 1995) and cleaning water use (Lammers *et al.* 2009b). For niche production, manure is handled as a solid, and is scraped and stockpiled in windrows within 300 m of the barn. For the purpose of calculating manure handling inputs for niche systems, bedding mass is added. All manure is applied to fields within 5 km of commodity farms and 1 km of the niche farms once yearly. Energy use for transportation and application of manure follows EcoInvent (2008) database processes. For niche production, we assume a 37% mass reduction for stockpiled, unturned manure (prior to spreading) following Tiquia *et al.* (2002).

Nitrogen and phosphorus emission rates are calculated using a nutrient balance based on feed composition and assuming that 2.42% of swine body mass is nitrogen and 0.55% is phosphorus following Koelsh and Lesoing (1999). Nitrogen excretion estimates are used to calculate direct nitrous oxide, ammonia and nitric oxide emissions from manure management and indirect nitrous oxide emissions from nitrate leaching following IPCC (2006) protocols and relevant Tier I and Tier II emission factors at time of deposition, storage and application. Methane emissions from enteric fermentation and manure management are calculated following IPCC (2006) Tier I protocols (for the emission factors employed in this analysis, see Table S11.2).

Feed Input Production And Processing

We assume the use of the same feeds for commodity and niche production. Feed input models and data follow those described in Pelletier *et al.* (*in press*). Inventory data for soy and corn-based feed inputs are derived from US National Agricultural Statistics Service (NASS) publications, Iowa State University extension publications and peer-reviewed literature. Yields are based on 5-year averages for 2003-2007 calculated from NASS data. Fertilizer mixes correspond to average US consumption as reported by NASS. Application rates of pesticides and fertilizers used in soy and corn production are based on 2005 NASS data for Iowa. Energy inputs are based on Iowa averages for 2001 (detailed inventory data provided in Tables S11.3-S11.4). Field-level ammonia, nitrous oxide, nitric oxide, nitrate and carbon dioxide (from urea fertilizers) emissions are calculated following IPCC (2006) Tier 1 protocols using relevant default emission factors. A 2.9% surplus phosphate emission rate is assumed following Dalgaard *et al.* (2008). All fertilizers and pesticides are assumed to be transported 1000 km by truck, and all seed inputs 100 km by truck. Both corn and soy are produced in high volumes in this region. Soy beans are assumed to be transported 100 km by truck to processors, and 100 km to farms. Corn is assumed to be transported 30 km to farms, reflecting the higher volume of corn production in Iowa. Processing of soy beans into meal and oil applies inventory data reported by Schmidt (2007), adjusted to reflect the US electricity mix (Table S11.4). Energy use for in-barn feed mixing, milling and distribution follows Lammers *et al.* (2009c).

To characterize whey production, inventory data for milk production are derived from Arsenault *et al.* (2009) and milk processing from Feitz *et al.* (2007). Inventory data for DL-methionine production reported by Binder (2003) are assumed to be representative of all synthetic amino acids used in the modeled swine feeds. Mineral input models are derived from the EcoInvent (2008) database.

In-barn Material And Energy Use

Direct in-barn material and energy use associated with climate control, lighting, feed distribution and cleaning in conventional and hoop swine production facilities in Iowa has previously been characterized by Lammers and colleagues (2009a,b). Our models draw directly on this work, modifying specific data points as appropriate to conform to the model assumptions of the present analysis (Tables 11.6-11.8). For bedding use in the niche systems, we assume the use of corn-stalks. Since these are returned to the fields after use, we account for the collection and transportation of bedding between fields and farms, but not for agricultural production following Lammers *et al.* (2009b).

Co-product Allocation

Co-product allocation is required to apportion resource use and emissions between the products of multi-output systems. Since the purpose of the present analysis is to describe the biophysical environmental dimensions of a food production system, it is appropriate to base allocation decisions on an inherent biophysical characteristic of co-products which both reflects the efficiency of the process and is relevant to the underlying causal impetus of the production system. To this end, the gross chemical energy content of co-product streams will be used as the basis for all allocation decisions because (1) producing caloric energy is the root driver of all food production activities and (2) the chemical energy of food products present in raw materials is apportioned between processed outputs in a quantifiable manner which speaks directly to the efficiency with which the system provides available food energy. For a detailed discussion of this rationale, see Ayer *et al.* (2007) and Pelletier and Tyedmers (2007). For the present analysis, allocation is required for soy bean processing and whey production only.

11.4.2 Life Cycle Impact Assessment

Impact assessment in LCA involves calculating the contributions made by the material and energy inputs and outputs tabulated in the inventory phase to a specified suite of environmental impact categories. We consider two resource use impact categories (energy use and ecological footprint) and two emissions-related categories (greenhouse gas emissions and eutrophying emissions). All impacts are calculated using the SimaPro 7.1 LCA software package from PRé Consultants (PRé 2008). Energy use (MJ) is quantified following the Cumulative Energy Demand method (Frischnect *et al.* 2001), which accounts for conversion efficiencies and the quality of energy inputs. The ecological footprint, which quantifies the area of productive global ecosystem required to furnish the material and energy resources and sequester the greenhouse gas emissions associated with a product or service (in m^2 of productive ecosystem) is calculated following the EcoInvent 2.0 method (EcoInvent 2008). This method is modified to include methane and nitrous oxide emissions. We believe this indicator to be more relevant than simple measures of land-use, which typically reflect occupation only rather than the quality of land-use and its relation to biocapacity considerations. Greenhouse gas emissions (CO_2- equiv.) (assuming a 100-year time horizon) and eutrophying emissions (PO_4^{-}equiv.) are quantified following the CML 2001 method (Guinee *et al.*, 2001). These assessment methods follow the problem-oriented mid-point approach, meaning that results are expressed in terms of total resource use and emissions rather than measured impact levels.

11.4.3 Life Cycle Interpretation

Impacts are calculated for producing individual weaners and finished pigs in each system, as well as per kg of total live-weight production. Cradle-to-farm gate supply chain impacts are assessed to identify impact hotspots and key leverage points for environmental performance improvements. Comparative impacts between production systems are evaluated and relationships to profitability are assessed.

We also assess the energy return on investment (EROI) ratios in commodity and niche swine production systems in order to estimate the ecological efficiency with which these competing swine production technologies provide valued outputs from a variety of perspectives. Specifically, we evaluate: (a) the amount of human-edible food energy produced relative to the total industrial energy inputs required (an anthropocentric perspective on non-renewable resource use efficiency); (b) the amount of human-edible food energy produced relative to the amount of human-edible food energy consumed by the swine (an anthropocentric perspective on renewable resource use efficiency); and (c) the amount of gross chemical energy produced relative to the gross energy consumption of swine in each scenario (an ecological perspective on renewable resource use efficiency).

11.5 Results

11.5.1 Life Cycle Inventory Results

Feed use is consistently higher for low-profitability (LP) compared to high-profitability (HP) systems for both commodity and niche production, with the exception of lactation feed use in the commodity systems. It is also consistently higher for niche production, with no overlap between HP niche and LP commodity systems in any of the phases considered. Our life cycle inventory analysis further suggests larger differences in feed consumption between high – and low-profitability niche swine production systems than are seen within commodity systems (Tables S11.6-11.7). As a result, since feed composition is the same between commodity and niche systems, estimates of protein, nitrogen and phosphorus intake are correspondingly higher for niche systems (Table S11.8).

Performance data for commodity and niche gestating and lactating sows suggest distinct differences related to respective production strategies. Annual output of weaners is much higher from commodity systems because sows have more litters, the weaning time is shorter, and piglets are weaned at lower weights. Sow weight gain and weight at replacement is also lower in the commodity systems. The highest mortality and replacement rate is seen in the LP commodity systems. For most variables considered, there is less difference between high- and low-profitability niche systems compared to commodity systems for the gestation and lactation phases (Table 11.1).

Table 11.1. Performance of gestating and lactating sows in high and low profitability commodity and niche sow herds in Iowa. All data from Stender *et al.* (IPIC 2009) for niche sows and FinBin (2009) for commodity sows unless otherwise noted.

Performance Criteria	Commodity (High)	Commodity (Low)	Niche (High)	Niche (Low)
Starting Weight[1] (kg)	118.8	110.5	126.0	123.7
Annual Gain (kg)	27.2[2]	41.1[2]	53.6	57.7
Litters/Year	2.2	1.97	1.6	1.6
Piglets Weaned/Year	21.32	16.94	10.9	10.2

Performance Criteria	Commodity (High)	Commodity (Low)	Niche (High)	Niche (Low)
Weaner Weight (kg)	6.8	6.4	15.9	12.7
Weaning Time (days)	21	21	42	42
Sow Mortality (%)	4.9	13.4	4.9	6.8
Replacement Rate (%)	38.3[3]	77.8[3]	63.8	65.9
Weight at Slaughter (kg)	204.5	170.5	238.6	215.9

(1) Replacements assumed to be derived from same production system.

(2) Calculated

(3) Pig Champ (2006)

Niche pigs finish at slightly higher weights than commodity pigs, and have substantially longer finishing times. Feed:gain ratios for niche production are notably higher, with HP niche production evincing a feed:gain ratio 31% higher than that observed in LP commodity production and 49% higher than HP commodity production. Mortality in HP niche production is close to that of HP commodity production (Table 11.2).

Table 11.2. Performance of finishing pigs in high and low profitability commodity and niche swine herds in Iowa. All data from Stender *et al.* (IPIC 2009) for niche herds, data for commodity herds from FinBin (2009) unless otherwise noted.

Performance Criteria	Commodity (High)	Commodity (Low)	Niche (High)	Niche (Low)
Starting Weight (kg)	6.8	6.4	15.9	12.7
Finishing Weight (kg)	118.8	110.5	126.0	123.7
Total Gain (kg)	112.0	104.1	110.1	111.0
Feed:Gain Ratio	2.44	2.77	3.63	3.96
Mortality (%)	6.7	9.2	7.8	13.7
Finishing Time (days)	130	141	192	203
Cycles Per Year[1]	2.70	2.50	1.85	1.75

(1) Calculated. Includes 5 day turn-around between cycles

With respect to farm-level material and energy inputs, the distinct differences between commodity and niche herds reflect differences in housing strategies; for example the use of cleaning water for commodity production compared to bedding for niche production, or greater energy inputs in the commodity system for climate control (Tables 11.3-11.5).

Table 11.3. Material and energy inputs/output associated with the maintenance of gestating sows in high and low profitability commodity and niche sow herds in Iowa.

INPUTS	Commodity (High)	Commodity (Low)	Niche (High)	Niche (Low)
Feed Per Sow[1] (kg/head/day)	2.27	2.27	2.61	4.19
Drinking Water[2] (l/head/day)	16.0	16.0	16.0	16.0
Cooling/Cleaning Water[2] (l/space/year)	138	138	0	0
Energy[2] (MJ/space/year)				
Electricity	201.1	201.1	39.9	39.9
LPG	497.5	497.5	0	0
Diesel	45.1	45.1	69.3	69.3
Cornstalk Bedding[3] (kg/space/year)	0	0	730	730
OUPUTS				
Manure [4]				
commodity (litres/head/day)	6.1	6.1		
niche (kg/head/day)			6.9	6.9

(1) Stender *et al.* (IPIC 2009) for niche and FinBin (2009) for commodity

(2) Calculated as per Lammers *et al.* (2009b)

(3) Harmon *et al.* (2004)

(4) ISU (1995). Cleaning water is added to reported commodity manure production volume for calculating inputs to manure management, whereas bedding is added to reported niche manure production mass.

Table 11.4. Material and energy inputs/output associated with the maintenance of lactating sows in high and low profitability commodity and niche sow herds in Iowa.

INPUTS	Commodity (High)	Commodity (Low)	Niche (High)	Niche (Low)
Feed Per Sow[1] (kg/head/day)	5.19	5.14	4.28	6.34
Drinking Water[2] (l/head+litter/day)	35	35	35	35
Cooling/Cleaning Water[2] (l/space/year)	1,143	1,143	0	0
Energy[2] (MJ/space/year)				
Electricity	3,198.0	3,198.0	105.3[3]	105.3[3]
LPG	1,359.4	1,359.4	0	0
Diesel	278.6	278.6	52.6[3]	52.6[3]
Cornstalk Bedding (kg/space/year)	0	0	730	730
OUPUTS				
Manure[4] (litres/head+litter/day)				

commodity (litres/head/day)	13.2	13.2		
niche (kg/head/day)			15.3	15.3

(1) Stender *et al.* (IPIC 2009) for niche and FinBin (2009) for commodity

(2)Calculated as per Lammers *et al.* (2009b)

(3) Energy inputs for niche lactating sows are expressed per litter rather than per space/year.

(4) ISU (1995) Cleaning water is added to reported commodity manure production volume for calculating inputs to manure management, whereas bedding is added to reported niche manure production mass.

Table 11.5. Material and energy inputs/output associated with the maintenance of finishing pigs in high and low performing commodity and niche swine herds in Iowa.

INPUTS	**Commodity (High)**	**Commodity (Low)**	**Niche (High)**	**Niche (Low)**
Feed Per Pig[1] (kg/head/day)	2.17	2.14	2.22	2.34
Drinking Water[2] (l/head/day)	10	10	10	10
Cooling/Cleaning Water[2] (l/space/year)	137	137	0	0
Energy[2] (MJ/space/year)				
Electricity	86.8	86.8	6.5	6.5
LPG	225.7	225.7	0	0
Diesel	51.1	51.1	20.8	20.8
Cornstalk Bedding[3] (kg/head)	0	0	91	91
OUPUTS				
Manure[4]				
commodity (litres/head/day)	5.30	5.30		
niche (kg/head/day)			5.0	5.0

(1) Stender *et al.* (IPIC 2009) for nice and FinBin (2009) for commodity

(2) Calculated as per Lammers *et al.* (2009b)

(3) Brumm *et al.* (2004)

(4) ISU (1995) Cleaning water is added to reported commodity manure production volume for calculating inputs to manure management, whereas bedding is added to reported niche manure production mass.

11.5.2 Life Cycle Impact Assessment Results

The attribution of life cycle impacts to specific aspects of producing weaned pigs (i.e. feed production, in-barn energy use, manure management, etc.) vary with impact category. For commodity production, in-barn energy use is the most important consideration for cumulative energy use, whereas GHG and eutrophying emissions are most strongly associated with manure management. Feed production is the dominant contributor to the ecological footprint of producing weaned pigs in the commodity system. In contrast, feed production is the most important contributor to all categories other than eutrophying emissions (manure management) in the production of weaned pigs in niche systems. High profitability operations have consistently lower impacts compared to low profitability operations for both commodity and niche weaner production (Table 11.9). On average, HP commodity systems have 70% of the impacts associated with LP commodity systems versus 63% for high compared to low profitability niche systems for weaner production. HP niche systems have higher energy use, euthrophying emissions and ecological footprint than LP commodity systems, but fall between high and low profitability systems in terms of GHG emissions. Overall, impacts per weaned pig produced are 86% higher for niche production, although it is important to note that niche weaners are heavier (Table 11.6).

Table 11.6. Cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-equiv.) and eutrophying (PO_4^{-} equiv.) emissions per weaned pig produced for finishing in high- and low-profitability commodity and niche pig production systems in the Upper Midwestern United states.

	Cumulative Energy Use (MJ)	**GHG Emissions (kg CO_2-e)**	**Eut. Emissions (g PO_4-e)**	**Ecological Footprint (m^2)**
Commodity (high)	**251**	**56.4**	**329**	**288**
Feed Production	37.0%	21.9%	12.3%	52.1%
In-barn Energy Use	53.4%	14.3%	0.8%	8.8%
Manure Management	1.3%	54.4%	76.5%	28.5%
Replacement Sows	8.3%	9.4%	10.4%	10.6%
Commodity (low)	**340**	**79.2**	**499**	**406**
Feed Production	33.7%	19.1%	10.0%	45.2%
In-barn Energy Use	47.1%	12.2%	0.6%	7.4%
Manure Management	1.1%	49.6%	67.9%	25.3%
Replacement Sows	18.1%	19.9%	21.5%	22.1%
Niche (high)	**392**	**74.0**	**879**	**543**
Feed Production	50.5%	35.4%	9.8%	58.8%

	Cumulative Energy Use (MJ)	GHG Emissions (kg CO_2-e)	Eut. Emissions (g PO_4-e)	Ecological Footprint (m^2)
In-barn Energy Use	20.4%	6.7%	0.4%	3.0%
Manure Management	2.1%	30.8%	62.6%	11.2%
Replacement Sows	22.1%	25.9%	27.9%	26.5%
Bedding	4.9%	1.2%	0.1%	0.5%
Niche (low)	**565**	**117**	**1590**	**861**
Feed Production	58.8%	37.8%	9.1%	62.2%
In-barn Energy Use	15.1%	4.6%	0.2%	2.0%
Manure Management	1.5%	33.5%	67.0%	12.2%
Replacement Sows	20.9%	23.3%	23.6%	23.3%
Bedding	3.7%	0.8%	0.1%	0.3%

The life cycle cumulative energy use and ecological footprint associated with finishing pigs are more strongly tied to feed production in the commodity system, whereas manure management is the largest contributor to greenhouse gas (75% of this from manure methane emissions during pit storage, and the balance from manure nitrous oxide emissions at application) and eutrophying emissions. For the niche system, feed production is the most important consideration for all categories other than eutrophying emissions, where manure management is the primary factor. Producing weaned pigs contributes strongly to all impact categories in both the commodity and niche systems, accounting for approximately 20-40% of impacts across systems and categories. Again, impacts are consistently higher for low profitability compared to high profitability production, and the best-performing niche systems have higher energy use, eutrophying emissions and ecological footprint compared to the low-profitability commodity systems, but lower GHG emissions (Table 11.7). Within commodity systems, high-profitability production averages 85% of the impacts attributable to low-profitability production on a whole-animal basis. For niche production, high profitability production averages 77% of the impacts of low profitability production.

Table 11.7. Cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-equiv.) and eutrophying (PO_4^-equiv.) emissions per finished pig in high- and low-profitability commodity and niche production systems in the Upper Midwestern United states.

	Cumulative Energy Use (MJ)	**GHG Emissions (kg CO_2-e)**	**Eut. Emissions (g PO_4-e)**	**Ecological Footprint (m^2)**
Commodity (high)	**1160**	**295**	**1900**	**1700**
Feed Production	52.7%	27.8%	14.2%	57.7%
In-barn Energy Use	22.5%	5.4%	0.4%	2.7%
Manure Management	1.6%	46.3%	66.9%	21.4%
Piglet Production[1]	23.2%	20.5%	18.5%	18.2%
Commodity (low)	**1340**	**344**	**2340**	**1950**
Feed Production	48.9%	25.5%	12.4%	53.7%
In-barn Energy Use	21.6%	5.1%	0.3%	2.6%
Manure Management	1.5%	44.0%	63.8%	20.8%
Piglet Production[1]	28.0%	25.4%	23.5%	22.9%
Niche (high)	**1480**	**327**	**4180**	**2460**
Feed Production	66.0%	40.0%	10.3%	63.4%
In-barn Energy Use	1.8%	0.5%	0.1%	0.2%
Manure Management	1.4%	34.3%	66.7%	12.3%
Bedding	2.1%	0.7%	0.1%	0.2%
Piglet Production[1]	28.7%	24.5%	22.8%	23.9%
Niche (low)	**1830**	**421**	**5810**	**3110**
Feed Production	59.5%	34.6%	8.3%	55.8%
In-barn Energy Use	1.7%	0.5%	0.1%	0.2%
Manure Management	1.0%	31.8%	59.9%	11.7%
Bedding	1.9%	0.4%	<0.1%	0.2%
Piglet Production[1]	35.9%	32.7%	31.7%	32.1%

1) Includes mortality.

Per live-weight kg of finished pig produced, impacts are consistently lowest for HP commodity production. Since niche pigs are marketed at higher weights than commodity pigs, a slightly different pattern of impacts is observed when assessed on a live-weight basis compared to a whole-animal basis. Here, both greenhouse gas emissions and energy use in HP niche production fall between the values estimated for high and low profitability commodity production. Eutrophying emissions and ecological footprint remain higher, however (Table 11.8). On average, there are larger differences in impact levels between high and low profitability niche systems (25%) compared to high and low profitability commodity systems (20%). The most profitable commodity systems produce live-weight pork with 77% of the average impacts associated with equivalent production in the most profitable niche systems. Differences are largest for eutrophying emissions (50%) and ecological footprint (25%), but quite small for energy use (15%) and GHG emissions (2%).

Table 11.8. Cradle-to-farm gate cumulative energy use, ecological footprint, and greenhouse gas and eutrophying emissions associated with live-weight pork production in high- and low-profitability commodity and niche pig production systems in the Upper Midwestern United States.

	Cum. Energy Use (MJ/kg)	**Ecol. Footprint (m^2/kg)**	**GHG Emissions (kg CO_2-e/kg)**	**Eutroph. Emissions (g PO_4-e/kg)**
Commodity (HP)	9.7	14.2	2.47	15.9
Commodity (LP)	11.9	17.3	3.05	20.8
Niche (HP)	11.4	18.9	2.52	32.2
Niche (LP)	14.4	24.6	3.33	45.9

Energy returns on investment for industrial and biotic (as measured in gross chemical energy) energy consumed in swine production were consistently greater for high-profitability compared to low-profitability systems. Commodity systems similarly evinced higher returns, except where high profitability niche production generated slightly higher energy returns on industrial energy investment compared to low-profitability commodity production. Congruent with observed patterns in cumulative energy use, human edible energy returns on industrial energy investment were greatest for HP commodity production (26.7%) and least for LP niche production (18.0%). Due to the fact that swine diets consist largely of human-edible products (corn and soymeal), human edible energy returns on edible energy investment ratios were quite low, ranging from 7.4% to 3.7%. Gross chemical energy return relative to gross chemical energy consumed by the swine varied between 13.1% and 6.7% (Table 11.9).

Table 11.9. Energy return on investment (EROI) ratios for high and low-profitability commodity and niche pork production in the Upper Midwestern United States as (a) human edible caloric energy return on industrial energy investment (b) human edible caloric energy return on human edible caloric energy investment and (c) gross chemical energy return on gross chemical energy investment.

EROI	Commodity (high)	Commodity (low)	Niche (high)	Niche (low)
Industrial Energy[1]	26.7%	21.8%	22.7%	18.0%
Human Edible Energy[1]	7.4%	6.3%	4.8%	3.7%
Gross Chemical Energy[2]	13.1%	11.3%	8.5%	6.7%

1) Assumes 56% yield of boneless meat per live-weight kg produced and an energy density of 4.63 MJ/kg of raw, boneless pork.

2) Assumes a whole-animal energy density of 4.63 MJ/kg.

11.6 Discussion

Our analysis of the comparative life cycle impacts of high- and low-profitability commodity and niche swine production systems in the Upper Midwestern United States points to a range of important considerations and nuances in describing and seeking to further environmental sustainability objectives in this industry. Of first order interest is that, according to the suite of biophysical environmental performance measures considered, the most profitable contemporary commodity production systems consistently outperform niche production systems in this region. However, the degree of difference in performance within and between production strategies varies widely. There is substantial overlap between commodity and niche systems in terms of GHG emissions and energy use, but no overlap for ecological footprint or eutrophying emission per live-weight kg produced. In some instances, such as greenhouse gas emissions, differences are so small as to be attributable to analytical uncertainties. In others, such as eutrophying emissions, the differences are substantial and clearly linked to characteristic management strategies.

It is also essential to note that, whereas the commodity systems evaluated operate at levels of efficiency in part attributable to historical subsidies in the form of extensive research and education which has served to optimize commodity pork production, the niche systems represent a relatively recent move to alternative production strategies and have not been similarly optimized [The IPIC data set collected by Stender and colleagues (2009) and used in this analysis represents the first large-scale effort to analyze efficiencies in niche pork production]. This is certainly evident in the much larger differences in impacts between high and low-profitability niche systems compared to high and low-profitability commodity systems. We therefore caution against interpreting our results as speaking definitively to the potential comparative efficiencies of these production technologies. Rather, they usefully inform considerations of current comparative performance and point towards important improvement opportunities for both commodity and niche production.

Also important to consider are the different *drivers* of ecological efficiencies (i.e. resource or emissions intensity per unit production) in commodity versus niche swine production. At a systems-level, one such obvious difference is the much higher feed efficiency attained in commodity production, which has a critical influence on all impact categories of concern. The much higher productivity of commodity sow herds, which produce twice the number of weaned pigs with lower feed inputs relative to the niche systems, is also important. HP niche weaners have, on average, 86% higher impacts than HP commodity weaners. For this reason, whereas the contribution of producing weaned pigs is roughly 20-25% of average impacts in commodity production, it contributes 25-30% of average impacts in niche production. Improving swine herd productivity is thus a key leverage point for improving the ecological efficiency of niche pork production as a whole. A possible alternative would be to source weaners from commodity systems for niche finishing in hoop buildings – particularly in summer, where feed diversion for thermoregulation in hoop production is not an issue. Also of critical importance are differences in manure management strategies. Finally, mortality rates are seemingly high in both commodity (6.7-9.2%) and niche (7.8-13.7%) production. Reducing mortality rates might thus be an effective means of improving feed use efficiencies and reducing related impacts in both systems. The implications of these differences for each impact category are explored in detail below.

11.6.1 Energy Use

Feed production is the largest contributor to energy use in both commodity and niche production systems. Certainly, this is consistent with observations from LCA research of swine production elsewhere (Zhu and Ireland 2004; Bassett-Mens and van der Werf 2005; Ericksson *et al.* 2005; Stern *et al.* 2005; Williams *et al.* 2006; Dalgaard *et al.* 2007), and also from comparable work of animal husbandry generally (Pelletier 2008, Pelletier *et al. in press*). Notable, too, is that feed is proportionally more important in niche production compared to commodity production. This reflects the much lower feed efficiencies currently achieved in the niche systems (45% higher feed use per kg live-weight production for HP niche compared to HP commodity and 90% higher for low profitability niche).

We did observe that in-barn energy use is slightly higher for commodity weaner production (18%) compared to niche, despite that twice as many weaners are produced and that the commodity weaners are lighter. Overall energy use, however, was higher for niche systems due to higher feed use and the energy costs of replacement sows.

Of particular interest here is that, although overall life cycle energy use is higher for niche finishing, in-barn energy use is a fraction of that observed in commodity finishing. Specifically, in-barn cumulative energy use for commodity finishing (largely for climate control) is more than eight times that consumed in niche production - an observation supported by the work of Lammers *et al.* (2009c), who reported lower energy use in hoop housing for swine finishing in Iowa. This underscores that, by investing in climate control, commodity producers effectively substitute industrial energy for feed energy. In contrast, since niche pigs must devote a fraction of feed energy to maintaining body temperature (particularly in winter), greater feed throughput is necessary, with cascading effects for all impact categories considered. This observation is consistent with that of Honeyman and Harmon (2002), who observed lower feed efficiencies for winter compared to summer hoop-raised pigs. However, there are certainly other forces at work in determining feed efficiency, including genetics, feeder style/adjustment/location/condition, particle size, losses, etc. For this reason, further research would be necessary to ascertain the extent to which the observed differences are attributable to climate control strategies.

11.6.2 Greenhouse Gas Emissions

The factors determining supply chain greenhouse gas emissions are distinctly different between commodity and niche production. In commodity systems, manure management is the primary factor, in particular the substantial methane emissions associated with the liquid manure system. This points to the importance of considering technological mitigation strategies such as anaerobic digesters in commodity production, or the trade-offs associated with solid manure management strategies. Although overall GHG emissions are slightly higher in HP niche compared to HP commodity production, manure-related emissions are higher for commodity production.

In contrast, feed production is the most important contributor to life cycle greenhouse gas emissions in niche production, followed by (and directly influencing) manure management-related emissions (i.e. greater feed throughput means greater manure production and associated manure nitrous oxide emissions – the most important component of manure-related emissions for niche production). Improved feed efficiency, as well as improved swine herd productivity, thus represent the most important leverage points for reducing greenhouse gas emissions in niche production systems. This might be achieved by several means, including the identification of optimal diets for niche production, targeted use of climate control, and better feed management to minimize waste and improve conversion efficiency.

11.6.3 Eutrophying Emissions

The higher eutrophication potential we observed for niche production is partially attributable to higher feed throughput, nutrient excretion, and associated nutrient losses during manure storage and application – again pointing to the desirability of further research to optimize niche diets. However, the primary driver here is differences in manure management strategy. Although liquid manure management in commodity systems is disadvantageous with respect to greenhouse gas emissions, the use of pit storage offers a tremendous advantage over the windrow storage of solid manure in niche systems for eutrophying emissions. Exposure to the elements during storage is a key vector for nutrient loss. This is a non-trivial consideration, particularly given that nutrient run-off from agriculture in the US is a known contributor to significant eutrophication problems in the Gulf of Mexico (Rabalais *et al.* 2002; Diaz and Rosenberg 2008). One potentially efficacious mitigation strategy is the use of covered manure storage facilities for niche swine production.

Although our models assumed similar eutrophication potential per unit nutrient at time of application, it should be noted that application strategy may also be important. Injection of liquid manure from commodity systems would likely reduce eutrophying emissions compared to surface application of solid manure. In contrast, surface application of liquid manure would have greater eutrophication potential than if solid manure were surface applied and incorporated. The influence of such considerations on life cycle eutrophication potential for commodity and niche swine production merit further research.

11.6.4 Ecological Footprint

The ecological footprint metric is unique among impact assessment methods available for use in LCA in that it facilitates a direct measure of the comparative biocapacity required to support the provision of a good or service. It does so by aggregating the area of productive ecosystem (an ultimately limited resource) required to both supply the material/energy inputs and assimilate a fraction (the GHG emissions) of the wastes associated with specific products, services, or levels of consumption (Rees and Wackernagel 1994). Not surprisingly, feed use was the primary contributor to the ecological footprint of both commodity and niche swine production. This insight alone is important, pointing as it does to the fact that many of the key determinants of environmental performance in swine production occur not on the farm itself but rather far upstream along industrial feed provision chains – hence the importance of the life cycle perspective to environmental management in this industry. However, the area of productive ecosystem required to assimilate associated greenhouse gas emissions was also important – making visible the macroscale waste sink requirements associated with swine production which are neither apparent at the level of production nor currently reflected in the cost structure of the swine industry.

11.6.5 Profitability And Ecological Efficiency

Also of great interest is the relationship between profitability and ecological efficiency. We observed that impacts were consistently higher in low-profitability production systems. Certainly, this is intuitive to the extent that profits are determined by input costs such as feed and industrial energy. Feed costs for niche finishing were \$0.47 (HP) and \$0.62 (LP)/live-weight kg produced, whereas for commodity finishing they were \$0.36 (HP) – \$0.46 (LP) (FINBIN 2008; IPIC 2009). All else being equal, systems that maximize productivity returns on resource investments are clearly desirable. However, all else is rarely equal. Both the qualities and quantities of resource throughput may vary, along with associated waste emissions – with the latter considerations currently excluded from price determination. This means that current production systems effectively externalize important environmental costs. The very similar market prices per unit live-weight production within and between HP and LP commodity and niche systems (IPIC 2009, FinBin 2009) bears ample testament. Moreover, tradeoffs are often inevitable when we seek to optimize multiple criteria. Certainly this is true in case of identifying best management practices for manure handling, with liquid manure exacerbating greenhouse gas emissions whilst reducing eutrophying emissions relative to solid manure management. Interestingly, we observed that the most profitable niche operations produced, on average, half of the pork produced in the least profitable operations (IPIC 2009). It would thus appear that both economic and environmental performance is superior in small niche operations, which challenges standard assumptions regarding economies of scale.

In light of the increasing scale of human activities relative to resource and waste sink availability, we believe that such biophysically-based returns on investment considerations need be increasingly considered in policy and management decisions. With attention to trade-offs, such considerations can help inform the identification and preferential promotion of productive strategies which maximize our capacity to meet specific objectives whilst minimizing resource and emissions intensities. However, it must be remembered that efficiencies can be considered from a variety of perspectives, and that both anthropocentric and ecocentric objectives must be weighed.

Clearly, growing concerns regarding both the availability of and environment costs of non-renewable energy resources underscore the important of maximizing industrial energy returns on investment. We found that such returns varied from as low as 18% (LP niche) to as high as 25% (HP commodity), but that the difference between high performing niche and commodity systems were not substantial.

Also meriting consideration are returns on edible food energy investment. Globally, the livestock industry currently consumes roughly one third of cereal crops, with pork production accounting for a large fraction (Steinfeld *et al.* 2006). In light of the projected doubling of meat production by 2050 to meet the demands of a growing population consuming diets higher in animal products, questions regarding optimal use of edible foodstuffs will necessarily gain increasing currency. We found returns on investment of 3.7-7.4%. This is less than comparable returns for grass-finished beef reported by Pelletier *et al.* (*in press*), but higher than for grain-finished beef.

While both of the prior EROI concerns are primarily anthropocentric, considering resource use from an ecological perspective is also important. Krausmann *et al.* (2008) estimate that 58% of human-appropriated, directly used biomass flows are currently directed through global livestock systems. This has obvious implications for energy flows through ecosystems, and their capacity to support biodiversity (Imhoff *et al.* 2004). We found a gross chemical energy return on investment of 6.7-13.1% for the swine production systems considered – much higher than comparable returns for beef production (Pelletier *et al. in press*) but lower than would be anticipated for poultry production, where feed efficiencies are higher (Pelletier 2008).

11.7 Conclusions

It is clear that both substantial differences and overlap occur within and between commodity and niche swine production strategies for the suite of environmental issues considered in this analysis. Opportunities exist for performance improvements in both systems – as evidenced by the wide range of performance within both commodity and niche production systems, as well as differences in management strategies giving rise to the life cycle environmental profiles of these systems. For niche systems, improving swine herd productivity and feed efficiencies, and reducing nutrient losses from manure storage are critical. Greenhouse gas emissions in commodity production may be much reduced through solid manure management or the use of methane digesters. However, mitigation strategies must clearly be sensitive to trade-offs along different dimensions of environmental performance. While profitability in swine production appears to be inversely proportional to feed-related environmental impacts, it is clear that many important negative externalities remain – in particular, greenhouse gas and eutrophying emissions, and the requisite ecosystem support services required to both provide material and energy inputs and assimilate the associated wastes. Here, policy intervention may be essential to preferentially promote environmentally superior management practices.

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11.9 Supporting Information

Table S11.1. Composition of pig feeds for commodity and niche swine herds in Iowa (from Lammers *et al.* 2008, Holden *et al.* 1996).

Ingredient (%)	Phase 1 Finishing	Phase 2 Finishing	Phase 3 Finishing	Phase 4 Finishing	Phase 5 Finishing	Gestation	Lactation
Corn	54	64.5	69.5	77.3	84.4	86.25	68.79
SBM	30.9	30.9	26.5	20	13.5	10	27.5
Soy oil	1	0.65	0.65	0	0	0	0
Dried whey	10	0	0	0	0	0	0
L-Lysine	0.27	0.17	0.15	0.1	0.09	0	0
DL-Methionine	0.09	0.06	0	0	0	0	0
L-Threonine	0.09	0.07	0.03	0	0	0	0
Dicalcium Phosphate	2.12	2.05	1.42	1.1	0.76	2	2.1
Ground limestone	0.73	0.7	1	0.77	0.55	1	0.79
Sodium chloride	0.26	0.4	0.35	0.33	0.3	0.4	0.45
other	0.54	0.5	0.4	0.4	0.4	0.35	0.37

Table S11.2. Emissions factors employed for modeling enteric and manure methane emissions as well as In-barn nitrous oxide, nitric oxide, ammonia, and nitrate emissions associated with excreted and stored manure N for commodity and niche swine production systems in Iowa (from IPCC 2006 unless otherwise noted). Does not include emissions when manure is applied to agricultural land, which are similar for both commodity and niche manure.

	Commodity (manure managed as slurry)	Niche (manure managed as solid)
Nitrous oxide (direct)	.2% of excreted N	1% of excreted N
Nitrous oxide (indirect)	1% of volatilized NH_3-N and NO_x-N during storage	1% of volatilized NH_3-N and NO-x-N, .75% of leached NO_3-N during storage
Nitric oxide[1]	4.8% of excreted N	4% of excreted N
Ammonia[1]	43.2% of excreted N	36% of excreted N
Nitrate	0	30% of excreted N

	Commodity (manure managed as slurry)	Niche (manure managed as solid)
Enteric Methane (kg/head/year)	1.5	1.5
Manure Methane (kg/head/year)	10	1

(1) We assuming volatized ammonia/nitric oxide fraction is 90% ammonia and 10% nitric oxide following Brentrup *et al.* (2000).

Table S11.3. Proximate composition (as-fed basis) of pig feed ingredients used in commodity and niche swine production systems in Iowa (from Sauvant *et al.* 2004).

	Protein (%)	N (%)	P (%)
Corn	7.65	1.224	0.26
SBM	43.12	6.90	0.62
Soy oil	0	0	0
Dried whey	11	1.76	0.8
L-Lysine	0	17.5	0
DL-Methionine	0	8.75	0
L-Threonine	0	10.94	0
Dicalcium Phosphate	0	0	22.4
Ground limestone	0	0	0
Sodium chloride	0	0	0

Table S11.4. Inputs and outputs per hectare of crop cultivated as feet inputs for swine production in Iowa (from Pelletier *et al. in press*).

INPUTS	Corn	Soy
Fertilizer (kg)		
N	145	4.2
P2O5	51	11.0
K2O	65	17.3
Sulphur	4.2	
Lime	321	
Energy		
Diesel (l)	43.0	31.8
Gas (l)	11.2	10.2
LPG (l)	67.3	
Elect. (kWh)	41.5	

INPUTS	Corn	Soy
Herb/Pesticides[1] (kg)	2.8	1.3
Seed (kg)	216	144
OUTPUTS		
Nitrous Oxide (kg)	4.7	1.1
Ammonia (kg)	21.8	8.6
Nitric Oxide (kg)	3.1	0.1
Carbon Dioxide[2] (kg)	142.7	1.8
Nitrate (kg)	0	0
Phosphate (kg)	0	0
Yield (tonnes)	10.7	3.2

(1) Active ingredients.

(2) From lime and urea fraction of N fertilizer as per IPCC (2006).

Table S11.5. Inputs and outputs per tonne of soy beans processed for soy meal and oil used in swine diets in Iowa (adapted from Schmidt 2007).

INPUTS	AMOUNT
Soy beans (tonnes)	1.0
Energy (MJ)	
Electricity	44.0
Light Fuel Oil	145.3
Natural Gas	282.1
Hexane (kg)	0.4
OUTPUTS	
Soy meal (kg)	772.8
Soy oil (kg)	192.4
Hexane (to air) (kg)	0.2
Nitrate (to water) (kg)	.004

Table S11.6. Feed consumption (kg) per sow per year during gestation and lactation for high and low profitability commodity and niche sow herds in Iowa (data for niche sows from Stender *et al.* (IPIC 2009), data for commodity herds calculated by Dave Stender).

Scenario	Gestation	Lactation
Commodity (High)	729.5	253.6
Commodity (Low)	740.9	212.7

Scenario	Gestation	Lactation
Niche (High)	777.3	287.9
Niche (Low)	1247.7	428.6

Table S11.7. Feed consumption (kg) per finished pig in high and low profitability commodity and niche swine finishing herds in Iowa (accounting for mortality).

Scenario	Phase 1 Finishing	Phase 2 Finishing	Phase 3 Finishing	Phase 4 Finishing	Phase 5 Finishing	Total[1]
Commodity (High)	8.46	8.46	45.12	104.33	115.61	281.98
Commodity (Low)	9.07	9.07	48.36	111.83	123.92	302.25
Niche (High)	12.49	12.49	66.62	154.06	179.72	425.38
Niche (Low)	14.23	14.23	75.91	175.54	194.52	474.43

(1) Total feed consumption from Stender *et al.* (IPIC 2009) for niche pigs and FinBin (2009) for commodity pigs. Feed use per finishing stage calculated as per Lammers *et al.* (2008).

Table S11.8. Calculated protein, nitrogen and phosphorus intake (kg) per finished pig and per gestating or lactating sow per year in high and low profitability commodity and niche swine herds in Iowa.

	Gestation (kg/head/year)			Lactation (kg/head/year)			Finishing (kg/head)		
Scenario	**Protein**	**N**	**P**	**Protein**	**N**	**P**	**Protein**	**N**	**P**
Commodity (High)	79.59	12.73	5.36	41.09	6.57	1.97	40.03	6.46	1.59
Commodity (Low)	80.83	12.93	5.44	36.42	5.83	1.74	42.91	6.93	1.70
Niche (High)	84.80	13.57	5.71	49.29	7.89	2.36	60.21	9.72	2.39
Niche (Low)	136.13	21.78	9.16	73.38	11.74	3.51	67.35	10.88	2.67

CHAPTER 12: COMPARATIVE LIFE CYCLE ENVIRONMENTAL IMPACTS OF THREE BEEF PRODUCTION STRATEGIES IN THE UPPER MIDWESTERN UNITED STATES

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12.1 Publication Information

This work has been accepted for publication in the journal Agricultural Systems. Nathan Pelletier was the principal investigator and lead author. Rich Pirog and Rebecca Rasmussen contributed to the study design, data acquisition, and manuscript preparation.

12.2 Abstract

We used ISO-compliant life cycle assessment (LCA) to compare the cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emissions associated with models of three beef production strategies as currently practiced in the Upper Midwestern United States. Specifically we examined systems where calves were either: weaned directly to feedlots; weaned to out-of-state wheat pastures (backgrounded) then finished in feedlots; or finished wholly on managed pasture and hay. Impacts per live-weight kg of beef produced were highest for pasture-finished beef for all impact categories and lowest for feedlot-finished beef, assuming equilibrium conditions in soil organic carbon fluxes across systems. A sensitivity analysis indicated the possibility of substantial reductions in net greenhouse gas emissions for pasture systems under conditions of positive soil organic carbon sequestration potential. Forage utilization rates were also found to have a modest influence on impact levels in pasture-based beef production. Three measures of resource use efficiency were applied and indicated that beef production, whether feedlot or pasture-based, generates lower edible resource returns on material/energy investment relative to other food production strategies.

12.3 Introduction

Beef is an important animal husbandry product, contributing roughly 30% of meat consumed in industrialized countries. The United States, a leader among beef-producing nations, was responsible for 20% of global beef production in 2007 (FAOStat 2008). Beef production in the US is largely characterized by cow-calf herds maintained on pasture and (winter) hay, and mixed-ration feedlot finishing. Less than one percent of beef cattle are currently finished in pasture systems. Nonetheless, there is considerable variability in management strategies for both pasture and feedlot-finished beef, each with characteristic resource use and emissions patterns.

Life cycle assessment is an ISO-standardized biophysical accounting framework used to (1) compile an inventory of the material and energy inputs and outputs characteristic of each stage of a product life cycle and (2) quantify how these flows contribute to specified resource use and emissions-related environmental impact categories (ISO 2006). This allows the identification of key leverage points for reducing environmental impacts within supply chains, as well as comparisons of the resource dependencies and emission intensities of competing production technologies. Moreover, by bringing a suite of environmental accounting protocols under the umbrella of a single, rigorous framework, LCA facilitates evaluations of the environmental tradeoffs associated with different production strategies along multiple dimensions of environmental performance.

Researchers have previously called attention to the substantial feed, water and land requirements for ruminant production (Pimentel and Pimentel 1996; Goodland 1997; Gerbens-Leenes and Nonhebel 2002). More recently, increased interest in the greenhouse gas intensity of food products has spurred a flurry of discussion in the popular media regarding the climate impacts of beef production and the comparative performance of feedlot and grass-based production systems.

LCA research has been used to examine the greenhouse gas intensity of conventional and organic beef production in Sweden (Cederberg and Darelius 2000), Ireland (Casey and Holden 2006a,b) and the UK; and the relative importance of the cow-calf and finishing phases for the farm-gate environmental impacts of Japanese beef production (Ogino *et al.* 2004, 2007). In the US, Koknaroglu *et al.* (2007) have compared energy use in grass and grain-based beef production and Phetteplace *et al.* (2001) investigated the influence of management strategies on greenhouse gas emissions in conventional beef production. However, full LCAs of US beef production strategies have not been reported to date. We contribute to this body of literature by using ISO-compliant LCA to evaluate four important measures of environmental performance (cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emissions) for three distinct beef production strategies as currently practiced in the Upper Midwestern United States when weaned calves are either: sent directly to Iowa feedlots; sent to out-of-state small-grain (wheat) pastures (backgrounded) then finished in Iowa feedlots; or finished on pasture and hay in Iowa. We do not attempt to characterize optimal herds, but rather average contemporary production conditions for the strategies and region of interest as communicated to us by producers, beef researchers, and regional extension staff. We do, however, present sensitivity analyses to test the mitigation potential of soil organic carbon sequestration and improved forage utilization rates in pasture-based beef finishing.

12.4 Methods

We used ISO-compliant life cycle assessment to compare the cradle-to-farm gate cumulative energy use, ecological footprint, greenhouse gas emissions and eutrophying emissions associated with one managed pasture and two feedlot-finished beef production strategies in the Upper Midwestern United States. Our beef production models were developed in consultation with producers, beef researchers and extension specialists, and all rations were formulated by staff at the Iowa Beef Center at Iowa State University using the BRANDS performance model (Dahlke 2004). We do not account for resource use and emissions associated with the production and maintenance of capital goods in any of the systems modelled.

12.4.1 Life Cycle Inventory

Cow-Calf System

The herd modeled for the cow-calf component of the beef life cycle comprises 100 cows, 15 heifers and 3 bulls. We assume a 90 percent annual calving rate, of which 15 are retained as replacement heifers. Seventy-five spring-born calves are sent to finishing in November at 216 kg and 15 cows to slaughter at an average weight of 636 kg (contributing to the total beef production for each system). Local bulls enter the system at 545 kg, having been raised on a diet of hay and grain equivalent to ¾ of the feedlot finishing diet. They are slaughtered after three years at a weight of 727 kg. We assume a similar cow-calf system provides calves directly from weaning to feedlots, out-of-state backgrounding on wheat pastures (see below) or pasture finishing. The cow-calf herd is maintained on legume frost-seeded (i.e. no tillage) pasture forage and hay, with small amounts of grain fed to the cows and heifers. No housing is provided.

Finishing Systems

Calves sent directly to Iowa feedlots with hormone implants finish in 303 days at 637 kg. This strategy represents close to 50% of production in the US Upper Midwest.

Backgrounding refers to the feeding and management of steers and heifers from weaning until they enter a feedlot and are placed on a high-concentrate finishing ration. Wheat pastures are wheat fields that are seeded at 50-100% above normal planting rates, producing a high quality forage which can be grazed from early winter to late spring, depending on seeding date, then subsequently harvested for wheat at season’s end (this allows wheat farmers to generate additional revenue per hectare without compromising yields) (KSU 1993). Iowa cattle which are backgrounded prior to feedlot finishing are trucked to Oklahoma and then Kansas after weaning in November, for grazing on wheat pasture, before returning to an Iowa feedlot at 455 kg. These cattle finish in 450 days

(300 days on pasture and 150 days in feedlots) at 637 kg. Hormone implants are employed during the feedlot stage only. This strategy similarly represents close to 50% of production in the US Upper Midwest.

Calves weaned to pasture in Iowa finish at 505 kg in 450 days on a ration of forage and hay. Hormone implants are not typically used in grass-finished beef production in the Upper Midwest, which largely serves niche markets demanding, among other things, hormone-free meat. Following consultations with regional producers and Upper Midwest beef researchers, we assumed a pasture utilization rate of 60% for both grass finishing and in the cow-calf system. This estimate falls midway between the 30-90% range suggested by Gerrish (2002) for pasture utilization rates in the Midwest.

Corn feed, which does not require processing and is typically sourced locally, is assumed to be transported 30 km by truck. All other non-pasture feed inputs are assumed to be transported 100 km. This reflects additional transport to and from processing in the case of soy co-products or lower Iowa production volumes relative to corn in the case of hay and wheat. In the absence of production strategy-specific on-farm direct energy input data, we apply average on-farm energy use for Minnesota beef production as reported by Ryan and Tiffany (1998) in proxy across all three production systems. This represents energy inputs associated with feed mixing and delivery in the feedlot systems, and with the movement of hay, fences, and cattle between paddocks in the pasture finishing system.

Nutrient Management And Gaseous Emissions

For the cow-calf system and grass finishing systems modeled, housing is not utilized hence all manure is assumed to be deposited directly to pasture. Manure production rates for feedlot finishing are estimated using the Excel-based Manure Nutrient and Solids Excretion Estimator model provided by the Iowa Beef Center at Iowa State University (Koelsch and Power 2005). Manure in the feedlot finishing systems is assumed to be scraped from feedlots and applied to agricultural land within a 5 km radius. Nitrogen and phosphorus emission rates are calculated based on feed composition and consumption, assuming that 2.6% of beef cattle body mass is nitrogen and .69% is phosphorus following Koelsh and Lesoing (1999). Nitrogen excretion estimates are used to calculate direct nitrous oxide, ammonia and nitric oxide emissions from manure management and indirect nitrous oxide emissions from nitrate leaching following IPCC (2006) protocols and Tier I emission factors. Methane emissions from manure management and enteric fermentation are calculated following IPCC (2006) Tier I and II protocols respectively. Tier I protocols are applied for manure management given the trivial methane emissions associated with solid manure management, which is common to all systems modeled. Tier II protocols are applied for calculating enteric methane emissions due to the sensitivity of emissions to diet composition and throughput, and the relative importance of methane emissions to overall GHG emissions in ruminant production. Tier II protocols stipulate a 3% +/- 1% methane conversion factor for feedlot diets containing > 90% concentrates and a 6.5% +/- 1% conversion factor for forage diets. We applied a 5.5% conversion factor for feedlot-finished cattle, since their rations contained high levels (but less than 90%) of concentrates. A 6.5% conversion factor was applied for the pasture phase and a 3% conversion factor for the feedlot stage of the backgrounded/feedlot-finished cattle, which have a feedlot finishing diet containing >90% concentrates. A 6.5% conversion factor was applied for the cow-calf phase and for grass-finished cattle, due to their forage and hay-based diet.

Fodder Production

Inventory data for fodder production (see Supporting Information Tables S12.1-S12.2) were derived from the US National Agricultural Statistics Service (NASS), Iowa State extension publications and peer-reviewed literature. Yields are based on 5-year averages for 2003-2007 calculated from NASS (2008) data. Fertilizer mixes correspond to average US consumption as reported by NASS (2008). Application rates of pesticides and fertilizers used in conventional soy and corn are based on 2005 data for Iowa (NASS 2006). Energy inputs are based on Iowa averages for 2001 (NASS 2004). Inputs to wheat production correspond to national averages (NASS 2004, 2006). Fertilizer inputs and yields for hay and pasture production in Iowa follow Iowa State Extension statistical bulletins and production recommendations (ISU Extension 1997; 2002; 2008). For out-of-state wheat pastures, they follow Oklahoma and Kansas State University Extension recommendations (OSU 2008, KSU 1993). Energy inputs for pasture and hay production are derived from EcoInvent (2008) models for farm machinery operation. All fertilizers and pesticides are assumed to be transported 1000 km by truck and all seed inputs 100 km by truck. Processing of crops applies inventory data reported by Pelletier and Tyedmers (2007) and Pelletier (2008).

Field-level emissions of carbon dioxide, nitrous oxide, ammonia, nitric oxide and nitrate related to nitrogen fertilizer application, biological nitrogen fixation and crop residues were calculated following IPCC (2006) Tier 1 protocols. Additional ammonia-nitrogen emissions at a rate of 5 kg/ha for all field crops was assumed following Andersen *et al.* (2001), and a standard atmospheric nitrogen deposition rate of 15 kg/hectare was assumed across production regions. Indirect nitrous oxide emissions from nitrate leaching to water were calculated based on a standard leaching rate of 30% of surplus nitrogen following a nitrogen balance calculation as per IPCC (2006) guidelines. A 2.9% surplus phosphorus leaching rate was assumed and phosphate emissions calculated using a phosphorus balance following Dalgaard *et al.* (2008).

Co-product Allocation

Co-product allocation is required to apportion resource use and emissions among the co-products of multi-output systems. Since the purpose of this analysis was to describe the biophysical environmental dimensions of a food production system, it was deemed appropriate to base allocation decisions on an inherent biophysical characteristic of crop co-products which both reflects the efficiency of the process and is relevant to the underlying causal impetus of the production system. To this end, the gross chemical energy content of co-product streams was chosen as the basis for all allocation decisions because (1) producing caloric energy is the root driver of all food production activities and (2) the chemical energy of food products present in raw materials is apportioned between processed outputs in a quantifiable manner which speaks directly to the efficiency with which the system provides food energy. For a detailed discussion of this rationale, see Ayer *et al.* (2007) and Pelletier and Tyedmers (2007).

12.4.2 Life Cycle Impact Assessment

Impact assessment in LCA involves calculating the contributions made by the material and energy inputs and outputs tabulated in the inventory phase to a specified suite of environmental impact categories. We considered two resource use impact categories (energy use and ecological footprint) and two emissions-related categories (greenhouse gas emissions and eutrophying emissions) which we believe are of global relevance for considering environmental performance in animal husbandry. All impacts were calculated using the SimaPro 7.1 LCA software package from PRé Consultants (PRé, 2008). Energy use (MJ) was quantified following the Cumulative Energy Demand method (Frischknect *et al.* 2003), which takes into account the conversion efficiencies of primary energy carriers. Whereas ecological footprints have historically been calculated using a stand-alone methodology, the recent incorporation of this method as an impact assessment option in the SimaPro software package now facilitates its use alongside more standard impact assessment methods. The ecological footprint method is unique among LCA impact assessment methods in that it provides a direct estimate of the ecological dependence of economic activity by expressing the resource inputs and waste assimilatory services underpinning specific economic goods and services in terms of the area of productive ecosystem required to furnish them (Rees and Wackernagel 1994). This includes direct land occupation for producing resources as well as the forest land required to sequester emissions. Since productive ecosystem is ultimately a limited resource, this metric facilitates management of cumulative demand relative to biocapacity (Rees and Wackernagel 1994). The ecological footprint was calculated following the EcoInvent 2.0 method (EcoInvent 2008). This method was modified to include methane and nitrous oxide emissions. We believe this method to be of value to our analysis because it facilitates quantification of the ecosystem support (as measured in area of productive ecosystem) required to underpin human activities, which are typically ignored in LCA research. We also believe this measure to be more relevant than simple estimates of land use, which are sometimes quantified in LCA research but are not sensitive to the quality of land use. Greenhouse gas emissions (CO_2- equiv.) were quantified using the

IPCC (2007) method, assuming a 100-year time horizon. Eutrophying emissions (PO_4^{-} equiv.) were quantified following the CML 2001 method (Guinee *et al.* 2001). These assessment methods follow the problem-oriented mid-point approach, meaning that results are expressed in terms of total resource use and emissions rather than actual impact levels.

12.4.3 Life Cycle Interpretation

Impacts were calculated on a whole-herd basis and per kg of live-weight production in each system. Cradle-to-farm gate supply chain impacts were assessed to identify impact hotspots and key leverage points for environmental performance improvements. Comparative impacts between production systems were also evaluated. Although our models assumed equilibrium conditions in soil organic carbon (SOC) flux associated with grain and grass-based beef production (pers. comm., Keith Paustian, Colorado State University, and Cindy Cambardella, National Soil Tilth Laboratory), we conducted a sensitivity analysis to test the potential impact any such differences might have on overall GHG emissions. Specifically, we applied estimates of 0.12 tonnes C sequestered/ha/year for improved cow-calf pastures and 0.4 tonnes C sequestered/ha/year for previously unmanaged pastures subjected to management-intensive grazing for pasture finishing following Phetteplace *et al.* (2001). We assumed SOC equilibrium conditions for all other feed input production systems. We also tested the sensitivity of model outcomes to differences in assumed forage utilization rates under management-intensive grazing in pasture-based beef finishing by alternately applying utilization rates of 30%, 60% and 90% following the range described by Gerrish (2002). Finally, we assessed the energy return on investment ratios in grain and grass-based beef production systems according to: (a) the amount of human-edible food energy produced relative to the total industrial (human-mediated) energy inputs required; (b) the amount of human-edible food energy produced relative to the amount of human-edible food energy consumed by the cattle; and (c) the amount of gross chemical energy produced relative to the gross energy consumption of cattle in each scenario. Whereas EROI measures typically focus exclusively on returns relative to industrial energy inputs, we believe that these additional

EROI measures speak effectively to equally important biotic resource use efficiency considerations (from an anthropocentric perspective for (b) and an ecocentric perspective for (c)) which are often overlooked in discourse regarding resource allocation and depletion issues.

12.5 Results

12.5.1 Life Cycle Inventory Results

Since we assumed a similar cow-calf herd provides calves to all three finishing systems, modelling a single ration plan was sufficient (Table 12.1). This consisted predominately of pasture and hay, with a small amount of wheat fed to cows and heifers. In contrast, inputs and performance in the finishing scenarios varied widely (Table 12.2) (for detailed inventory data for inputs and emissions associated with the production of feed inputs, see Tables S12.1-S12.2). In particular, the annual daily gain in the feedlot-only finishing scenario as estimated by the BRANDS model (Dahlke 2004) was more than twice that of the pasture finishing scenario, with the backgrounding/feedlot finishing scenario falling between these extremes. Feedlot finishing utilized a range of grain and crop co-product inputs.

Table 12.1. Annual ration plan (tonnes, on an as-fed basis) and outputs for the cow-calf phase, which produces 75 calves for finishing.

Feed Inputs[1]	**Cows (100)**	**Bulls (3)**	**Heifers (15)**
40% Legume Pasture[2]	1178.4	36.3	107.2
Mixed Grass Hay	296.1	11.2	18.1
Wheat	9.3	-	1.5
Outputs			
Calves	75		
Live-weight Beef	5445 kg	727 kg	

(1) Minerals and/or supplements not included in analysis

(2) Direct consumption, but 60% utilization rate assumed hence cultivated area scaled accordingly in actual models.

Table 12.2. Ration plan (on an as-fed basis) and performance for finishing 75 weaned calves in feedlots on mixed-ration diets, on a combination of out-of-state wheat pasture and mixed-ration feedlot diets, or on pasture and hay.

Feed Input[1] (tonnes)	**Feedlot**	**Backgrounding/Feedlot**	**Pasture**
Brome Pasture[2]			112.5
40% Legume Pasture[2]	-		1031.2
Wheat Pasture		726.7	
Mixed Grass Hay			54.3
Alfalfa Hay - mid			36.2
Alfalfa Hay - mature	25.1	21.2	
Corn Silage	60.4		
Corn Grain	150.4	109.6	
Corn Gluten Feed	52.7		
Corn Wet Distillers		90.2	
Soy Meal	5.5		
Weight In (kg)	216	216	216
Weight Out (kg)	636	636	505
Days Fed	303	450	450
Average Daily Gain (kg)	1.4	0.9	0.6

(1) Minerals and/or supplements not included in analysis

(2) Direct consumption, but 60% utilization rate assumed hence cultivated area scaled accordingly in actual models.

Similarly, whereas only one set of protein, nitrogen and phosphorus excretion estimates were necessary for the cow/calf phase, these estimates varied considerably between the finishing scenarios (Tables 12.3-12.4) (for details regarding the composition of feed inputs used for these calculations, see Table S12.3). On a whole-herd basis, protein and nitrogen intake and excretion were lowest for the feedlot finishing scenario, and highest for the backgrounding/feedlot scenario. In contrast, calculated phosphorus excretion was lowest for grass finishing, and highest for backgrounding/feedlot finishing (Tables 12.3-12.4).

Table 12.3. Estimated annual protein intake and nitrogen/phosphorus and manure excretion rates for the cow-calf system based on feed intake projections.

	Protein Intake (kg)	Nitrogen Excretion (kg)	Phosphorus Excretion (kg)
Cows (100)	61,688	9,870	922
Heifers (15)	4,780	641	37
Bulls (3)	2,075	327	30

Table 12.4. Estimated protein intake and nitrogen/phosphorus and manure excretion rates for 75 head herds of feedlot-finished, backgrounded/feedlot (FL)-finished, and pasture-finished beef cattle.

	Protein Intake (kg)	Nitrogen Excretion (kg)	Phosphorus Excretion (kg)	Solid Manure for Spreading (kg)
Feedlot	34,149	4,657	634	95,000
Backgrounding/FL	48,650	6,983	847	47,000
Pasture	39,503	5,772	380	NA

12.5.2 Life Cycle Impact Assessment Results

Maintaining cows is, by an order of magnitude, the most resource and emissions intensive aspect of the cow-calf phase across impact categories (Table 12.5). Within impact categories, feed production is the dominant contributor to cumulative energy use as well as the ecological footprint of beef production in both the cow/calf and finishing stages (Tables 12.5-12.6). The size of the ecological footprint is primarily determined by land occupation for crop and pasture production, although the area of global average productive ecosystem required to sequester an amount of carbon dioxide equivalent to the methane and nitrous oxide emissions produced by the cattle and their manure is also substantial (Tables S12.4-S12.5). Nutrient losses from manure, whether directly deposited on pasture by grazing animals or scraped and applied to agricultural fields in the case of feedlot finishing, make the largest contribution to eutrophying emissions for all three scenarios, followed closely by feed production. For greenhouse gas emissions, enteric methane is the leading factor, although both feed production and manure management (primarily nitrous oxide emissions) also make substantial contributions. On a whole-herd basis, impacts are consistently lowest for the feedlot scenario across impact categories. The backgrounding/feedlot herd has the highest cumulative energy demand, greenhouse gas and eutrophying emissions, but a slightly smaller ecological footprint than the grass-finished herd (Tables 12.5-12.6). Direct, farm-level inputs account for only 2% of cumulative energy use for grass-finishing and backgrounding/feedlot finishing, and 4% for feedlot finishing.

Table 12.5. Annual cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-e.) and eutrophying (PO_4^{-}e.) emissions associated with a cow-calf herd providing 75 calves for beef production in the Upper Midwestern United states.

	Cumulative Energy Use (GJ)	**GHG Emissions (tonnes CO_2-e.)**	**Eutroph. Emissions (tonnes PO_4^{-}e.)**	**Ecological Footprint (ha)**
Bulls (3)	**71.2**	**27.8**	**.179**	**14.0**
Feed Production	59.8%	21.5%	27.0%	55.0%
Enteric Methane	-	28.9%	-	15.3%
Manure[1]	-	13.6%	41.3%	7.1%
Other[2]	40.2%	36.0%	31.7%	22.6%
Heifers (15)	**106**	**39.7**	**.269**	**25.9**
Feed Production	90.7%	34.6%	46.2%	73.2%
Enteric Methane	-	44.4%	-	18.2%
Manure	-	19.3%	53.3%	7.9%
Other	9.3%	1.7%	0.5%	0.7%
Cows (100)	**1320**	**531**	**3.74**	**330**
Feed Production	95%	33.4%	40%	71.3%
Enteric Methane	-	44.1%	-	19.0%
Manure	-	21.6%	59.7%	9.3%
Other	5%	0.9%	0.3%	0.4%
Total	**1500**	**599**	**4.18**	**370**
Feed Production	93%	32.9%	40%	70.8%
Enteric Methane	-	43.4%	-	18.8%
Manure	-	21.1%	59.5%	9.1%
Other	7%	2.6%	0.5%	1.3%

1) Predominately nitrous oxide, but also includes manure methane.

2) Predominately legacy cost of producing bull.

Table 12.6. Cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-e.) and eutrophying (PO_4^{-}e.) emissions for 75 head herds of beef cattle finished in feedlots, a combination of backgrounding followed by feedlot finishing, or grass-based pasture finishing systems in the Upper Midwestern United States.

	Cumulative Energy Use (GJ)	**GHG Emissions (tonnes CO_2-e.)**	**Eutrophying Emissions (tonnes PO_4^{-}e.)**	**Ecological Footprint (ha)**
Feedlot	**714**	**262**	**1.85**	**119**
Feed Production	85.6%	26.7%	12.6%	56.8%
Enteric Methane	-	40.2%	-	23.7%
Manure[1]	0.6%	30.4%	86.9%	17.9%
Other[2]	13.8%	2.7%	0.5%	1.6%
Backgrounding/Feedlot	**1110**	**340**	**2.74**	**198**
BACKGROUNDING	577	188	1.42	129
Feed Production	45%	20.4%	20.9%	49.4%
Enteric Methane	-	20.6%	-	9.5%
Manure	-	12.7%	30.8%	5.8%
Other[3]	6.9%	1.6%	0.1%	0.5%
FEEDLOT	533	152	1.32	69
Feed Production	43.4%	16%	6.5%	21.5%
Enteric Methane	-	11.4%	-	5.3%
Manure	< 0.2%	16.4%	41.5%	7.5%
Other[3]	4.5%	0.9%	0.2%	0.5%
Pasture	**830**	**325**	**2.67**	**208**
Feed Production	93.7%	36.6%	51.3%	73.5%
Enteric Methane	-	41.5%	-	17.3%
Manure	-	20.9%	48.4%	8.7%
Other	6.3%	1%	0.3%	0.5%

1) Includes nitrous oxide and methane emissions, as well as energy-related inputs/emissions associated with manure handling.

2) Includes on-farm energy use as estimated by Ryan and Tiffany (1998) for Minnesota beef production.

3) Includes transport of calves to out-of-state pastures in Oklahoma and Kansas, then back to Iowa.

Cumulative energy use is greater during the finishing phase than the cow-calf phase for the feedlot finishing scenario. For all other impact categories and scenarios, however, the cow-calf phase is the greater contributor to resource use and emissions in beef production. Averaged across impact categories, the cow-calf phase is responsible for approximately 63% of impacts per live-weight kg of beef produced in all three of the finishing scenarios (Table 12.7).

Since the BRANDS model (Dahlke 2004) predicts that feedlot-finished and backgrounding/feedlot-finished animals in the systems we modelled are 132 kg heavier than the grass-finished cattle, impacts per live-weight kg produced follow a different pattern than those observed on a whole-herd basis. Here, impacts are consistently highest across impact categories for grass-finished beef and lowest for feedlot-finished beef. The backgrounding/feedlot-finished beef falls roughly mid-way between these extremes (Table 12.7).

Table 12.7. Cradle-to-farm gate life cycle cumulative energy use (MJ), ecological footprint (area of productive ecosystem), and greenhouse gas (CO_2-e.) and eutrophying (PO_4^{-}e.) emissions per kg of live-weight beef produced in feedlot, backgrounding/feedlot, and pasture-finishing beef production systems in the Upper Midwestern United States

	Cumulative Energy Use (MJ)	**GHG Emissions (kg CO_2-e.)**	**Eutrophying Emissions (g PO_4^{-}e.)**	**Ecological Footprint (m^2)**
Feedlot	**38.2**	**14.8**	**104**	**84.3**
Cow-calf phase	67.8%	69.6%	69.3%	75.6%
Finishing	32.2%	30.4%	30.7%	24.4%
Backgrounding/Feedlot	**45.0**	**16.2**	**119**	**97.8**
Cow-calf phase	57.5%	63.8%	60.4%	65.2%
Finishing	42.5%	36.2%	39.6%	34.8%
Pasture	**48.4**	**19.2**	**142**	**120**
Cow-calf phase	64.4%	64.8%	61.1%	64.0%
Finishing	35.6%	35.2%	38.9%	36.0%

If the rates of 0.12 tonnes C sequestered/hectare/year for improved pastures (cow-calf system) and 0.4 tonnes C sequestered/hectare/year for pastures converted to management-intensive grazing (grass-finishing) employed by Phetteplace *et al.* (2001) are realistic for the Upper Midwestern systems modelled, estimated greenhouse gas emissions per live-weight kg produced would be 1.8 kg less for feedlot-finished or backgrounding/feedlot-finished beef, and 8.2 kg less for beef finished on intensively-grazed improved pastures and hay during the transition phase. Here, rather than the 30% difference in emissions calculated based on assumed equilibrium conditions, grass-finished beef would be15% less greenhouse gas intensive than feedlot-finished beef (Table 12.8).

We also tested the sensitivity of our models to variation in forage utilization rates in pasture finishing. At a utilization rate of 30%, average impacts were 22% higher than at our assumed utilization rate of 60%. A 90% utilization rate would reduce average impacts by 7% (Table 12.9)

Table 12.8. Anticipated GHG emissions per live-weight kg of beef produced in grain and grass-finishing systems in the Upper Midwestern United States assuming either equilibrium conditions for soil organic carbon in pasture systems or 0.12 kg C sequestration/ha/year for improved pastures (cow-calf-phase) and 0.4 tonnes C sequestration/ha/year for intensive grazing following Phetteplace *et al.* (2001).

SOC Sequestration Rate (tonnes/ha/year)	**Feedlot GHG Emissions (kg CO_2-e./kg)**	**Backgrounding/Feedlot GHG Emissions (kg CO_2-e./kg)**	**Pasture GHG Emissions (kg CO_2-e./kg)**
0 (equilibrium assumed)	14.8	16.2	19.2
.12 for cow/calf phase, .4 for pasture finishing phase	13.0	14.4	11.0

Table 12.9. Life cycle impacts per kg of live-weight beef produced under management-intensive pasture finishing regimes in the Upper Midwestern United States as a function of forage utilization rate (bracketed values represent difference relative to assumed 60% utilization rate).

Forage Utilization Rate	Cumulative Energy Use (MJ)	GHG Emissions (kg CO_2-equiv)	Eutrophying Emissions (g PO_4^{-}equiv.)	Ecological Footprint (m^2)
30%	63.8 (+32 %)	21.5 (+12 %)	169 (+ 19%)	150 (+ 25%)
60%	48.4	19.2	142	120
90%	43.3 (- 10%)	18.4 (- 4%)	133 (- 6%)	110 (- 8%)

Human-edible energy return on industrial energy investment in each system closely followed observed patterns in cumulative energy use between systems, with highest returns (5.2%) for feedlot finishing and lowest returns (4.1%) for pasture finishing. In contrast, because of the fractions of corn and soy (which could be directly consumed by humans) used in feedlot and backgrounding/feedlot finishing rations, returns on human-edible energy investment were an order of magnitude higher for pasture-finishing (where the only human-edible material consumed is the small amount of grain used in the cow/calf phase) compared to feedlot-finishing, although still less than 100%. However, due to the higher feed throughput volumes per unit production in pasture-finishing, gross chemical energy returns on investment were highest for feedlot-finished beef (2%) and lowest for pasture-finished beef (1.6%) (Table 12.10).

Table 12.10. Energy return on investment (EROI) ratios for grain and grass-based beef production in the Upper Midwestern United States as (a) human edible caloric energy return on industrial energy investment (b) human edible caloric energy return on human edible caloric energy investment and (c) gross chemical energy return on gross chemical energy investment.

EROI	Feedlot	Backgrounding/Feedlot	Pasture
Industrial Energy[1]	5.2%	4.4%	4.1%
Human Edible Energy[1]	4.2%	5.9%	69.1%
Gross Chemical Energy[2]	2.0%	1.8%	1.6%

1) Assumes 43% yield of boneless meat per live-weight kg produced and an energy density of 4.63 MJ/kg of raw, boneless beef.

2) Assumes a whole-animal energy density of 4.63 MJ/kg.

12.6 Discussion

12.6.1 Cumulative Energy Use

Consistent with numerous analyses of animal husbandry systems, we found that feed production was the largest contributor to life cycle energy use in beef production (for example, see Basset-Mens and van der Werf (2005) and Thomassen and de Boer (2008)). However, our results contradict previous suggestions that pasture-finished beef production is necessarily less energy intensive than feedlot-finished beef production (Pimentel and Pimentel 1996; Koknaroglu *et al.* 2007). Our finding is somewhat surprising given the widely perceived notion that pastures are largely solar-driven systems, whereas grain-based production is clearly underpinned by fossil energy inputs in the form of fuel for farm machinery, pesticide and fertilizer production and application, and crop processing and transportation. Three important distinctions are necessary in explanation. First, in the temperate climates characteristic of the US Upper Midwest, hay comprises a substantial fraction of the winter diets of grass-fed animals. Hay production and transportation have associated energy costs which may be similar or greater than those of substitutable feed inputs (in the southern US, where pasture is available for a greater proportion of the year, it is possible that energy inputs may be much lower). Second, the managed pastures modeled in our scenarios are quite distinct from unmanaged rangeland in terms of both inputs and forage yields, requiring energy inputs in the form of fertilizer production and application, seeding, and periodic renovation (the latter was not included in our models) to maintain productivity. Third, the large feed throughput volumes and significant trampling rate associated with forage diets in the systems we modeled serve to amplify the areas of managed pastures required. It should be noted that beef produced on unmanaged rangeland may, indeed, be considerably less energy intensive than the systems we modeled, although this would also result in trade-offs in terms of animal performance and associated emissions. Interestingly, the transportation of animals to and from out-of-state pastures in the backgrounding/feedlot scenario contributed negligibly to overall energy use.

We also recognize that because we were unable to accurately characterize direct feedlot or pasture-level energy use, our use of average energy consumption for Minnesota beef production systems as a proxy for system-specific estimates is a weakness in our analysis. Preliminary work examining the economic costs of energy use in forage-only versus conventional cow herds in Iowa suggests slightly higher energy costs for forage-only systems (*pers. comm.*, Denise Schwab, Iowa County Extension). However, given the trivial contribution made by these direct energy inputs in our analysis (only 2% of cumulative energy use for grass-finishing and backgrounding/feedlot finishing, and 4% for feedlot finishing) relative to the energy use associated with feed provision (Koknaroglu *et al.* 2007) in all systems modeled, we are confident that this deficiency does not significantly influence our results.

12.6.2 Greenhouse Gas Emissions

Our results suggest that pasture-finished beef from managed grazing systems as currently practiced in the US Upper Midwest is more greenhouse gas intensive than feedlot-finished beef when viewed on an equal live-weight production basis. This conclusion is consistent with previous research, which has shown that higher quality diets and increased growth rates reduce ruminant methane and manure nitrous oxide emissions, both of which are key contributors to life cycle emissions (Holter and Young 1992; Lovett *et al.* 2005; Benchaar *et al.* 2001). For example, Casey and Holden (2006a) found that high concentrate diets and the associated reduction in finishing time reduced greenhouse gas emissions in Irish beef production. Hyslop (2008) made similar observations from simulations of UK beef production, and Phetteplace *et al.* (2001) suggested that shorter finishing times achieved by moving calves directly to the feedlot reduced emissions in US beef production. These reductions serve to mitigate the greenhouse gas emissions associated with producing, processing, and transporting feedlot ration inputs. However, greenhouse gas-reduction benefits have also been attributed to intensive grazing systems. Phetteplace *et al.* (2001) found that a transition to intensive grazing during the cow-calf phase as opposed to less management-intensive grazing can reduce emissions, and DeRamus *et al.* (2003) reported that best management practices in grazing systems could reduce enteric methane emissions by as much as 22% compared to continuous grazing. Casey and Holden (2006b) suggest that extensifying beef production through organic practices may result in lower GHG emissions in Irish suckler-beef herds.

Certainly, substantial reductions in greenhouse gas emissions may be possible in both feedlot and pasture-based production systems through genetic selection, forage selection and management, methane inhibition and animal management (Wittenberg 2008). For example, several studies point to the potential of improving the greenhouse gas performance of forage diets through the inclusion of specific legumes (see McCaughey *et al.* 1999; Waghorn *et al.* 2002). Although beyond the scope of the current analysis, it would be interesting to assess the comparative greenhouse gas emissions of the feedlot and pasture-finished beef productions systems modelled here where diets are tailored expressly for reduced methane emissions. As 1 kg of methane is equivalent to 55.2 MJ of lost feed energy (gross energy intake basis), mitigation strategies can have the additional benefits of improved feed utilization (Wittenberg 2008), and decreased energy use and emissions for feed production.

The majority of models of greenhouse gas balances in agricultural production assume that established systems achieve equilibrium conditions in soil organic carbon (SOC) flux (for example, see Watson *et al.* 2002; Freibauer *et al.* 2004). In other words, although grass-based systems may certainly store larger soil carbon stocks than cultivated systems, no net differences in annual flux should be anticipated between well established systems. Following expert consultation (pers. comm., Keith Paustian, Colorado State University, and Cindy Cambardella, National Soil Tilth Laboratory), we adopted this assumption in our modelling endeavours. Several authors, however, have suggested that some pasture lands may, in fact, sequester carbon on an on-going basis. For example, Soussana *et al.* (2007) challenge the concept of carbon sink saturation in European grasslands. Less controversial is the idea that changes in management strategies may also change SOC dynamics (Conant *et al.* 2003). Our sensitivity analysis, which employed the soil organic carbon sequestration rates used by Phetteplace *et al.* (2001) for US pastures undergoing improvement or a transition to management-intensive grazing, suggested a substantial net reduction in the total GHG balance in grazing systems, with pastured beef producing 15% less net GHG emissions compared to feedlot-finished beef. This sensitivity analysis did not, however, account for decreasing SOC sequestration rates over time as the systems approach new equilibrium conditions.

12.6.3 Eutrophying Emissions

Again, the higher estimated eutrophication potential for grass-finishing in the systems we modeled was a direct product of the large feed throughput for forage diets, compounded by a significant trampling rate, the associated scaling effect for the area of managed pasture required, and the greater amount of manure produced relative to live-weight production. Even though losses from the feedlot system were accounted for both at time of excretion and when manure was applied to agricultural land, these were ultimately less than total emissions in the grass-finishing system. We recognize, however, that leaching rates are context-sensitive. While our application of standard leaching factors across production systems provides a reasonable first order estimation of eutrophication potential, we recommend further research in this area. Also of note is that phosphorus excretion was lowest in pasture-based production, which may have important implications for comparative eutrophication potentials of alternative beef production strategies in the US Upper Midwest. Haan *et al.* (2006) found that percent surface cover was the most important determinant of P losses, with well-managed pasture lands not increasing surface water P-levels relative to ungrazed grassland. Similarly, Boody *et al.* (2005) reported that replacing 7-14% of cultivated lands with grasslands in two Minnesota watersheds resulted in a 71-75% decrease in phosphorus loading.

12.6.4 Ecological Footprint

Direct land occupation contributes the largest share of the ecological footprint of both feedlot (for grain and co-product production) and grass-finished (for pasture and hay) beef production. Although the footprint method weights pasture and cropland differently (because pasture provides a greater range of ecosystem services than does cultivated cropland) (Frischknect *et al.* 2003), the large areas required ultimately contribute to a larger ecological footprint for grass-finished beef. The footprint tool also points to the large indirect ecosystem requirements of beef production to assimilate an amount of carbon dioxide equivalent to the greenhouse gas emissions (including methane and nitrous oxide) produced.

12.6.5 Whole-System Perspective

While certain parallels may be drawn to other major animal husbandry sectors (for example, the importance of feed production in life cycle energy use), the patterns of resource use and emissions in beef production are in many ways unique. Foremost among these is the tremendous importance of the cow-calf phase. Similar to Phetteplace *et al.* (2001), we found that the cow-calf phase is the dominant contributor to most impact categories regardless of finishing strategy. This is largely attributable to the low fecundity of cattle compared to other species such as pigs and chickens. Since a cow will produce at most one calf per year, a mature cow is maintained (along with bulls and heifers) for every marketed animal (Williams *et al.* 2006). This more than doubles the resource requirements and emissions per live-weight kg of beef produced. Casey and Holden (2006a) and Cederberg and Stadig (2003) both recommend combined dairy and beef systems as a means of reducing the impacts of calf production, since dairy cows produce both milk and calves whereas beef cow/calf herds are maintained for calf production only.

Also important is the relative feed use efficiency of ruminants compared to monogastric species. Ruminants are able to subsist on relatively high volumes of low-quality forage due to their unique digestive system, which relies on symbiotic methanogenic bacteria to break down cellulose. However, this throughput volume not only magnifies the resource and emissions burdens of feed production, but also results in considerable methane generation, manure nitrous oxide, and eutrophying emissions.

Taken together, these factors contribute to the relatively low resource returns on material/energy investment observed in the beef production systems we modelled. For edible energy return on industrial energy investment, these range from 4.1% (grass finishing) to 5.2% (feedlot finishing). The energy intensity of beef relative to other food commodities (Flachowsky 2002; Carlsson-Kanyama *et al.* 2003; Carlsson-Kanyama 2004) thus places the sector in a disadvantaged position.

Also of interest are the edible food energy returns relative to the amount of feedstuffs consumed by cattle which could have otherwise been directly consumed by humans. Despite the large fraction of human-inedible co-products consumed by feedlot cattle, the small volumes of grain consumed by the cow-calf herd and the corn and soy consumed in the feedlot result in a return of only 4.2%, with slightly higher returns for backgrounded cattle. Since the pastured beef cattle we modelled are not consuming any human-edible products during finishing, the returns are an order of magnitude higher than in the feedlot system. From an anthropocentric perspective, this underscores the benefits of producing pastured beef on land not suitable for agricultural crops. Even the pastured system modelled, however, resulted in a net deficit of human-edible food energy due to the low-level grain consumption during the cow-calf phase.

In a related vein, it has been argued elsewhere that ruminant production provides an efficient means of converting otherwise inedible biotic resources (forage and crop processing co-products) into a human-edible food source (Garnett 2009). From an ecological perspective, however, efficiency returns are even less. Since the chemical energy content of biological materials represents a crude but reasonable proxy for the limited net primary productivity underpinning almost every trophic web, gross energy return on investment provides a reasonable first-order approximation of the ecological efficiency with which our food systems supply food energy relative to the demands they place on ecological communities (Pelletier and Tyedmers 2007). Our analysis suggests low returns (2.0% and 1.6% for feedlot and pastured beef respectively) in all systems modelled.

We should also acknowledge some important limitations to our study. First, the very act of defining “representative” systems masks the variability which exists within and between management strategies, with important implications for apparent environmental performance. For example, following consultation with producers and beef researchers, our models of pasture-based beef finishing in the US Upper Midwest assumed a pasture utilization rate of 60%. While this may, indeed, be a reasonable average for the systems modelled, context-specific utilization rates may range from 30-90%, with the highest utilization rates achieved in well-managed temperate pastures where stock are rotated through paddocks daily (Gerrish 2002). At the high end of this range, our sensitivity analysis suggested that modestly improved resource efficiencies and lower net emissions per unit production would be anticipated. At the low end, impacts per unit production are significantly higher. We also have not considered the potential benefits of organic pasture and feed input production strategies. Research in Ireland suggested that organic beef production may lower emissions and improve resource use efficiencies (Casey and Holden 2006b), although earlier work in Germany using an LCA-like analysis found no net benefits per unit production (Flessa *et al.* 2002). We further recognise that pasture-based beef finishing systems elsewhere in the US which have selected for superior genetics and which have longer grazing seasons may have considerably better environmental performance than the Iowa systems we modelled.

Also of critical importance is our use of IPCC Tier 1 default emission factors for modelling field-level emissions related to fertilizer and manure application on pastures and cropland. These are reasonably well-suited to modeling at the macroscale, but mask considerable variability at the microscale. We recommend further research of alternative beef production strategies which applies process-based models such as DAYCENT (Parton *et al.* 2001) and DNDC (Li 2000) at a systems-levels in order to develop more nuanced insights of the variability characteristic of context-specific management regimes, taking into account soil and climatic factors.

12.7 Conclusions

Life cycle assessment is increasingly used to describe the macroscale environmental dimensions of products and services. By making visible the resource flows and emissions characteristic of specific technologies, it thus provides a starting point for evaluating certain aspects of the relative environmental performance of competing products and services.

According to the metrics employed in this analysis, it would appear that feedlot-finished beef products are less resource and emissions-intensive relative to management-intensive pastured beef production in the US Upper Midwest production systems we modelled along four important dimensions of environmental performance. We recognize, however, that in some cases there may be substantial reductions in net greenhouse gas emissions for pasture systems under conditions of positive soil organic carbon sequestration potential. Furthermore, optimally-managed pasture systems would perform better than our modelled "average" system.

We would also stress that none of the systems analyzed can be described as ecologically efficient. Certainly, our measures of resource returns on investment provide strong indications to the contrary. Our work does not, however, provide insights into the social and economic dimensions of these activities. For example, we do not consider costs and benefits related to variables like job creation or quality of life, nor do we address a spectrum of proximate ecological considerations, including biodiversity impacts, or concerns such as animal welfare. Our results should therefore not be taken as stand-alone metrics of the sustainability of feedlot versus pasture-finished beef production in the US Upper Midwest. Rather, they are intended to contribute to our necessarily evolving and increasingly nuanced understanding of beef production and food system sustainability issues generally, and offer insights into how the beef production systems considered here might best pursue improved environmental performance.

12.8 Acknowledgements

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12.10 Supporting Information

Table S12.1. Inputs and outputs per hectare of crop cultivated as feet inputs for beef production in the Upper Midwestern United States.

INPUTS	Corn	Corn Silage	Soy	Wheat
Fertilizer (kg)				
N	145	195	4.2	68.8
P_2O_5	51	84	11.0	52.0
K_2O	65	195	17.3	66.5
Sulphur	4.2	4.2		
Lime	321	321		
Energy				
Diesel (l)	43.0	95.0	31.8	41.1
Gas (l)	11.2	11.2	10.2	9.4
LPG (l)	67.3	67.3		11.9
Elect. (kWh)	41.5	41.5		37.1
Herb/Pesticides (kg)[1]	2.8	2.8	1.3	0.3
Seed (kg)	216	216	144	117
OUTPUTS				
Nitrous Oxide (kg)	4.7	5.1	1.1	1.9
Ammonia (kg)	21.8	27.3	8.6	13.8
Nitric Oxide (kg)	3.1	4.2	0.1	1.5
Carbon Dioxide (kg)[2]	142.7	222.5	1.8	29.3
Nitrate (kg)	0	0	0	6.6
Phosphate (kg)	0	0	0	0.4
Yield (tonnes)	10.7	42.2	3.2	2.8

1. Active ingredients.

2. From lime and urea fraction of N fertilizer as per IPCC (2006).

Table S12.2. Inputs and outputs per hectare of forage crop cultivated as feet inputs for beef production in the Upper Midwestern United States.

INPUTS	**Alfalfa Hay**	**Mixed Grass Hay**	**Brome Pasture**	**40% Legume Pasture**	**Wheat Pasture**
Fertilizer (kg)					
N	0	168	112	0	114
P_2O_5	68	45	34	34	52
K_2O	224	263	34	45	66.5
Sulphur					
Lime					
Energy					
Diesel (l)	38.5	36.7	6	7[1]	41.1
Gas (l)					9.4
LPG (l)					11.9
Elect. (kWh)					37.1
Herb/Pesticides (kg)					0.3
Seed (kg)	0	0		12.5[2]	275
OUTPUTS					
Nitrous Oxide (kg)	1.3	3.9	2.7	0.7	2.9
Ammonia (kg)	6.0	24.3	18.2	5.9	18.5
Nitric Oxide (kg)	0	3.6	2.4	0	7.3[4]
Carbon Dioxide (kg)[3]	0	71.6	47.7	0	48.6
Nitrate (kg)	93.7	17.2	106.5	39.6	67.6
Phosphate (kg)	0.7	1.1	0.7	0.7	0.4
Yield (tonnes)	8.9	8.1	9.5	9.5	7.2[5]

(1) Includes annual fertilizer application, biannual frost seeding and harvesting.

(2) Biannual frost seeding.

(3) Carbon dioxide from urea fraction of N fertilizer as per IPCC (2006) guidelines.

(4) Calculated based on fall/winter stocking density of 0.8 ha/head, spring stocking density of 0.5 ha/head and total consumption of 9.7 tonnes/head. Also yields 2.8 tonnes of wheat/ha, with allocation between wheat and forage as per gross chemical energy content.

Table S12.3. Feed composition (on an as-fed basis) used for performance projections and to calculate manure and nutrient excretion and methane emissions for beef production in the Upper Midwestern United States.

Feed	Moisture %	DM Digest. %	Crude Prot. %	Ash %	Phosphorus %	Gross Energy (MJ/kg)
Brome Pasture	85	53	2.1	3.00	0.04	2.8
40% Legume Pasture	85	65	2.6	2.64	0.03	2.8
Wheat Pasture	80	65	3.6	2.73	0.06	3.3
Alfalfa Hay - mid	16	59	13.4	8.01	0.21	15.5
Alfalfa Hay - mature	16	50	10.1	7.04	0.21	15.5
Mixed Grass Hay	16	44	10.1	8.01	0.18	15.5
Wheat	12	88	12.3	1.76	0.38	15.5
Corn Grain	15	88	10.0	1.70	0.30	16.4
Corn Gluten Feed	10	79	23.0	6.16	0.49	16.4
Corn Silage	60	68	3.4	1.75	0.09	7.3
Corn Wet Distillers	64	63	10.4	1.44	0.28	8.1
Soybean Meal	12	88	44.0	6.16	0.60	17.1

Table S12.4. Ecological footprint (ha of productive ecosystem) of the cow-calf herd, including shares attributable to greenhouse gas sequestration, nuclear energy and land occupation, in the Upper Midwestern United States.

	Greenhouse Gas Sequestration	Nuclear	Land Occupation	Total Ecological Footprint (ha)
Bulls (3)	**7.3**	**0.1**	**6.6**	**14.0**
Feed Production	21.0%	61.8%	92.4%	55.1%
Enteric Methane	29.4%	-	-	15.3%
Manure[1]	6.0%	-	-	7.0%
Other[2]	43.6%	38.2%	7.6%	22.6%
Heifers (15)	**10.5**	**0.1**	**15.3**	**25.9**
Feed Production	33.7%	100%	100%	73.1%
Enteric Methane	44.9%	-	-	18.1%
Manure	19.6%	-	-	8.1%
Other	1.8%	0%	0%	0.7%
Cows (100)	**140**	**1.63**	**188**	**330**

	Greenhouse Gas Sequestration	Nuclear	Land Occupation	Total Ecological Footprint (ha)
Feed Production	31.3%	100%	100%	71.1%
Enteric Methane	44.8%	-	-	19.1%
Manure	21.9%	-	-	9.3%
Other	1.0%	0%	0%	0.5%
Total	**158**	**1.83**	**210**	**370**
Feed Production	30.1%	97.9%	99.7%	70.6%
Enteric Methane	44.0%	-	-	18.9%
Manure	21.0%	-	-	9.1%
Other	4.9%	2.1%	0.3%	1.4%

1) Predominately nitrous oxide, but also includes manure methane.

2) Predominately legacy cost of producing bull.

Table S12.5. Ecological footprint (ha of productive ecosystem) of 75 head feedlot finished, backgrounded/feedlot finished, and grass-finished beef herds in the Upper Midwestern United States, including shares attributable to greenhouse gas sequestration, nuclear energy and land occupation.

	Greenhouse Gas Sequestration	Nuclear	Land Occupation	Total Ecological Footprint (ha)
Feedlot	**68.7**	**0.9**	**49.5**	**119**
Feed Production	25.4%	82.9%	99.8%	56.8%
Enteric Methane	40.9%	-	-	23.6%
Manure[1]	31.1%	-	-	18.0%
Other[2]	2.6%	17.1%	0.2%	1.6%
Backgrounding/Feedlot	**89.1**	**1.1**	**108**	**198**
BACKGROUNDING	49.4	0.5	79.6	130
Feed Production	19.9%	45.5%	73.8%	49.4%
Enteric Methane	21.1%	-	-	9.5%
Manure	12.9%	-	-	5.8%
Other[3]	1.6%	0%	0%	0.7%
FEEDLOT	39.7	0.6	28.1	68.4
Feed Production	15.3%	45.5%	26.0%	21.3%
Enteric Methane	11.6%	-	-	5.2%
Manure	16.7%	-	-	7.5%
Other[3]	0.9%	9.0%	0.2%	0.6%

	Greenhouse Gas Sequestration	Nuclear	Land Occupation	Total Ecological Footprint (ha)
Pasture	**86.0**	**1.1**	**121**	**208**
Feed Production	35.9%	100%	100%	73.5%
Enteric Methane	41.9%	-	-	17.3%
Manure	21.1%	-	-	8.7%
Other	1.1%	0%	0%	0.5%

1) Includes nitrous oxide and methane emissions, as well as energy-related inputs/emissions associated with manure handling.

2) Includes on-farm energy use as estimated by Ryan and Tiffany (1998) for Minnesota beef production.

3) Includes transport of calves to out-of-state pastures in Oklahoma and Kansas, then back to Iowa.

Table S12.6. Ecological footprint (m^2 of productive ecosystem) per live-weight kg of feedlot-finished, backgrounded/feedlot-finished, and grass-finished beef produced in the Upper Midwestern United States, including shares attributable to greenhouse gas sequestration, nuclear energy and land occupation.

	Greenhouse Gas Sequestration	Nuclear	Land Occupation	Total Ecological Footprint (ha)
Feedlot	**39.1**	**0.5**	**44.7**	**84.3**
Cow-calf phase	69.6%	66.9%	81.0%	75.6%
Finishing	30.4%	33.1%	19.0%	24.4%
Backgrounding/Feedlot	**42.6**	**0.5**	**54.7**	**97.8**
Cow-calf phase	64.0%	63.4%	66.1%	65.2%
Finishing	36.0%	36.6%	33.9%	34.8%
Pasture	**50.7**	**0.6**	**68.8**	**120**
Cow-calf phase	64.8%	61.7%	63.4%	64.0%
Finishing	35.2%	38.3%	36.6%	36.0%

CHAPTER 13: DISCUSSION

13.1 Introduction

The ecological economic understanding of sustainability prioritizes the fulfillment of three conditions for economic activity (Day 1992; Daly and Farley 2004). First is that the scale of economic activities must not exceed the resource provisioning and waste assimilatory capacity of host ecosystems. This requires that economic activity, in aggregate, be constrained relative to biocapacity at all relevant scales. Within this context, the second is that the benefits and burdens of economic activity must conform to some shared conception of distributive justice in material relations. Subject to fulfillment of the first two criteria, the third condition is that economic activities must be executed in a maximally efficient manner. Here, the ecological economic understanding of efficiency is broader than that espoused in mainstream neoclassical economics in that it includes consideration of the comparative resource and waste intensities per unit economic good or service provided (Daly and Farley 2004). In combination, these criteria provide a starting point against which the sustainability of economic activities may be assessed.

The overarching purpose of this thesis was to develop and apply an ecological economic research framework which unites these three conditions in an internally-consistent manner to assessing a limited but important subset of environmental sustainability considerations for livestock production – both in terms of specific sectors, and for the global livestock industry as a whole. The intent was to highlight and explore how these considerations might be brought to bear in furthering environmental sustainability objectives in the livestock industry at present, as well as to assess the relationships between current and projected production volumes for this sector over time and sustainability concerns for the human enterprise as a whole. The preceding chapters each presented important elements of this research program. Chapters Two through Eight developed the theoretical and conceptual foundations for the ecological economic research framework and situated the subsequent empirical modeling work relative to existing literature. Chapters Nine through Twelve applied ecological economic accountancy-type methods to elucidate a subset of the biophysical environmental dimensions of livestock production and consumption in terms of sustainable scale and efficiency considerations. The purpose of this final chapter is to bring together the various threads presented into a coherent whole that can provide the basis for consumers, producers and policy makers to formulate strategies vis-à-vis the future of the livestock sector in light of our individual and collective environmental sustainability objectives.

I begin by reviewing the three supporting elements of the ecological economic research framework that has been developed to underpin this effort. First is the ecological communitarian vision of distributive justice. This vision is intended to provide internal consistency to the understanding of and pursuit of sustainability as the organizing principle of ecological economic activity. The second is the rationale for a biophysically-consistent approach to empirical ecological economic modeling using life cycle assessment accountancy techniques. It is argued that such an approach is necessary to producing the information required to support the development of effective environmental governance strategies. The third element is the ecological economic imperative for scale-based environmental policy and management to achieve sustainability objectives, including the identification and exploitation of potential eco-efficiency measures.

I next interpret the application of this framework to understanding and managing livestock production and consumption for sustainability objectives at regional and global scales. Towards this end, I describe the implications of the ecological communitarian conception of distributive justice, and the moral imperative of environmental sustainability towards which it points, for evaluating and responding to measures of scale and efficiency in livestock production.

I then review a subset of the environmental implications of current and projected growth in the global livestock sector from the perspective of sustainability boundary conditions for human activities as a whole. On this basis it is concluded that the scale of the livestock sector now and as projected for 2050 is fundamentally unsustainable, and that all possible opportunities for reining in this sector need be exploited in the interest of preventing irreversible ecological change. While it is recognized that efficiency measures alone will be insufficient, it is nonetheless clear that all such measures must be vigorously pursued. Towards this end, I next review my evaluations of each of the three major livestock sectors in the United States, along with several alternative production technologies, with respect to efficiency concerns. I conclude by summarizing the implications of this work for environmental policy and management for both scale and efficiency objectives, with reference to current and potential environmental governance regimes. In particular, I point to the inadequacies of existing multilateral approaches, and the need for much stronger environmental governance, potentially operationalized by a centralized World Environment Organization. Future research needs and opportunities are described.

13.2 Discussion

13.2.1 Ecological Communitarianism And Environmental Sustainability In Economic Activity

Economic paradigms comprise internally consistent traditions of theory and tools whose purpose is to understand and provide guidance for the management of economic activities in pursuit of specified societal objectives. This thesis began by examining and rejecting the currently dominant neoclassical economic paradigm as a basis for managing human activities for environmental sustainability objectives. One obvious reason for this rejection is the failure of the neoclassical model to provide a compelling empirical account of the relationships between human economies and the finite biophysical environment which provides the supporting infrastructure. A less immediately apparent but also compelling reason is the tenuous normative foundation of neoclassical economics, both in its understanding of the economic actor and its interpretation of distributive justice. Instead, it is argued that sustainability concerns can be better accommodated by an ecological economic paradigm whose internal consistency is based in an ecological communitarian conception of distributive justice.

Consistent with the insights of ecology, this perspective begins with a reconception of the economic actor as community member, and the concomitant recognition that environmental sustainability constitutes the first principle of distributive justice. Simply put, when we accept the reality of ecological interdependence and the fundamental contribution of the ecological communities in which we are embedded to the conditions necessary to our existence and well-being, we are compelled to accept that its maintenance provides the overarching moral imperative which circumscribes the realm of ethically acceptable behaviour for the economic actor – in other words, the backdrop against which further ethical considerations in human economic organization might be entertained.

At its most basic level, this imperative serves to legitimize the primacy of the scale variable in economic organization. Taking into account the Laws of Thermodynamics and the finite nature of our biosphere, constraining the scale of economic activities relative to the resource provisioning and waste assimilatory capacities of host ecosystems so as to ensure their sustainability comprises the most important performance measure for assessing sustainability in economic activity. This requires attention to sustainable scale considerations at local, regional, and global levels. Economic activities, when viewed in either a discrete or aggregate sense, are unacceptable where they conflict with objective measures of sustainable scale. In this light, individual and collective production and consumption choices that result in an unsustainable scale of resource use and emissions are fundamentally unjust from an ecological communitarian perspective, and hence must be restructured accordingly. Such considerations need necessarily become a cornerstone of policy and management considerations if sustainability is to be realized.

Subject to the assurance of sustainable scale, considerations of efficient allocation in economic activity are also morally charged to the extent that alternative allocations result in resource and emissions intensities per unit economic good or service delivered which contribute variably to our capacity to meet competing human needs whilst achieving sustainable scale objectives. This might be considered in the context of alternative production strategies for a given good or service. It could equally be applied to considering the functions served by similar but non-identical products that satisfy the same human need.

It is also critical to note, however, that these considerations represent only the necessary departure point for assessing the moral legitimacy of economic activities. Such a departure point is necessary because, in the absence of environmental sustainability, humanity simply cannot and will not achieve sustainability in any other sphere. Once environmental sustainability is assured, then a wealth of other considerations need subsequently be entertained and will serve to further constrain the realm of the legitimate. These further considerations are obviously also critical, but fall well beyond the scope of the present analysis.

13.2.2 Ecological Economics As Empirical Analysis – Minimum Necessary Conditions

Understanding and managing the environmental dimensions of economic activity for sustainability objectives requires analytical frameworks conducive to reinterpreting economic activities in terms of biophysically relevant currencies – in other words, resource and waste intensities per unit economic good or service provided. Such information provides the basis for managing for either scale or efficiency considerations. Towards this end, a variety of biophysical accounting metrics have been advanced. Life cycle assessment has emerged as a leading technique because, by bringing a suite of related impact assessment methods under the umbrella of a single, standardized accounting framework, it affords opportunities for nuanced, multi-criteria environmental performance assessments. For this reason, LCA was chosen as the basis for the empirical ecological economic modeling work conducted as part of this dissertation research.

However, as argued in Chapter Six, the usefulness of such biophysical accounting frameworks in managing for sustainability objectives is largely a function of the extent to which the models developed actually meaningfully represent the environmental dimensions of the economic activities of interest. Since an important objective of biophysical modeling is to inform optimally efficient resource allocation towards meeting human needs within the constraints posed by the limits of sustainable scale, this requires, at a minimum, that the models be constructed so as to realistically mirror the flows of resources and wastes mobilized in the pursuit of meeting specific human needs. Unfortunately, it would appear that much contemporary biophysical modeling work is undermined by the uncritical application of market information in biophysical models. The examples considered here are the use of economic allocation in attributional LCA, and market-based system delimitation in consequential LCA. Whereas biophysical modeling work is ostensibly conducted to provide a basis for understanding the environmental dimensions of economic activity precisely because current market signals are typically unreflective of such information, these practices effectively reintroduce environmentally myopic market information at critical junctures into what are otherwise biophysical models. As a result, the research outcomes frequently simply mirror existing market signals, rather than providing a biophysically-enlightened basis for decision making.

To overcome this weakness, the case was made for the exclusion of market information from life cycle models. Instead, it was argued that practitioners should apply best-fit biophysical parameters in all cases, and that models should be constructed and interpreted towards the end of understanding the biophysical environmental dimensions of satisfying fundamental human needs – which is the ultimate purpose of economic activity, and the end towards which scarce resources must be most efficiently allocated. This includes the most basic need of the ecological economic actor for ecological integrity, which requires first the assurance of sustainable scale. Towards this end, it was further argued that biophysical accountancy-type ecological economic modeling such as life cycle assessment research can and should be interpreted with respect to both scale and efficiency concerns.

13.2.3 Ecological Economics For Policy And Governance

Policy and governance institutions serve to operationalize societal objectives. Sustainability is fast becoming a key priority for policy and governance across scales of organization. Yet, as argued in Chapter Seven, our current approaches to global environmental governance are inadequate to the task of ensuring environmental sustainability in economic organization. In large part, this failure is attributable to the fact that the very same worldview and suite of fallible assumptions which have given rise to the crises of sustainability we face also permeate the governance institutions we advance in recourse. This includes, among other things: an instrumental conception on non-human nature; an unwarranted faith in the power of human ingenuity to advance technologies to overcome scarcity; and an economic system whose organizing principle is the desirability and necessity of ceaseless growth. As a result, the current understanding and institutionalization of sustainability as “sustainable development” has served to mask the fundamental incompatibility between growth and sustainability. In positioning green industrialism as the rational ideal for global society, it has also much narrowed the scope of sustainability discourse.

All of this is in direct contradiction with the ecological economic understanding of sustainability, which requires at a minimum a scale-based approach to governance that recognizes environmental sustainability as the first principle of distributive justice and, as such, the foundational imperative for the governance of economic relations. For this reason, it is argued that the requirements of global environmental governance transcend the capacity of existing institutions to provide. The current, predominantly UN-administered system of uncoordinated multi-lateral environmental agreements, which guarantee the sovereignty of self-interested nation states negotiating on a single-issue basis whilst ignoring the issue of growth, is clearly failing. While a number of authors have argued for strengthening the existing UN system – and, indeed, there may be much scope for improving current arrangements which can and should be vigorously pursued - it is doubtful that simply tinkering with the status quo will be sufficient. Congruent with the Principle of Subsidiarity, effective global environmental governance requires a governance institution of commensurate scope. Towards this end, it is argued that a strong World Environment Organization (WEO), with the powers to legislate and enforce limits on use of the global commons, is a necessary first step. Although some have called such a proposal “utopian,” it is argued that this proposal is certainly far more pragmatic than the expectation that global industrial society has any chance of longevity if we continue to cling to a system of environmental governance that is hopelessly inadequate to the task.

In invoking the Principle of Subsidiarity, however, it must also be recognized that sound environmental governance institutions are also required to effectively deal with issues which manifest primarily at local and regional scales. A World Environment Organization should therefore be seen as the appropriate institutional forum for the governance of global-scale common pool resources such as the carbon cycle, the nitrogen cycle, and aggregate biomass appropriation. Within the limitations prescribed by a WEO, environmental governance institutions at the level of the nation state or (preferably) more ecologically relevant jurisdictional delineations can and should remain operational or be developed. In concert, this range of institutions must coordinate the efficient allocation of resources towards the ends of meeting human needs whilst ensuring an environmentally sustainable scale for economic activities in aggregate. While acknowledging the necessity of appropriate governance institutions at all relevant scales, and pointing to a range of existing instruments for implementing environmental governance objectives at regional and national scales, the focus of this dissertation is largely on global scale governance considerations.

13.2.4 Ecological Communitarian Distributive Justice And The Livestock Sector

To apply and operationalize this ecological economic understanding of sustainability for the livestock sector is first to seek to understand the environmental dimensions of livestock production and consumption and, second, to formulate policy prescriptions that ultimately serve distributive justice by way of the environmental sustainability objective. In short, this is a *normative* program in that the intent is to furnish the information necessary to achieve a prioritized value outcome, in this case environmental sustainability. By this light, policy prescriptions do not proceed solely from an interpretation of empirical modeling outcomes. Rather, they are motivated and legitimized by the recognition that environmental sustainability is the first principle of distributive justice in economic organization, and the necessary precursor to sustainability in any other sphere of the human endeavour.

What, then, are the normative imperatives for governance of livestock production and consumption in the interest of sustainability? As with any economic activity, the first criterion is sustainable scale. From the ecological communitarian perspective, the legitimacy of specific configurations and levels of livestock production and consumption must first be weighed relative to their demands on the resource provisioning and waste assimilatory capacity of host ecosystems. Those that fall within the dictates of sustainable scale pass the litmus test for further consideration. Those that do not must be redressed prior to consideration of other variables. Of course, treatment of the entirety of sustainable scale criteria which must be brought to bear far exceeds the scope of the present analysis. Rather, I have chosen a limited but important and illustrative subset of environmental criteria by which livestock production might be assessed at global scales.

Within the context prescribed by considerations of sustainable scale, there is similarly normative weight in the allocation of what are inherently limited resources to livestock production. Such allocations must take into account measures of resource and waste intensity, as compared to alternative production technologies, alternative livestock products, and alternative food production strategies, generally. From this perspective, right policy and governance is that which moves the livestock sector, and the food system as a whole, towards production and consumption norms which maximize allocative efficiency by meeting human needs for food energy at least environmental cost and within the buffering capacity of host ecosystems.

13.2.5 The Global Livestock Sector: Sustainable Scale

Animal husbandry is a major driver of anthropogenic environmental change (Steinfeld *et al*. 2006). Increasingly, the global food economy is being influenced by a shift in food consumption patterns towards livestock products (Delgado *et al.* 1999). To meet the demands of a growing population consuming diets higher in such products, world-wide production levels are anticipated to double from year 2000 levels by 2050, putting further stress on planetary resources and biogeochemical cycles (Steinfeld *et al.* 2006; FAO 2006). As demonstrated in Chapter Nine, there are strong reasons to believe that the scale of the global livestock sector – both at present and as projected into the future - is fundamentally unsustainable according to the three dimensions of environmental performance evaluated here: greenhouse gas emissions; biomass appropriation; and reactive nitrogen mobilization.

Here, sustainable scale for human contributions to these concerns was based on the concept of "safe operating space" advanced by Rockstrom and colleagues (2009). Consistent with the ecological economic understanding of sustainable scale as a level of activity that falls safely within the buffering capacity of planetary ecosystems and biogeochemical cycles, sustainable scale for each of these criteria was approximated based on the best available knowledge regarding the current status of human/environmental interactions in each dimension.

For greenhouse gas emissions, the sustainability boundary condition was defined as an atmospheric CO_2 concentration of 350 ppm, which is the level currently deemed necessary to reduce the probability of an increase in mean surface temperature above 2^0 C to 20 percent. Increases above this level are thought to be potentially dangerously destabilizing (Malte *et al.* 2009). As reported in the most recent synthesis of climate science provided by Allison and colleagues (2009) in the Copenhagen Diagnosis, this corresponds to anthropogenic CO_2-e emissions of well below 1 metric tonne per capita by 2050. I hence chose a generous boundary condition, based on UN median population scenario forecasts, of 8.9 Gt (i.e. one tonne per capita per year). For biomass appropriation, Bishop and colleagues (2009) suggest that biodiversity preservation requires that human use not exceed 9.7 Gt annually. For reactive nitrogen mobilization, Rockstrom and colleagues (2009) suggest a sustainability boundary condition of 35 Mt of nitrogen removed from the atmosphere per year.

As of 2000, the global livestock sector occupied 52% of humanity's safe operating space for greenhouse gas emissions, 72% for biomass appropriation, and 217% for reactive nitrogen mobilization. It was further conservatively estimated that, based on FAO (2006) production forecasts, this will increase to 72%, 88% and 294% of humanity's safe operating space for these criteria respectively by 2050. Substituting marginal production of beef above year 2000 levels with poultry (the most efficient of the major terrestrial animal husbandry sectors) would reduce anticipated impacts by 5-13%. At one end of the spectrum of possibilities for meeting global protein consumption requirements at levels recommended by the USDA, fulfilling these requirements via the production of livestock products at projected production ratios would further increase anticipated impacts for 2050 by 12-28%. At the other extreme, meeting recommended protein consumption levels in entirety via the production of soy protein would reduce anticipated impacts by 77-98%.

Given the already substantial and increasing role of the livestock sector in appropriating our safe operating space in multiple dimensions, meeting the objective of sustainable scale for the global livestock sector will require the exploitation of all possible leverage points. If the livestock sector is to maintain its current proportional contribution to anthropogenic greenhouse gas emissions, biomass appropriation and reactive nitrogen mobilization, and our activities as a whole are to be constrained relative to sustainability boundary conditions, this will require a reduction in contributions from the livestock sector to 19%, 42% and 21% of projected levels respectively. For this reason, attention to the spectrum of production, consumption, and technological variables – in other words, *what* is produced, *how much* is produced, and *by which methods* production occurs – is therefore critical. As indicated, substitution of less impactful production systems would result in non-trivial reductions in aggregate impacts but, alone, is far from adequate. More promising is reducing the extent to which livestock products contribute to fulfilling human nutritional needs. What is clear from this analysis, however, is that the scale of the livestock sector relative to sustainability boundary conditions underscores the moral imperative of developing strong governance institutions and policy initiatives to rein in growth in this sector in order to ensure environmental sustainability.

Given that the models employed generously assumed a 35% average reduction in impacts per unit livestock protein produced globally by 2050 relative to year 2000 levels, continued efficiency gains are clearly also critical. Towards this end, I next review efficiency considerations for the three major sectors of the US livestock industry.

13.2.6 Efficiency Considerations In The US Livestock Sector

Taken as a whole, the US livestock sector is the largest and most productive in the world. Of total global production of poultry, pork and beef in 2007, 16% was produced in the continental United States (FAOStat 2008). For this reason, a substantial share of global material and energy resources are directed through the US livestock sector, which in turn results in diverse waste streams that burden the assimilatory capacity of receiving ecosystems and biogeochemical cycles at multiple scales. By many measures, this sector might also be considered the most efficient livestock industry in the world (Capper *et al.* 2009). In recent decades the application of science and technology to optimize resource use per unit of livestock product produced has much improved resource use efficiencies. Nonetheless, environmental problems remain pervasive (Steinfeld *et al.* 2006), to say nothing of socio-economic and animal welfare concerns. Moreover, industrialization of the sector has transformed what were previously locally-integrated, small-scale production systems into high volume commodity systems predicated on extensive networks of interlinked industrial activities. This has served to both increase absolute impacts despite improved resource use efficiencies per unit production, as well as to obscure the relationships between production and consumption choices and environmental changes that occur at spatially and temporally disparate locales. Understanding and managing these systems for environmental objectives hence requires a systemic perspective and analytical tools of commensurate scope.

Here, I applied an ISO-compliant life cycle assessment framework to characterize the flows of material/energy resources and associated wastes in the three largest sectors of the US livestock industry. Specifically, I evaluated the resource use and emissions characteristic of the US broiler poultry sector as a whole, as well as commodity and alternative beef and pork production strategies in the Upper Midwestern United States, in order to quantify cumulative energy use, greenhouse gas emissions, eutrophying emissions and ecological footprint per unit live-weight production in each system.

Efficiency Considerations For The US Broiler Poultry Sector

By many measures, industrial broiler poultry production is currently the most efficient means of large-scale terrestrial animal husbandry. This is primarily a function of the low feed conversion ratio (FCR) typically achieved, where as little as 1.9 kg of feed per kg live-weight production is required. Interestingly, despite this low FCR, feed production nonetheless contributes the bulk of supply chain impacts in broiler production – in this study accounting for, on average, 73% of measured cradle-to-farm gate impacts. Litter management (in particular, storage and application) is also important in terms of eutrophying emissions. Given that little of the measured impacts along the poultry supply chain actually occur in or near the poultry production facility itself, this clearly underscores the fallacy of the concept of "landless farming" for which intensive animal husbandry is sometimes promoted. It further points to the necessity of full supply chain environmental management, in particular of the upstream production, processing and distribution chains from which feed inputs are derived. Here, the concept of least-environmental cost feed sourcing becomes important. Since broiler feeds may potentially include a range of crop, fisheries, and livestock-derived materials, each with its own characteristic resource and emissions intensities, prioritizing this consideration in feed formulation and sourcing should be a priority concern for environmental management in this sector. Certainly, a key leverage point for reducing feed-related impacts is a reduction in the use of fishmeal and poultry by-products, both of which have substantially higher impacts per unit mass than the corn and soy fractions of poultry feed. One option is increased reliance on synthetic amino acids to create a nutritional profile most conducive to broiler growth (Binder 2003). Improved nitrogen use efficiencies at the level of field crop production are also essential, since nitrogen fertilizer production and use figures among the most energy and emissions-intensive aspects of the broiler supply chain, and of industrial food systems as a whole (Pelletier *et al.* 2008).

Efficiency Considerations For US Commodity And Niche Pork Production

Similar to poultry production, feed provision is also a central driver of impacts in intensive swine production. Here, however, other aspects of husbandry were proportionally more important than for broiler production, and distinct differences were noted between high volume, commodity production and low volume, niche production technologies. Whereas the hatchery phase made negligible contributions to supply chain impacts in broiler production, producing weaned piglets contributed roughly 20-30% of measured impacts in pork production. In-barn manure management strategies were also influential. For commodity production, handling manure as a liquid resulted in substantial methane emissions, making this the most important contributor to greenhouse gas emissions. In contrast, handling manure as a solid in niche production much reduced manure-related greenhouse gas emissions, but resulted in higher eutrophying emissions from storing manure in windrows. What becomes evident here is the importance of considering tradeoffs when seeking to mitigate particular environmental impacts, as gains in one arena may come at the expense of losses in another. Potential win-win strategies include the use of methane digesters for manure management in commodity production, and covered manure storage for niche production. The former strategy could simultaneously offset the higher energy demands of commodity production. The latter would reduce eutrophying emissions as well as improve nitrogen use efficiencies in the crop-livestock system as a whole. This is of particular import given that the livestock sector is by far the largest contributor to reactive nitrogen pollution, and that optimizing nitrogen use efficiencies perhaps constitutes the most important priority for global food systems looking forward.

While niche production has been promoted on health, animal welfare, and environmental grounds, for the suite of impact categories considered here this technology is outperformed by commodity production on all counts. However, there is substantial overlap in the range of performance characteristic of each strategy in some arenas, most notably greenhouse gas emissions. Moreover, whereas commodity production has been optimized over many years through significant levels of research and development, niche production is a much more recent phenomenon and has not benefited from the same level of support. This suggests that if a comparable level of research and education were leveraged for improving efficiencies in niche production as has been applied to optimizing commodity production in this region, then niche pork production might, in fact, offer the least-environmental cost production strategy in several important respects.

Also of interest here was the observation that higher profitability farms had consistently superior environmental performance to less profitable operations. At first glance, this might suggest correspondence between market signals and environmental performance. It is important to note, however, that this relationship is indirect and highly imperfect. Although profits are strongly influenced by the volume and costs of feed throughput, and a large fraction of impacts are related to feed provision and use, market signals remain oblivious to all non-priced impacts, in particular non-marketed resources (for example, gaseous emissions and the biocapacity required to sequester wastes). For this reason, prices received by producers are the same within and between production systems, regardless of comparative environmental performance. Were the market mechanism accounting for these externalities, then one would expect variable pricing, and a streamlining towards environmentally optimal production norms. This points to the necessity of increased internalization of environmental costs in the cost structure via both market as well as non-market mechanisms, since many ecosystem goods and services are pure public goods and hence not commodifiable.

Efficiency Considerations For US Feedlot And Grass-finished Beef Production

The environmental impact profile of beef production is quite distinct from that of either poultry or pork production. Here, maintaining the cow-calf herd is the single largest contributor to impacts across all three production technologies considered. Although feed production remains the largest contributor to supply chain energy use and ecological footprint, its contribution to greenhouse gas and eutrophying emissions are much lower. Instead, greenhouse gas emissions are dominated by enteric methane production and manure nitrous oxide, and manure management is the primary contributor to eutrophying emissions. On all counts these are higher for the cow-calf phase than for the finishing phase. In other words, maintaining cow herds more than doubles impacts per kg of live-weight beef sent to market.

Despite widespread preconceptions regarding the environmental preferability of pastured versus feedlot beef production, this assumption was systematically discounted for the suite of impact categories considered. At first glance, this is counter-intuitive given that pasture-based systems appear to be largely solar driven, whereas feedlot production is clearly underpinned by intensive agricultural crop production predicated on fuel inputs for farm machinery, energy intensive fertilizers, and pesticides, along with downstream crop processing and distribution systems. However, the managed pasture systems considered here require periodic renovation, seeding to maintain legume forage densities, and fertilization for optimal animal performance. At the same time, the long winters characteristic of temperate climes mean that pastured animals must be fed on stock-piled forage over winter seasons, during which time a large fraction of feed energy is diverted to thermoregulation and much less to growth. Hence, any such comparisons must take into account the environmental burdens of producing and transporting stockpiled forages and the amount and impacts of feed consumed. Overall, due to the longer lifespan and greater feed throughput volumes (with associated methane, manure, and manure nitrous oxide), measured impacts for pastured beef were, on average, 34% higher per unit live-weight production compared to feedlot-finished beef. It should be noted that cattle produced on unmanaged rangeland may, indeed, have lower associated energy use, but impacts in all other categories considered here would increase.

Comparative Drivers Of Impacts In US Poultry, Pork, And Beef Production

In concert, these analyses highlight the desirability and necessity of biophysical accountancy-type research to understanding and managing the environmental dimensions of animal husbandry systems. First is the necessity of the life cycle perspective to identifying opportunities for improving environmental performance within existing supply chains, which will differ substantially depending on the species and production technology of interest. Second is the capacity it affords to make comparisons between competing production technologies, alternative livestock products, and alternative food products more generally.

As has been noted, improving environmental performance in poultry production for the suite of measures considered here is largely a function of maximizing feed conversion efficiencies and identifying opportunities to reduce the use of high-impact animal-derived feed inputs. For pork production, feed remains important but maximizing efficiencies in producing weaned pigs, and choice of manure management strategy, are also determining factors in environmental performance. In contrast, feed-related impacts are of lesser importance in beef production, where maintaining the cow-calf herd along with impacts related to feed throughput volume and associated manure production are of particular concern. Key to understanding the differences in the intensity and distribution of environmental impacts between livestock products are considerations of physiology.

The varying importance of different aspects of the life cycle to overall impacts in each of these systems can be explained in three ways. First, is the tremendous difference in fecundity characteristic of each species. In poultry production, where a single breeder hen may produce hundreds of offspring annually, the impacts of maintaining breeding stock are insignificant. With pig production, a sow may give birth to ten or more piglets at a time and as many as twenty-five yearly. Each of these will carry a share of the environmental burdens associated with maintaining the sow. In contrast, cattle will give birth to at most one calf per year. This means that for every calf finished for beef, an adult cow is simultaneously maintained as breeding stock. The maintenance of breeding herds, including bulls and heifers, more than doubles the environmental burdens per marketed animal.

Second is the varied feed conversion efficiencies achieved by each species, which range from as low as 1.9 kg of feed per kg of live-weight production for poultry to as high as 10-30 kg of feed per kg of beef produced. Higher feed intake means larger feed-related emissions, as well as greater rates of manure production with associated nitrous oxide emissions.

The third factor relates to fundamental differences between monogastric (poultry and pigs) and ruminant animals like cattle. Ruminants possess a distinctive digestive process that allows for the utilization of low-energy forage, which is largely indigestible for monogastric species. However, enteric fermentation also results in the production of significant methane emissions. Due to these differences, each supply chain will require distinct interventions and management strategies to improve efficiencies and will also have very different impact levels.

Comparative Efficiencies In Poultry, Pork And Beef Production

There is tremendous variability in the impact intensity per unit live-weight production between the three livestock systems considered (Figure 13.1). Of first order interest is that beef production is much more impactful (2.3-5.6 x) across all performance measures considered. In contrast, performance is much closer between poultry and pork production, with the poultry supply chain consuming more energy and producing more eutrophying emissions per unit live-weight production, but resulting in lower greenhouse gas emissions and ecological footprint than pork production.

Energy returns on investment are also highly variable both between measures and livestock production technologies (Figure 13.2). Producing beef generates consistently lower energy returns on investment for all measures. This is particularly noteworthy for human-edible energy returns on human-edible energy investment, since a purported benefit of ruminant production is that the animals in question can be raised on forage and co-product materials not directly consumable by humans, hence reducing competition for scarce food crops (Garnett 2009). Here, although human-edible products contribute a smaller fraction of the diet of feedlot finished cattle compared to pork and poultry diets, the much higher feed conversion ratio far outweighs this difference. It should be noted, however, that pastured beef actually performs quite favorably on this metric. With returns of 69%, this is far superior to all other systems considered. However, for gross chemical energy returns it performs poorest at 1.6%. This difference underscores the necessity of considering efficiency from a variety of perspectives. While ruminant production on nonarable lands certainly provides opportunities to produce food products for human consumption without putting pressures on arable croplands used to produce human-edible products, the amount of biomass directed through grazing ruminants at the expense of availability to other trophic webs is substantial. This efficiency bears careful consideration with respect to biodiversity objectives.

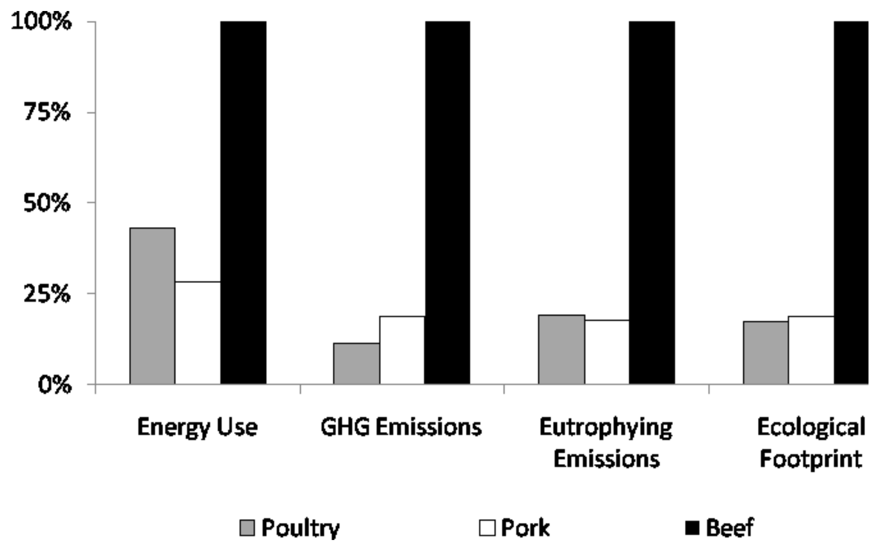

Figure 13.1. Comparative life cycle impacts of live-weight poultry, pork (commodity) and beef (feedlot finished) production in the United States.

For pork and poultry, comparative efficiencies are less clear cut. For example, pork production delivers the highest human-edible energy returns on industrial energy investment by a considerable margin, largely due to the small fraction of energy-intensive animal products used in poultry feeds, which serve to greatly increase supply chain energy use. In contrast, due to better feed conversion ratios, returns on human-edible energy and gross chemical energy are superior for poultry production (Figure 13.2).

Clearly, substantial differences in efficiencies within and between livestock systems point towards improvement opportunities for each sector and for the production of livestock as a whole. Such opportunities provide important avenues to improve the efficiencies with which we meet human needs for food energy. However, regardless of the efficiencies which might be achieved under different production norms, efficiency should not be seen as an end in itself but rather solely as a means to meet the overarching imperative of sustainable scale.

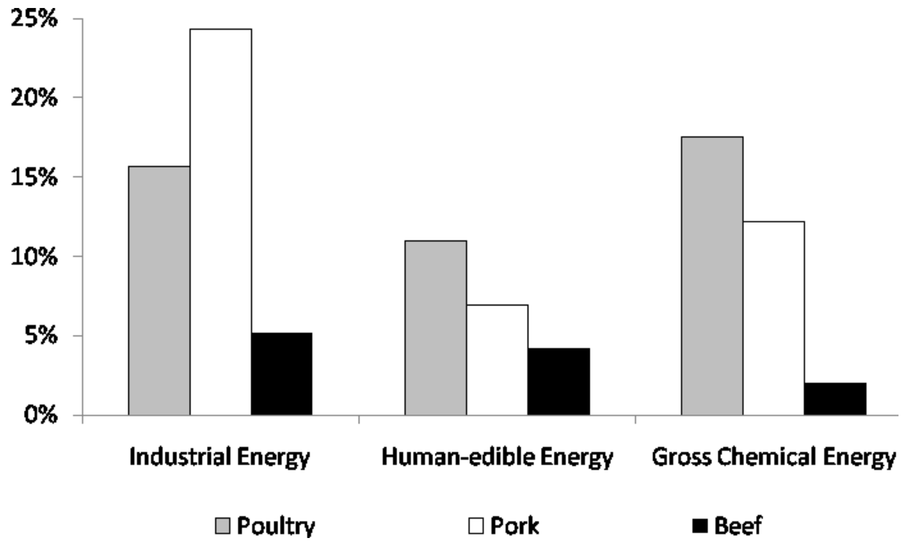

Figure 13.2. Return on investment ratios for live-weight poultry, pork and beef production in the United States.

Comparative Efficiencies Of Food Products And Dietary Choices

Growing awareness of the role of food systems in anthropogenic environmental change has stimulated increased interest in evaluating the comparative environmental efficiencies of food products and consumption patterns. In particular, a number of researchers have compared the environmental performance of plant and animal products, and of plant-based versus meat-based diets (Carlsson-Kanyama 1998, 2004; White 2000; Gerbens-Leenes and Nonhebel 2002; Reijinders and Soret 2003; Zhu and Ierland 2004; Deutsch and Folke 2005; Eshel and Martin 2006; Garnett 2007). What emerges is that, in most cases, livestock products are considerably more resource and emissions intensive than are plant-based alternatives. This is a somewhat intuitive observation given the inefficiencies inherent to biological feed conversion, which serve to compound the impacts per edible unit of livestock product produced compared to the feed inputs consumed.

More recently, large scale studies funded by the European Commission have evaluated the macroscale environmental mitigation potential of alternative dietary patterns for the

EU as a whole (Weidema *et al.* 2008; Tukker *et al.* 2009). For example, Weidema *et al.* (2008) found that meat and dairy products contribute 24% of consumption-related environmental impacts in the EU-27, and that changes at the level of primary production are most important for reducing supply chain impacts. Tukker *et al.* (2009) subsequently investigated the mitigation potential of dietary changes in this region. They found that the Europe-wide adoption of a more plant-centric Mediterranean diet would lead to moderate reduction in total impacts, but that first and second order effects related to changes in consumer spending (i.e. spending money saved on lower-meat diets elsewhere) would offset this benefit. In other words, such a dietary change would have negligible impacts in terms of overall environmental impacts in the EU. This somewhat defeatist acknowledgement of the rebound effect should, however, be tempered by the observation that policy and market interventions that serve to raise the cost of livestock products in concert with reducing consumption can and should be implemented to curtail any such trends. This is, in many ways, in parallel with the well-established Jevons Paradox, whereby increases in technological efficiency often result in increased resource use overall unless coupled with appropriate governance interventions (Binswanger 1993, 2001).

Here, it was observed that the environmental implications of meeting recommended levels of protein consumption globally differed by close to two orders of magnitude if provided in entirety by either livestock or legume sources. While neither of these extremes is realistic, they nonetheless point strongly to the relevance of food policy to achieving both efficiency and scale objectives via the promotion of low-impact diets – in particular, diets lower in livestock products. It bears noting that the majority of soy produced globally is currently consumed by livestock, and that the US alone produces sufficient soy to meet the protein needs of the entire developed world. According to FAO (2006) projections for soy production in 2050, global production levels at that time would be adequate to supply the entirety of human protein needs at USDA-recommended consumption levels.

As estimated by Waterlow (2001) and Young (2001), a balanced diet need not contain more than 20 grams of animal protein daily. At present, the livestock sector provides an amount of protein equivalent to 30 g per person per day globally, with a variation of six to eighty grams between countries (Flachowsky 2007). In other words, if distributed equitably, balanced diets could be achieved at present globally whilst reducing the current scale of the livestock sector by 50%. However, the contribution of animal protein to global average diets is anticipated to increase from 33% in 2003 to 50% in 2050 (Steinfeld *et al.* 2006), which will seriously undermine resource use and emissions intensity efficiency objectives for consumption patterns, as well as exacerbate problems of sustainable scale.

It should also be noted that the majority of this increase will occur in developing tropical and subtropical regions, where livestock production efficiencies are currently much lower (Steinfeld *et al.* 2006; FAO 2009). These currently account for two thirds of the global ruminant population, but contribute only one third to ruminant protein production for human consumption (Herrero and Thorton 2009). Achieving productive efficiencies comparable to those in temperate zones would vastly improve the global efficiencies of the livestock sector as a whole. This must include attention to the spectrum of water, fuel use, soil and nutrient use, and feed use efficiency considerations.

13.2.7 Policy And Governance For Scale Objectives In Livestock Production

Given that livestock currently provide 17% of food energy and 33% of protein consumption globally, and that this industry is central to the livelihoods of 1.3 billion people worldwide (Herrero and Thorton 2009), instituting widespread change for sustainability objectives will be challenging. What becomes clear from the current analysis, however, is that the scale of the global livestock sector, at present and looking forward, is fundamentally unsustainable. Strong governance institutions and policy regimes are therefore urgently necessary to rein in this sector. Important questions thus include: (1) what are the most appropriate governance and policy options in each domain of concern; (2) to what extent can existing governance regimes contribute (and what are the gaps); (3) which specific policy options might be entertained to operationalize scale-oriented environmental governance objectives; and (4) what do these imply for regulating the livestock sector?

Following Daly and Farley (2004), ecological economics has three policy goals – sustainable scale, just distribution, and efficient allocation – each of which requires an independent policy instrument. Ensuring sustainable scale is prioritized because it implies putting constraints on what were historically open access regimes to specific ecosystem goods and services such as natural resources and waste sink capacities. This subsequently raises questions of ownership, or distribution – which, from an ecological communitarian perspective, encompasses the primary good of ecological integrity. Once scale is established and, at a minimum, just distribution is defined as that which ensures sustainability, then policies geared towards efficient allocation may be determined. Daly and Farley (2004) further suggest that good policy will (among other things) result in the appropriate level of macro-control at minimum cost to micro-level flexibility, allow for a sufficient margin of error when dealing with the environment, and correspond to the Principle of Subsidiarity, whereby the policy regime is operationalized at a level commensurate with that of the issue it addresses.

For the three environmental dimensions of global livestock production considered here – greenhouse gas emissions, biomass appropriation, and reactive nitrogen mobilization – the appropriate level of governance operationalization is at the global scale. For greenhouse gas emissions, this is because the climate system on which they impact is a global-scale system. Emissions will contribute to climate change equally, regardless of where they occur, and the depletion of atmospheric assimilatory capacity and climate change effects will be global in scope. Similarly for reactive nitrogen, the source of mobilized nitrogen is the global atmosphere. Although many of the downstream effects will be experienced at local and regional scales, the nitrogen cycle is a global biogeochemical cycle, and transport of reactive nitrogen species as they cascade through ecosystems is a transboundary phenomenon. For biomass appropriation, it might seem odd at first glance to consider this a global-scale concern. After all, biological productivity is context specific, and the scale of use impacts on local environmental concerns. However, biomass management is also a key consideration in the global carbon balance, with significant implications for atmospheric CO_2 concentrations. It is similarly a primary determinant of reactive nitrogen flows. Moreover, the aggregate scale of global appropriation ultimately plays a determining role in the level of biodiversity that ecosystems can support. It is, in the least, arrogant to treat biodiversity, which is the product of billions of years of evolutionary history, as a thing that can be owned by humans. It is more problematic still to restrict ownership or management rights for biodiversity to extant generations within the confines of politically defined nation states. Since biodiversity is critical to ecological integrity, and hence to sustainability, it is by nature the concern of the global human community within and across generations.

As has already been discussed, the concept of “safe operating space” advanced by Rockstrom and colleagues (2009) provides the scale of anthropogenic activities in each of these areas which might be considered sustainable. However, given current limits to knowledge, it is difficult to know with certainty what margin of safety the suggested boundary conditions provide.

The Precautionary Principle, adopted by the international community through Agenda 21 and the Rio Declaration at the 1992 United Nations Conference on Environment and Development (UNCED 1992), stipulates a reverse onus of proof, whereby the lack of scientific consensus on the probability of harm to the public or the environment is not sufficient excuse to forestall the implementation of safeguards. Rather, this principle provides the basis for policy makers to institute appropriate measures to protect the public interest in the absence of certainty. In the words of Principle 15 of the Rio Declaration, "In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (UNCED 1992).

In economics, this principle has been interpreted with respect to the decisions of the rational actor in light of irreversibility and uncertainty (Gollier *et al.* 2007), which might be said to be omnipresent conditions associated with macroscale environmental change.

Establishing boundary conditions and regulating human activities with respect to these conditions is hence imperative from a precautionary perspective. Moreover, given the reality of uncertainty and that the stakes are obviously high, this requires that interim boundary conditions be defined with a substantial margin of error, and on-going research and monitoring to assess their suitability.

The establishment of necessary reference points for managing within safe boundary conditions has already been operationalized within an important international governance agreement. The United Nations Conference on Straddling Fish Stocks and Highly Migratory Fish Stocks on the High Seas (UN General Assembly 1994) calls for the application of more "precautionary management reference points" than have been historically applied in order to prevent further overexploitation of fisheries resources. It further suggests a "precautionary toolbox" for fisheries resources management, including:

- "Promote the collection and use of the best scientific evidence;
- Adopt a broad range of reference points;
- Agree on a set of rules and guidelines;
- Adopt action-triggering thresholds;
- Agree on acceptable (tolerable) levels of impact and risk;
- Improve participation of non-fishery users;
- Improve decision-making procedures;
- Promote the use of more responsible technology;
- Introduce prior consent or prior consultation procedures;
- Strengthen monitoring, control and surveillance;
- Adopt experimental management and development strategies;
- Institutionalize transparency and accountability;
- Re-establish natural feedback controls."

This toolbox has obvious application to the development and regulation of "safe operating spaces" for human activities across a spectrum of relevant domains.

A further question which must be resolved is what policy approaches will be most effective to achieving the scale-based governance objectives of interest? For example, should throughput be controlled at the level of source or sink, and is price or quantity the appropriate control variable? According to Daly and Farley (2004), regulating throughput at the source is often more practical since this is effectively akin to damming a river at its narrowest point. However, this may imply changes to existing property rights regimes. In contrast, regulating at the level of the sink may sometimes be more practical because sink resources have historically been frequently unowned. Novel property rights regimes should therefore not meet with as much resistance as modifying existing arrangements. With respect to price versus quantity, quantity is the preferred control variable. This is because, once quantity is set, prices can adjust and fluctuate accordingly whereas when prices are controlled the market may nonetheless determine an inappropriate quantity. For this reason, quantity, in the form of hard, absolute caps on the aggregate scale of resource use and emissions are chosen here. However, it is recognized that, where feasible, the market may offer a desirable mechanism for allocating within regulated limits – for example, via tradable permits. This is because such an approach maintains micro-freedom in the pursuit of cost-effective means of meeting regulatory goals (Daly and Farley 2004).

For greenhouse gas emissions, the scale-based concern is the assimilatory capacity of the global climate system, hence regulating use of this sink capacity through first establishing a hard limit on aggregate emissions and, second, allocating emission rights via a variety of market or non-market mechanisms is likely the most feasible solution. In contrast, for reactive nitrogen mobilization, a hard cap at the source level (mobilization of atmospheric nitrogen in reactive form) may be more sensible than seeking to control outflows to sinks, which are highly diverse. For biomass appropriation, this is fundamentally an issue of source-level extraction, hence the necessary limitation is the scale of extraction.

Policy And Governance For A Sustainable Scale Of Greenhouse Gas Emissions

Of the three dimensions of global environmental concerns evaluated, only for climate change has any significant international governance apparatus emerged. The United Nations Framework Convention on Climate Change (UNFCCC), which was signed by 192 countries, is an international treaty whose goal is the prevention of dangerously destabilizing anthropogenic climate change. Towards this end, the UNFCCC (1997) provides a forum for the establishment of goals and rules with respect to greenhouse gas emissions. The Kyoto Protocol, the first international agreement created under the auspices of the UNFCCC, set binding GHG reduction targets for the European Union and thirty-seven other industrialized countries to reduce emissions to 5% below 1990 levels by 2012. More recently, parties to the UNFCCC met to negotiate an agreement to replace the Kyoto Protocol when it expires in 2012 at the Conference of the Parties (COP)-15 in Copenhagen. In many respects, the Copenhagen negotiations resulted in positive outcomes, including commitments from the three largest polluters (China, India and the United States) to a joint agreement, and securing consensus on the need to limit climate change to an increase in mean surface temperature at or below 2 degrees Celsius. However, the parties failed to agree on concrete global emissions reduction goals, whether for individual countries or in aggregate. In combination with the lack of arrival at, or even a timetable for establishing, a legally binding agreement, we hence remain without an effective, scale-oriented global environmental governance regime for greenhouse gas emissions.

The absence of such a regime renders it exceedingly difficult to manage greenhouse gas emissions from the global livestock sector. Beyond this, it should also be noted that agriculture generally, and the livestock sector specifically, have never been directly regulated under the UNFCCC, nor have there been promising movements in this direction – despite their significant contributions to overall emissions. In part, this may be attributable to the fact that we simply cannot do without food production, and that the current state of knowledge regarding low-GHG food production strategies is decidedly underdeveloped. Starkly put, targets developed for other sectors could not be met for agriculture under existing scientific knowledge, without stopping the use of mineral N fertilizer and limiting the scale of cultivation (EIS 2009). However, given the variable greenhouse gas intensities of different food production strategies – which are generally much higher for livestock compared to other food products – this blanket avoidance of directly regulating food systems emissions is unwarranted. Moreover, it must also be recognized that issues of agriculture, food security, poverty and changing climate are intimately coupled. It has been estimated that decreased crop yields in South Asia as a result of climate change could undermine food security for 1.6 billion people, and that, without adaptation, climate change effects may increase levels of malnourished children by 10 million in Africa by 2050 (Global Donor Platform for Rural Development 2009). Strong measures to curb food system greenhouse gas emissions will thus impact on conflicting objectives both within and between generations.

Despite the failure to directly include primary food production sectors in the existing climate governance framework, there has nonetheless been significant interest in this subject from numerous parties – particularly with respect to potential policies to decrease food sector emissions. One commonly endorsed mitigation strategy is the use of existing climate governance tools to improve soil and livestock management, such that agriculture can be leveraged to rebuild stocks of soil organic carbon and hence offset emissions in the short to medium term. This is not, however, a long-term solution since soil systems inevitably attain new equilibrium conditions, whereas throughput-related emissions are constant (Smith *et al.* 2008). Possible ways forward for directly including agriculture in existing climate governance agreements are mandatory reporting for agriculture under Land Use, Land Use Change and Forestry inventories, as well as increased attention to agriculture in both the Clean Development Mechanism (CDM) and Nationally Appropriate Mitigation Actions (NAMAs). At COP-13, the Bali Action Plan defined a course of action for Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (REDD and REDD-plus). The use of agriculture for enhancing soil organic carbon stocks could potentially be included in REDD-plus under future agreements (Global Donor Platform for Rural Development 2009).

The consideration of the livestock sector specifically in climate governance regimes has received even less attention in policy discourse, despite increasing attention in the research literature. For example, McAlpine *et al.* (2009) demonstrate that beef consumption is a key driver of both regional and global environmental change, and make a direct call for policy action. Suggested strategies include governance intervention to control the expansion of soybean production and extensive grazing in South America; policy measures to reallocate resources towards less impactful land use strategies; ending subsidies for beef production; and discouraging advertising which promotes beef consumption. Herrero and Thorton (2009) call for coupled policies, technologies and incentives to reduce livestock-related greenhouse gas emissions.

Although there has been considerable interest in the role of policy in shifting consumption patterns in order to reduce greenhouse gas emissions in the food system via reduced consumption of livestock products, to date it would appear that actual climate policy discourse has carefully avoided this subject. Nor is such an ambition necessarily easily achieved. In discussing possible means to shift diets in socially desirable directions Tukker *et al.* (2009) note that the insights provided by behavioural and systemic theories imply that existing societal infrastructures, including both physical and value structures along with artifacts of culture, habit and bounded rationality, likely seriously reduce the efficacy of food system environmental literacy campaigns and consumer-oriented labeling programs promoting low-meat diets.

One potential recourse that has been suggested by several authors are policies aimed at shifting consumption patterns by coupling health and environmental objectives, where the former are thought to provide more powerful motivation for the consumer than the latter (Tukker *et al.* 2009). Suggested measures to stimulate lower carbon diets by coupling health and sustainability messaging include: working directly with retailers to improve “health and sustainability marketing”; promoting coupled health and sustainability labeling campaigns and banning advertising of unhealthy dietary habits; introducing sustainable public procurement of healthy, meat-free foods; and providing indirect incentives such as lower insurance fees for individuals meeting specified health standards (Tukker *et al.* 2009).

The use of market measures such as carbon taxes to shift consumption patterns has also been entertained. Certainly, given the wide range of greenhouse gas emissions characteristic of different livestock products and of food products generally, this approach might prove particularly efficacious. The recent petition filed by the National Cattlemen's Association in the DC Circuit Court of Appeals against the US Environmental Protection Agency's "endangerment finding" with respect to greenhouse gas emissions, suggests that beef producers are already highly sensitized to such a possibility (National Cattlemen's Beef Association 2010). However, it must be kept firmly in mind that the market alone will not determine an appropriate scale of emissions. This must therefore be seen as a complementary strategy only.

Policy and Governance For A Sustainable Scale Of Reactive Nitrogen Mobilization

With respect to governance of reactive nitrogen mobilization, little global-scale infrastructure has been developed, although efforts by both UNEP and the EU are underway (Sint 2001). In particular, the International Nitrogen Initiative has emerged as a leading coordinator at the nexus of reactive nitrogen research and policy discourse. To date, specific reactive nitrogen species have been the subject of international regulation – for example via the LRTAP Convention and Gothenburg Protocol – but not reactive nitrogen, generally. The establishment of the Global Partnership on Nutrient Management by UNEP in 2007 represents an important first step towards coordinating actions to improve nutrient use efficiency in agriculture and reduce losses to the environment, in particular to aquatic systems. The Task Force on Reactive Nitrogen, established under LRTAP to coordinate science and policy development in Europe, is similarly instrumental.

At a regional scale, this issue has certainly gained prominence over the course of the last decade. For example, the US Environmental Protection Agency has regulated the disposal of feedlot manures, as well as end-of-pipe emissions (Fisher 2001) to reduce water and air pollution, and ceilings for NO_x emissions have been implemented in Europe via the Gothenburg Protocol. Nutrient losses to water are similarly addressed by several marine conventions, including the Oslo and Paris Commission for the North-East Atlantic, the Barcelona Convention for the Mediterranean Sea, the Black Sea Convention, and the Helsinki Commission for the Baltic Sea.

In large part, these initiatives have been motivated by particular regional concerns, including: air pollution and subsequent impacts on human health in Europe; coastal eutrophication from untreated human sewage in Latin America; and the eutrophication of Chesapeake Bay as well as the dead zone in the Gulf of Mexico in the United States (Cowling *et al.* 2001). In contrast to greenhouse gas emissions, which will have the same effect regardless of where they occur, managing reactive nitrogen for environmental objectives is considerably more complex. The challenge is significantly compounded by the mobility of reactive nitrogen species, their variable effects in different ecosystems, their variable longevity, and the phenomenon of nitrogen cascades (Galloway *et al.* 2004, 2008). The need for an integrated approach is hence increasingly recognized (Fisher 2001; Sint 2001).

In light of the diverse environmental concerns related to reactive nitrogen and the challenges inherent to end-of-pipe management, the earlier suggestion that limiting the scale of nitrogen mobilization represents the most promising recourse would seem reasonable. Nonetheless, restricting the scale of reactive nitrogen mobilization has not been seriously debated to date. Given the pivotal role of food systems, and of livestock production specifically, in contributing to reactive nitrogen mobilization, it should not be controversial to suggest that governance regimes targeting the livestock sector hold the most promise for responding to this challenge. This suggestion is strongly supported by the recent Barsac Declaration (2009), initiated by the Nitrogen in Europe research programme in 2009 and thus far signed by over 250 scientists and other experts, which explicitly recognizes the need for policies to curb meat consumption, particularly in developed countries.

At a global scale, this will require the establishment of a ceiling for total allowable N_r mobilization – perhaps based on the sustainability boundary condition proposed by Rockstrom and colleagues (2009) - with regional allocations which take into account the current distribution of N_r. For example, Asia currently mobilizes the most reactive nitrogen, whereas per capita mobilization is highest in North America. Both regions suffer from an excess of N_r. In contrast, much of Africa is considered N_r deficient (Cowling *et al.* 2001). Such allocations would serve to curtail the scale of livestock production on a regional basis, as well as stream livestock production sectors towards the most efficient norms possible. However, this must also be sensitive to the role of trade in transferring nitrogen between regions. Given the much higher reactive nitrogen mobilization associated with ruminant production due to low feed conversion efficiencies, coordinated policies to phase out ruminant production might also be particularly efficacious.

Policy And Governance For A Sustainable Scale Of Biomass Appropriation

Of the three global-scale environmental concerns considered, the state of discourse on potential governance regimes to limit the scale of aggregate biomass appropriation is least developed. More precisely, it is notable only by its absence. Although the mobilization of biomass flows towards human ends is a pivotal driver of environmental change, and this topic has been the subject of research and discussion in the peer-reviewed literature for at least fifteen years (for example see Pauly and Christensen 1995; Vitousek *et al.* 1997; Haberl *et al.* 2004, 2007; Krausmann *et al.* 2008), it has not yet attracted the interest of policy makers at a global scale beyond an interest in *increasing* the flow of biomass use for biofuel production.

Given that livestock production is the single largest consumer of biomass globally, any future governance regimes for biomass appropriation must necessarily directly address this sector. In particular, in light of the conservative estimate advanced here that by 2050 the global livestock sector alone will occupy 88% of humanity's safe operating space for biomass appropriation, it will be critical to determine priorities for the allocation of this limited resource.

Although, as suggested by Bishop and colleagues (2009), the absolute scale of biomass use must be constrained, this will similarly require allocative mechanisms which take into account regional productivity and biodiversity indicators. What is certain is that any such mechanisms will clearly variably affect different livestock sectors, primarily as a function of the feed conversion efficiencies they are able to achieve and the trophic levels of feedstuff. A priori, ruminant production will be fundamentally disadvantaged. Moreover, livestock production generally will be disadvantaged relative to agricultural crop production for direct human consumption.

Potential Synergies In Governance Regimes

Effective global scale governance is currently lacking for all three of the issues considered, and only partially developed for one of these (greenhouse gas emissions). There is nonetheless comfort to be had in recognizing that strong governance for any of these variables will likely have positive, synergistic impacts for the others. For example, given that ruminants contribute disproportionately to biomass appropriation as well as greenhouse gas emissions and reactive nitrogen mobilization per unit production, a governance regime that, among other things, limits the scale of biomass allocated to this sector would have multiple environmental benefits. Attention must be paid, however, to potential tradeoffs.

A recent International Nitrogen Initiative side-event held at the COP-15 in Copenhagen focused on the potential role of nitrogen management in reducing GHG emissions (Palm and Havlikova 2009). If the food sector is directly regulated under future climate agreements, the important role of improving nitrogen use efficiencies in reducing emissions will simultaneously serve to advance scale-based nitrogen management objectives. Similarly, governance regimes to manage the scale of anthropogenic perturbation of the nitrogen cycle will impact on a variety of themes, including; food security; energy security and industry; climate change; human health; and ecosystem health and biodiversity (Palm and Havlikova 2009). Any regimes to limit biomass use will ultimately also impact on climate and nitrogen management objectives.

Relevance Of A WEO To Governance For Sustainable Scale In The Livestock Sector

As noted by Bartley *et al.* (2009), the Doha Round of the World Trade Organization has continued the push for liberalization of trade in agricultural commodities through the removal of tariffs and subsidies. In other words, the market continues to play an ever more central role in determining what is produced and where. Ironically, there is no attention paid to the subsidies provided by nature, as most environmental costs are external to the market system. Although there likely are inter-regional comparative advantages in terms of food production efficiencies in the ecological economic sense, the free market system is largely unable to communicate such nuance. Moreover, the market system does not respect limits to sustainable scale. As a result, liberalization alone will do little besides promote a race to the bottom in terms of environmental regulations for agricultural production, whilst driving changes in production patterns and volumes in response to consumer demand and other factors rather than sustainability objectives.

This points strongly to the need for a level playing field in terms of scale-oriented food system environmental regulations, such as could by created by an institution like a World Environment Organization with strong policy-making and compliance mechanisms. In particular, in light of the considerable contributions of the livestock sector to the three global scale environmental concerns considered here, the need for a global institution which can allocate and enforce compliance with respect to use of the finite available shares of greenhouse gas assimilatory capacity, allowable reactive nitrogen mobilization and biomass appropriation is likely essential. Furthering these objectives would also be much expedited through coordination by a single body, since all are intimately related.

However, it must be recognized that, despite the pressing need for strong global environmental governance and the various arguments advanced for a World Environment Organization, the creation of such an institution is unlikely to occur in the short term.

Considerable political will is required to initiate its formulation and, once set in motion, it would likely require substantial time to formulate and operationalize the necessary environmental governance strategies and tools. In the interim, what then might be the opportunities for improved governance of the livestock sector for environmental objectives within existing institutional frameworks?

Certainly, repeated calls have been advanced for improved international environmental governance by actors both within and outside the United Nations program. Such calls have been formalized in diverse venues, including the Nairobi Declaration, the Malmo Declaration, the Cartagena package, and Paragraph 169 of General Assembly resolution 60/1 (UNEP Executive Director 2009; UNEP 2010). However, substantive progress towards such a reality has been exceedingly slow. The on-going discussions of the United Nations Environment Program on ways forward for improving international environmental governance represent yet another in a long series of halting initiatives, where strong positions and forward action are exceedingly rare. Most recently, a Consultative Group has been established by the Governing Council of UNEP, with the task of drafting a suite of recommendations for improving international environmental governance to the Governing Council/Global Ministerial Environment Forum. One of the more interesting elements that has emerged from this dialogue to date is the suggestion for reformulating the traditional "three pillar" conception of sustainability in recognition that a stable environment forms the necessary foundation for sustainability. The institutionalization of this recognition would certainly be much more conducive to underpinning scale-oriented governance initiatives than the current, Brundtland version.

For the most part, these discussions continue to focus primarily on tweaking the status quo – in other words, seeking more effective coordination and implementation of the existing, UN-administered multilateral environmental governance system. In particular, the potential for an enhanced role for UNEP, possibly involving grouping and oversight of existing and future multilateral environmental governance agreements, has received significant attention. However, beyond clustering agreements and providing more oversight to UNEP, better mechanisms to establish targets, monitor progress and enforce compliance would also clearly be needed. So, too, would be enhanced capacities to assist developing countries in meeting goals, including capacity building and access to appropriate technologies and resources (UNEP Executive Director 2009).

Such improvements may, indeed, be possible within the existing architecture of international environmental governance. In the absence of a stronger, centralized regime, they may represent the best hope. For example, directly addressing greenhouse gas emissions from the livestock sector via the previously described Kyoto mechanisms is clearly preferable to inaction and would require fairly modest modifications of existing agreements. Moreover, these could be expanded to more directly include issues related to reactive nitrogen mobilization (potentially by building direct linkages with the International Nitrogen Initiative and existing agreements governing emissions of specific reactive nitrogen species such as LRTAP). This is similarly true for incorporating biomass appropriation (supported by principles expressed in the World Charter for Nature, the World Food Plan of Action, and the Convention on Biological Diversity), since both have strong synergies with livestock-related greenhouse gas emissions. Indeed, given the current lack of multilateral governance regimes for either of the latter two issues, coupled approaches may represent the most feasible short-term strategy.

However, the track record of multilateral environmental governance via the current UN system is not particularly auspicious. Despite considerable lip service to improved, coordinated governance (UNEP Executive Director 2009; UNEP 2010), global-scale environmental crises continue to accelerate. In particular, the trump of state sovereignty appears to systematically undermine the potential efficacy of UN-based multilateral environmental governance regimes. So, too, does the lack of an overarching international legal framework for international environmental governance, along with means to ensure compliance. Certainly, the recent failures in Copenhagen speak volumes about the weaknesses of the status quo approach, and underscore the need for a much stronger, global environmental governance system – both to institute an effective climate regime as well as to mobilize the necessary development of comparable regimes for global nitrogen and biomass management. In the absence of such a framework and a strong coordinating governance body, it seems unlikely that we will arrive at the mechanisms necessary to manage the global livestock sector for sustainability objectives. For this reason, despite the significant challenges to its establishment, pursuit of a robust World Environment Organization remains imperative.

13.2.8 Policy And Governance For Efficiency Objectives In Livestock Production

In contrast to the relative dearth of attention to scale-based governance of the livestock sector, there has been considerable interest in policies geared towards improving efficiencies in livestock production from a variety of perspectives. This reflects a growing awareness of the importance of the food sector in environmental change, and in improving resource use efficiencies in order to meet current and projected needs whilst minimizing the strain on host ecosystems.

Whereas scale-based governance might be thought of as a “top-down” approach, governance for efficiency objectives requires a “bottom-up” focus. This is because the potential for particular efficiency gains in the livestock industry will typically be context-specific, and hence most applicable at the local and regional scale. In this way, policies for efficiency brought to bear by regional and national governance bodies can contribute strongly to meeting overall scale objectives. This is consistent with the imperative for maintaining micro-level flexibility to maximize the effectiveness of macroscale policies (Daly and Farley 2004).

Policy recommendations have varied widely in foci. Many have focused on supporting research and development to improve resource use and reduce emissions in animal husbandry systems. Others have recommended technological fixes such as manipulation of ruminant microflora (which may be able to reduce methane emissions by as much as 10%), substitution of monogastrics for ruminants, and regulation of both manure nitrous oxide and methane emissions (Herrero and Thorton 2009). Funding mechanisms for research and education programs for low-emission diets for ruminants (particularly in developing countries), and payments to small-holders for undertaking management strategies which promote soil organic carbon sequestration have also been proposed (Herrero and Thorton 2009).

As reported by Cowling *et al.* (2001), average N-use efficiency in producing protein in livestock systems varies widely, from as high as 40-50% for poultry and eggs, 35-40% for dairy, and 30-40% for swine, to as low as 15-30% for beef. Although much of the remaining N fraction can be recycled via manure use for fertilizer, a large portion is inevitably lost to air or water. Cowling and colleagues (2001) advance a suite of policy recommendations for improving N use efficiencies and reducing losses from food production systems. Suggested strategies include: improving animal housing to decrease ammonia and nitrous oxide losses; maximizing cycling of nutrients between intensive crop and livestock systems; leveraging yield increases in livestock production via genetic manipulation; and improved tracking of site-specific N needs via the mandated use of nutrient balance sheets. These authors also argue for the development and application of best management practices schemes based on experimental farm research, introduction and tracking of performance-based standards for nutrient management, improved education for farmers, and dedicated research programs to identify and resolve knowledge and technology gaps. The application of "end-of-pipe" technologies such as riparian buffer strips to sequester leached N, constructed wetlands, improved irrigation and drainage systems, and new manure management technologies are similarly encouraged. In general, policies to support the development and application of precision agricultural techniques in nutrient management are promoted (Cowling *et al.* 2001).

Of course, since most intensive livestock production is predicated on arable crop production, improving efficiencies in livestock production will be much advanced by a similar focus on the underpinning crop production, processing and distribution chains – particularly for monogastric production, where feed provision contributes a large share of overall impacts. Towards this end, the Royal Society (2009) calls for a "sustainable intensification" of global agriculture through the funding of research aimed at improving crop genetics with respect to photosynthetic potential and nitrogen use efficiencies via breeding and genetic modification. Better soil management, ecosystem approaches, and producer education are also recommended. This will require identification and promotion of appropriate N fertilizer application rates, timing and placement. It will also require maximally efficient cycling of organic nitrogen sources, optimal crop rotations, and the incorporation of legumes (Flynn 2009). At the same time, it should be recognized that there is no single technological panacea, and that diverse, context-specific strategies and engagement of local stakeholders will be necessary (Royal Society 2009).

Also of interest from an efficiency perspective are policies intended to improve energy use efficiencies. Heinberg and Bomford (2009) suggest that the industrialization of farming has transformed food production from a historical net energy producer to a current net energy consumer, with roughly 7.3 calories of input for every calorie of output. In light of the finite nature of non-renewable fossil energy reserves, however, the removal of fossil fuels from the food system may be inevitable. Early action is therefore necessary to develop alternative production strategies in order to avoid future spikes in food prices and decreased food availability (Heinberg and Bomford 2009). This must include, among other things, a return to biological nitrogen fixation and improved nitrogen use efficiencies (Pelletier *et al.* 2008). It will also be served by integrated pest management strategies, and on-farm power generation from wind, solar and biomass sources (Heinberg and Bomford 2009).

Heinberg and Bomford (2009) also point out that consumer preferences have been shaped by mass media marketing, producing dietary patterns in the interests of major agribusinesses over those of health and environment. They encourage policy makers to undertake public re-education on the coupled health and environmental benefits of dietary alternatives, including a reduction in meat intake. Governments are also encouraged to adopt sustainable procurement policies to support environmentally superior food production practices. Other options include removal, shifting, or application of subsidies as required to influence production and consumption norms, and implementing taxes on resource throughput and emissions rather than products.

Others have highlighted the desirability of policies which use a combination of market and non-market mechanisms to further internalize the environmental costs of food production. According to Tegtmeier and Duffy (2004), the externalized costs of agricultural production are 9.4-20.6 billion dollars per year in the US alone. Burke *et al.* (2009) note that so long as livestock products are not priced to reflect the full environmental costs of their production, consumption and production choices will be sub-optimal. Possible strategies include the use of carbon taxes on inputs such as fertilizers (polluter pays principle), which would make less greenhouse gas intensive livestock products more competitive. The use of payment schemes to farmers for practices which maintain or enhance ecosystem goods and services are also a possibility (Bartley *et al.* 2009). However, it is important to keep firmly in mind that, although certain ecosystem goods and services may be amenable to commodification, in most cases non-market instruments will be essential.

Many of the recommended policy approaches to governance for sustainable scale globally are similarly applicable to governance for efficiency objectives at lesser scales. However, what must remain clear is that the latter is a necessary element of, but not a substitute for, the former. While significant opportunities do exist for improving efficiencies in the livestock sector, the ultimate goal must be to constrain its scale relative to biocapacity. This may require fundamental changes in the species composition and volume of livestock production, with commensurate changes in consumption patterns and volumes.

Of interest here is the potential role of the United Nations Food and Agriculture Organization (FAO) in facilitating improved food system governance for both scale and efficiency objectives. To date, the activities of the FAO have largely focused on improving agricultural productivity and food security (as defined in terms of access), with little emphasis on environmental considerations. This is symptomatic of the fragmented and reductionist approach to issue areas characteristic of the UN organization as a whole, which was formulated before environmental concerns gained significant prominence in the international arena. In light of a growing recognition of the interconnectivity of food production potential, food security, and environmental integrity, it would seem appropriate for the FAO to take an increasingly active role in setting the agenda for policy discourse regarding environmental sustainability concerns in global food systems. The 2006 report from the FAO titled "Livestock's Long Shadow: Environmental Issues and Options" (Steinfeld *et al.* 2006) provided a strong basis for such a discourse with respect to the livestock sector. More recently, the State of World Agriculture "Livestock in the Balance" report (FAO 2009) took bolder steps in expressly recognizing the need to bring environmental sustainability considerations to the fore. Also notable here was the suggestion that, alongside technological efficiency measures, questions of consumption level deserve serious consideration in governance of the livestock sector for environmental objectives. This represents a significant departure from their historical, productivity-oriented focus. As the sole existing international organization with a mandate specific to food production, the FAO is certainly well-positioned to further environmental objectives in the livestock sector via education, capacity-building, and as a strong voice in policy discourse. In particular, it might play an important role in altering projected trends in the livestock sector in developing countries with respect to species choice, production technologies, production levels, and alternative (non-livestock) food production strategies.

13.2.9 Limitations Of The Research And Future Directions

As with any body of interdisciplinary scholarship, a defining challenge of this work has been to find the appropriate balance between breadth and depth. One might argue that any of the three major components of the research presented here could have been taken on and developed more richly as stand-alone dissertation projects. And, indeed, it is my ambition moving forward to continue to flesh out and further develop each aspect of the work. At the same time, however, it was and remains my strong belief that the telling of a compelling and constructive story demanded the approach I have taken.

Sustainability is, by nature, a broad and encompassing concept. Its elusiveness can, in part, be ascribed to the fact that it defies a reductionist approach. Its resolution requires attention to the spectrum of normative, empirical and governance dimensions by which it must be understood and approached. The challenge of sustainability will not be resolved, or even substantially furthered, by otherwise excellent scholarship in the ivory towers of the academy. Rather, it will require the articulation and alignment of societal values, our empirical knowledge of the world we inhabit and the relationships in which we are embedded, and the governance institutions which are ultimately required to operationalize our sustainability objectives. This necessitates scholarship that transcends traditional disciplinary boundaries, and that is made accessible to the spectrum of social actors. I have endeavoured to bring such an approach to bear.

By the same token, however, the work presented is necessarily limited on numerous fronts. First, in terms of the treatment of sustainability, I have narrowed the scope of focus to a small subset of concerns. This is consistent with the ecological economic understanding of the minimum necessary conditions for sustainability. For global, industrial society, maintaining biospheric integrity is, without doubt, the necessary precondition for sustainability as viewed from any other perspective. For this reason, it provides the backdrop against which all else must be considered. This is not to discount the relevance of the wealth of social, economic and other considerations which must be brought to bear to understand and manage the systems considered for a diversity of sustainability objectives. These were, however, considered beyond the scope of the present work.

Similarly, the ecological communitarian vision of distributive justice that I have proposed as a suitable normative platform for the ecological economic understanding of sustainability is exploratory rather than complete. I have provided only a broad-stroke rationale for such a vision, which depends for its validity on an acceptance of ecological interdependence. I have suggested that ecological communitarianism represents a viable alternative to the overemphasis on individualism that permeates conventional economics, but I have not developed, beyond the most basic of assertions, why this alternative is more legitimate nor how such an alternative might be operationalized. Certainly, within our current cultural context of hyper-individualism, charges of implicit authoritarianism might be anticipated. In short, there is a need here for on-going development of the epistemological and ontological foundations of ecological communitarianism. The work has only just begun.

In modeling the biophysical environmental dimensions of livestock production and consumption, the work is similarly limited in several important respects. First, although life cycle assessment (LCA) continues to evolve, the framework is suitable to addressing only a narrow range of environmental performance criteria. Also notable is that I have considered only a very narrow subset of the performance criteria that must be brought to bear in evaluating the environmental sustainability of livestock production and consumption. At the global scale, I have examined only three scale-based concerns. There are many more of which we are aware, and likely as many of which we are not. Similarly, at the regional scale, the application of four performance measures along with three additional resource efficiency indicators is alone inadequate to understanding and acting on sustainability concerns. Moreover, many of these indicators need necessarily be linked with local and regional scale environmental sensitivities. Such dynamic modeling represents the next era of life cycle modeling, which must be sensitive to real-time, spatially explicit economy-environment interactions.

Second, while I have endeavored to produce models that are broadly representative of regional production norms by applying a consistent set of modeling techniques and assumptions, such models are inevitably simplified representations of reality. In reporting average values, they mask the considerable variability characteristic of such systems over time and space. As importantly, the limits of data availability as well as the application of system boundaries which result in non-trivial exclusions of environmentally relevant flows associated with the production systems of interest likely result in underestimates of their true environmental costs. For example, Garnett (2009) argues that life cycle assessment provides a limited understanding of food system greenhouse gas emissions, and the complex dynamics which underpin them because such research seldom accommodates indirect, second-order effects like land use change, opportunities costs of alternative land use strategies, and market-mediated outcomes.

Furthermore, although two alternative production technologies were considered, I have largely limited my treatment of livestock production to highly industrialized production systems. The intent here was to describe the status quo on the basis that we must know where we stand before we can determine where we need to be and what will be required to get us there. As suggested by the scenario modeling exercise undertaken here, even the most resource-efficient livestock production technologies are fundamentally unsustainable at the scale at which they currently operate, and further efficiency improvements alone will not take us the necessary distance. Moreover, it may well be, all things considered, that less resource efficient production technologies integrated into closed loop farming systems, which better serve socio-economic, animal welfare, and other sustainability concerns, are the most desirable end goal within a steady-state economy. In other words, efficiency concerns may legitimately be trumped by other considerations so long as sustainable scale is achieved. In such an economy, however, what is clear from the present analyses is that livestock products must necessarily play a much less important role in fulfilling human nutritional needs.

On the policy front this work has only just begun to explore the question of global environmental governance, the conditions and institutions necessary to its effective operationalization, and how these might be brought to bear in governing the livestock sector for sustainability objectives. Following the Principle of Subsidiarity, institutions at local and regional scales are also necessary to manage livestock-environment interactions of commensurate scope. The existence and effectiveness of such institutions has not been addressed here, but must certainly form a cornerstone of future research. Moreover, while the goal of a strong World Environment Organization must be kept firmly in mind, the establishment of such an institution will require the mobilization of tremendous political will. In the interim, there is potential to improve existing environmental governance regimes – for example, by establishing hard limits on allowable greenhouse gas emissions, and directly including the livestock sector in mandatory reduction commitments. Certainly, on-going discourse regarding future governance of the global nitrogen cycle should be informed by an ecological economic perspective on sustainable scale in place of the conventional economic emphasis on cost-effectiveness and efficiency measures. The FAO might potentially play a strong coordinating role with respect to translating environmental objectives into operational realities in the livestock sector.

Finally, if we accept the formula for the determination of environmental impact as a function of population, affluence and technological variables advanced by Ehrlich and Holdren (1971), this work is obviously further limited in scope. Although I have addressed the role of affluence in terms of per capita consumption levels of livestock products, and a range of technological variables with respect to what and how livestock are produced, I have largely ignored the population variable beyond the observation that increasing population is partially responsible for increasing aggregate demand. Like much research and policy discourse, I have effectively treated population growth as given. Of course, this needn't be so. Indeed, concerted efforts to limit population growth and reduce the size of the human population over time surely constitute two of the most powerful leverage points for ensuring the sustainability of the human endeavor. Moreover, this variable will certainly be a critical determinant of future challenges related to disease, conflict over scarce resources, and our ability to respond to natural disasters and changing environmental conditions. If our population was significantly smaller, the pressing urgency of reducing consumption levels and advancing eco-efficient technologies would be much alleviated.

Taken as a whole, however, I am confident that the framework developed and applied here represents a novel and important contribution to ecological economics specifically, and to our evolving understanding of the critical task of managing human/environment interactions – with governance of the livestock sector provided as a strong example. It does not replace the neoclassical economic model, with its focus on cost-effective means of maximizing human preference satisfaction. Rather, it goes well beyond this model in attempting to resolve one of its primary deficiencies, which is an inability to provide any basis for managing human activities in the interest of environmental sustainability. Moreover, by offering up ecological communitarianism as an appropriate normative foundation for economics, it provides an internally consistent basis for ecological economic modeling and its application to environmental governance for sustainability concerns.

13.3 Conclusions

This work should hence not be interpreted as an end point, but rather as an introduction, and as a roadmap for further research. The treatment of sustainability advanced here provides the departure point for a much richer discussion of sustainability issues in global livestock production and consumption, and of the human endeavour, generally. Sustainability is a complex, multifaceted objective. In the end, there will be no single framework capable of providing the information necessary to understand and manage for sustainability in all its richness. There will be no simple answers. The ecological economic research framework developed here is intended to help us understand the minimum necessary conditions only, and the barest outlines of the target we must aim for. The way forward is open. The choices we make will figure strongly in determining the course of our civilization, and the distances we will travel.

Two decades ago Wendall Berry (1990) reminded us that "eating is an agricultural act." "The industrial eater," he wrote, "is one who no longer knows or imagines the connections between eating and the land, and who is therefore necessarily passive and uncritical in his consumption choices." More recently, Michael Pollan (2006) added to this that eating is an ecological act, and that it is also a political act. What is certain is that our individual and collective food production and consumption choices and policies will play pivotal roles in determining the fate of our agricultural landscapes, the efficiencies with which we use limited resources and, ultimately, the integrity of the biosphere we inhabit. Given the role of the livestock sector in current and projected anthropogenic environmental change, in terms of targeted personal and public policy interventions intended to move humanity towards that elusive safe and sustainable operating space, their truly are some low hanging fruit to be exploited. It's time that we bring this awareness to the table.

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