Considerations for comparative evaluation of environmental costs of livestock and salmon farming in southern Chile

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ABSTRACT

The farming of salmon and cattle is a quite unique characteristic of the Lake Region in southern Chile. There, both activities are of great relevance for economic and social development of the region, and government authorities should face a difficult task when dealing with the decision-making process regarding prioritizing one or the other. Comparative environmental cost assessments are needed in order to properly regulate and manage these food producing sectors and in order to establish comparable environmental requirements when both activities take place in the same areas or regions. This paper reviews available information necessary to perform comparative assessments of environmental costs of livestock and salmon production focusing on simple mass balances of nutrients, especially nitrogen (N). We also explore the use of different scales for the comparisons, local (e.g. basins) and regional.The accounting of N loads of both activities at the regional scale, including freshwater and marine environments, showed a larger impact from salmon farming than from cattle farming, although at local levels the latter was in some cases much greater (e.g., Lake Llanquihue). Therefore, it is crucial to define the scale of the approach/comparison according to the impacts and effects that need to be controlled or mitigated. However, although N surplus and loads were identified as impacts, there are limited data on the associated effects except for the information on critical nitrogen loads or critical carrying capacities in lakes.

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INTRODUCTION

Food producing sectors such as intensive livestock farming and intensive aquaculture may have relevant environmental effects on ecosystems. Comparative assessments of their environmental costs are needed in order to properly manage and regulate such productive sectors and to establish comparable environmental requirements when both activities take place in the same areas or regions. Additionally, a significant problem is the need to recognize the values of environmental goods and services, that may be affected by terrestrial and aquatic farming; once this is done the challenge is to include them in relevant comparisons as part of the decision-making process towards sustainable development. Such exercises may help guiding local and regional authorities to adopt better decisions regarding farming priorities or to identify "preferable development pathways" (Tyedmers and Pelletier, 2007).

The Lakes Region in Southern Chile is a territory dominated by lakes, coastal channels, islands and fjords (39º to 44º S; 71º to 73º W). In this region, cattle farming had existed since European colonization times (*ca*. 200 years ago), yet environmental impacts to land and waterways have been more evident, or regarded as such, only after salmon farming started in Southern Chile in early 1980, when attention was focused on the potential deterioration of the water quality of these lakes (Soto and Campos, 1995).

The occurrence of salmon and dairy/meat production is a rather unique characteristic of the Lakes Region as both activities are of great relevance for economic and social development. Thus, government authorities often have to face decisions related to both activities which in this case are not being mutually exclusive.[2] This fact also offers the opportunity to carry out comparative analyses of environmental issues associated with both production systems. In addition, the exercise may enhance the possibilities for the integration of fish and dairy production (*i.e.* recycling salmon waste as an input to agriculture) as a way to reduce environmental effects (Teuber *et al*., 2005).

There are several potential methods for comparative studies of environmental costs associated with industry and different commodity sectors; one method is material flow analysis (MFA) as shown by Gowing and Ocampo-Thomason (2007) when comparing rice and shrimp farming, and Life Cycle Assessment (LCA) for shrimp farming (*e.g.* Mungkung and Gheewala, 2007). The later has been shown to be a valuable technique for the environmental evaluation of food production systems (Van der Werf and Petit, 2002) and it has been applied to several agricultural products in different countries (Cederberg and Mattson, 2000; Haas *et al.*, 2001). For agricultural products, the cycle considered is generally from "cradle-to-gate,[3]" for aquaculture products the corresponding would be from eggs to gate. These methodologies are usually focused on global effects, using broad-scale environmental impacts, such as total energy United States of Americage, CO_2 emissions, production of acid gases, etc. (Tyedmers and Pelletier, 2007). While local decision making at the farm level or farming area (both terrestrial and aquatic) often has to deal with local effects/costs and more importantly, with the benefits of one or other activity to the local interest or development, etc.

The objective of this paper is to review the information available in order to perform comparative assessments of environmental costs of livestock and salmon production in Southern Chile. We particularly explore the potential benefits of simple mass balances of nutrients, especially N, as an exercise and preliminary approach in absence of other more complete data. We also explore the use of different spatial scales for the comparisons, local (*e.g.* basins), and regional.

[2] Many tools for the comparison of environmental costs focus on activities which are mutually exclusive or alternatives such as rice farming vs. shrimp farming (Gowing and Ocampo-Thomason, 2007).

[3] The term **cradle-to-gate** is often used to refer to life cycle analysis applied to the overall performance starting upstream at the cradle of material and energy inputs extracted from the earth and ending at the "gate" before being transported for consumption.

Comparative impacts of livestock and salmon farming under different management practices

According to the categories of environmental impact assessed in LCA or other approaches (resource depletion, human health impacts, and ecosystem consequences) common impacts from livestock farming and salmon farming through all the productive cycle include; i) organic and inorganic outputs from the feeding/digestion process, ii) discharge of chemicals (pesticides, antibiotics, etc.), iii) energy uses for the farming process, and iv) use of feeding resources (*e.g.*, fish meal, maize, etc.). While some impacts are exclusive of livestock farming (*e.g.*, soil trampling and erosion) other potential impacts are exclusive to salmon farming (*e.g.*, those caused by escaped fish). However, these latter are usually not considered by LCA or MFA because they are related to local effects, such as biodiversity reduction, with presumably associated environmental costs which are usually unknown.

From all of the impacts mentioned above, (i) is one of the best studied for both farming systems, as it is related to the use of feeding resources and feeding process. Nutrient balances are only one component/portion of the relevant information for an LCA or MFA analysis; it seems a logical first step, however, to focus on them because more information is available, and also because impacts and associated consequences and potential environmental costs are better known by public and better considered in government regulations (*e.g.*, Environmental Impact Assessments, EIA). Clearly, we cannot underestimate other impacts such as discharge of chemicals, resource depletion for feeds, energy costs, etc. However, in this paper we focus mostly on the inputs/outputs of feeding and digestion processes and its environmental consequences. We disregarded a full LCA (or MFA) approach for the comparison of these production systems; this was due to information limitations but also to the value of the present approach for local decision making. More specifically, for the most part of this exercise we concentrated on nitrogen (N) balances because more information was available to compare salmon and cattle farming systems, but also because productivity and plankton biomass in those southern freshwater and marine environments may be more limited by N and therefore could be more sensitive to these inputs (Soto, 2002).

Cattle production systems generally have a low nutrient efficiency,[4] which represents potential risks of pollution to the environment and economic losses for the farmers (Jarvis, 1993; Oenema and Van den Pol-Van Dasselaar, 1999). On a global scale, N efficiency in all terrestrial animal production is estimated around 10 percent, while it is estimated at 7.7 percent for cattle production only. For beef and dairy farms studies in developed countries have shown N efficiency values ranging from 14 percent to 30 percent, and N surpluses or excesses to the environment of up to 470 kg N ha^{-1} yr^{-1} (Table 1). Also, low efficiencies have been reported for phosphorous (P) utilization; Haygarth *et al.* (1998) working with dairy systems in England have shown a surplus of 43 kg P ha^{-1} yr^{-1} with a P efficiency of 37 percent. In Chile Alfaro *et al.* (2005) reported P surplus of 37 kg P ha^{-1} yr^{-1} with a P efficiency of 10 percent in beef production in grasslands.

The low N efficiency of cattle production systems is caused by the inefficiency of the ruminant species in converting ingested N into milk and live weight gain. In dairy cattle, Van Vuuren and Meijs (1987) showed that the maximum N utilisation of lactating cows was 43 percent of the ingested N, whereas the average efficiency was about 15-20 percent. The unabsorbed N is excreted in dung and urine and directly deposited on pastures during grazing or accumulated in animal houses (Jarvis, 1993). Nitrogen efficiency vary greatly from country to country and within the same country, because of different cattle productions systems and management (*e.g.,* extensive all day grazing *vs.* intensive all day in housing).

[4] Efficiency estimates, consider all nutrient (e.g., N and P) inputs as fertilizers and feeds compared to what is retained by the animal.

TABLE 1
Nitrogen gate balance for dairy or beef farms in selected countries

Country	System	N surplus (kg N ha^{-1} yr^{-1})	N efficiency (%)	Reference
New Zealand	Dairy	131	30	Ledgard, Penno and Sprosen (1997)
Holland	Dairy	470	14	Aarts, Biewinga and Van Keulen (1992)
England	Dairy	270	20	Jarvis (1993)
Canada	Beef and dairy	288	17	Paul and Beauchamp (1995)
Sweden	Dairy	173	21	Cederberg and Mattson (2000)
France	Beef and dairy	150 -200	-	Le Gall, Legarto and Pflimlin (1997)
United States Of America	Dairy	-	19	Bacon, Lanyon and Schlauder (1990)

TABLE 2
N efficiency for salmonids in different reports, mostly for laboratory testing

Country	System	N efficiency (%)	Reference
Canada	Atlantic Salmon in tanks	41[fm]	Azevedo et al., (2004)
Canada	Rainbow trout in tanks	30[fm]	Azevedo et al., (2004)
France	Rainbow trout in tanks	30[l]-40[fm]	Burel et al., (1998)
Norway	Atlantic Salmon in tanks	35[s]-45[fm]	Refstie et al., (2000)
Norway	Atlantic Salmon in tanks	48-52[h]	Refstie, Ollic and Standald (2004)

[fm] diets mainly based on fishmeal
[l] diets containing lupin flour instead of fishmeal
[s] diets containing soybean flour
[h] Protein hydrolizate

Environmental impacts of intensive fish farming and particularly salmon farming have been well documented in the literature ranging from mild or negligible to high in some cases (Gowen and Bradbury, 1987). Comparatively, N and P retention efficiency are much greater in fish than in cattle, with N values ranging from 30 to 50 percent, although most common estimates are above 40 percent (Table 2) particularly for Atlantic salmon being estimated as 44 percent on improved diets (Kolstad, Grisdale-Helland and Gjerde, 2004; Refstie, Ollic and Standald, 2004). Phosphorus efficiency in fish on the other hand is quite stable around 28 to 35 percent even under diverse protein origins (Denstadli *et al*., 2006; Glencross *et al.*, 2006). For each ton of harvested salmon fed with dry pellets (*circa* 44 percent protein), 35-78 kg of N and 7-10 kg of P are released into the environment (Ackefors and Enell, 1994; Niklitschek, Soto and Lafon, 2006) Nitrogen is mostly lost as dissolved matter: ammonia (62 percent) and urea (9 percent), the remaining solid portion is lost in the faeces (29 percent).

Nevertheless the N surplus to the environment per kg of salmon produced has been decreasing as the feed conversion ratios (FCRs) have been improving during the last decade. In early 1980 economic FCR[5] values were between 4 and 6, when salmon feeds were locally made as moist pellets; during the last 20 years these feeds have been replaced with dry commercially manufactured steamed pellets, characterized by their high protein and low fat content (Tacon, 2005). Such feeds provide a much better FCR of 1.6-1.8. Yet with more recent lowering protein contents and increasing lipids (up to 40 percent by weight) salmon feeds can yield economic FCRs bellow 1.3 (Tacon, 2005). Therefore, in 20 years the industry has increased feeding efficiency at a rate not seen

[5] Feed conversion ratio provides the relationship between the amount of feed used (total dry weight) and the amount of fish harvested (total wet weight). The economic FCR takes into account all the feed used, meaning that the effects of feed losses and mortalities are included.

[6] According to information provided for the WWF Salmon Dialog; Infante and Pizarro's Report (http://www.worldwildlife.org/cci/dialogues/salmon.cfm)

for other animal production. Today in Norway, economic FCR can be very close to 1.2 although in Chile it is still a little higher (1.35).[6]

In contrast, food conversion efficiency for dairy cattle (amount of milk solids produced per kg of dry matter intake) has not improved noticeable in the last decade (Oldenbroek, 1988; Thomson, Kay and Bryant, 2001).

There are numerous studies offering estimates for N balance of salmon farming, but few provide loads on an area basis (*e.g.*, kg ha^{-1}) as shown for cattle in Table 1; yet there are some case studies amenable for comparison. At a large scale approach, in the whole Baltic sea where the Nordic salmon farming industry produced 200 000 tonnes in 1994, discharges to the sea of N and P were equivalent to the amounts in untreated sewage from a population of 3.9 and 1.7 million people, respectively (Folke, Kautsky and Troell, 1994). Ackefors and Enell (1994) assuming 60 kg of N surplus per ton of salmon, estimated that the annual loads of nitrogen and phosphorus from salmon farming in Nordic countries were around 15 and 2.5 thousand tonnes per year respectively by 1990. However, these values represented less than 1 percent of the total loads coming from other human activities (including agriculture and livestock production).

If we focus on the impacts at the farm site scale it is possible to make better estimates of loads per area after evaluating the sedimentation shadow below the cages. In Norway average production per site is about 1 200 metric tonnes of salmon with maximum production around 4 500 metric tonnes of fish at a site, although few farms are that big. Kutti, Ervik and Kupka-Hansen (2007) describe a fjord farm site of *circa* 2 900 tonnes with a shadow area of 9 000 m^2. However, because the cages at this site have a mobile floating system the real "impact zone" could be of approximately 76 000 m^2 which diminish the organic matter load per unit area and therefore the effects on benthic ecosystems.

Comparing livestock and salmon farming impacts and environmental effects/ costs in southern Chile

Although global environmental costs of activities such cattle and salmon farming, can be indeed compared with approaches such as LCA, it may not be very practical or relevant for the local decision making. For example the contribution of cows to green house gases is not an issue in agriculture in Southern Chile, while the exports of excess nutrients to aquatic environments with eutrophication potential and negative effects on biodiversity can be much more relevant to local communities and decision making.

Production and impacts related to nutrient balances and efficiency

The Lakes Region, located in the south of Chile has suitable climatic and edaphic conditions for cattle production. As a result, 56 percent of the national cattle herd is concentrated in the Lakes Region relying on natural and improved pastures for feed. The Lakes Region produces 70 percent of the country's milk (ODEPA, 2005; Anrique, 1999), accounts for 80 percent of the dairy farmers, and for 67 percent of the total land dedicated to dairy production (Anrique, 1999). In 2004 meat and milk production in Chile were 208 258 tonnes and 2 250 million of litres, respectively (Banco Central de Chile, 2005). Most of this cattle production is consumed locally with only US$ 23 million and US$ 84 million exported in 2004 for meat and milk, respectively (ODEPA, 2005).

At least 80 percent of the Chilean salmon and trout production is concentrated in this same region. Nowadays, Chile is the second salmonid producer in the world, generating important income for the national economy (FAO, 2006). In 2005, the production of Chilean salmon and trout was 598 thousand tonnes, which generated an income (mostly from exports) of US$ 1 700 million (Chilean Salmon Farming Association, SALMONCHILE, 2006). The Chilean "salmon farming" industry (thereafter referring to culture of both salmon and trout) is mostly based on fish cages, which are located in lakes for a large proportion of the smolt production and in the inland seas and fjords for the grow out.

One of the major issues for the comparison between livestock and salmon farming relates to the delimitation of affected areas, understanding the fate of nutrients and estimating their effects. For example, in the case of livestock, some surplus nutrients are retained in the soils until being reutilized or they are lost to bacterial degradation with CO_2 emission, while some relevant amounts may go directly to waterways. Similarly, nutrients from salmon farming can be reutilized quickly in the water column around cages while those settling on sediments could be lost to bacterial degradation, and some proportion can be recycled.

Nitrogen gate balance for selected cattle farms in Southern Chile, calculated as the difference between N entering the farm (*e.g.*, fertilizers and concentrated feed) and N exported from the farm (*e.g.*, milk and/or meat), are shown in Table 3. High variability exists between the different dairy and beef production systems. For example, N inputs are 8 times larger in intensive dairy production systems than in those for beef or extensive milk production systems (Table 3). Nitrogen efficiencies ranging from 16 to 28 percent are similar to those observed in developed countries (Table 1), with the lowest values obtained in intensive production systems. A case study carried out in 69 Chilean dairy farms (Table 3) showed a wide range of N balances, going from 10 to 99 percent of N efficiency. On the other hand, the highest efficiency values resulted in very low or negligible N inputs to the farming areas, being these the best management cases when careful attention is given to feeding grounds and feeding conditions, such as timing and movement of the cattle.

When attempting to compare cattle *vs.* salmon production it is important to identify typical or average cattle production systems, since they are highly variable compared to salmon farming systems which are now a day more homogeneous.

Indeed, most salmon farms, both freshwater and marine, are of intensive production with a narrow variability range in fish density and management conditions (Rojas and Wadsworth, 2007). In addition, it is crucial to base such analyses of environmental impacts on accurate data, which may prove to be problematic, particularly in this case where more information is available from salmon farming than from livestock farming. Probably a better analysis could be achieved by implementing a characterization of cattle production systems with clear definitions of homogeneous farm types/systems.

Nitrogen balances for salmon farming are mostly available from studies funded by the Chilean Fisheries Research Fund (FIP[7], 2007) for freshwater production; nevertheless some estimates are possible for marine sites. In Chile salmon farming characteristics such as farm structure, feeding systems and fish densities are quite

TABLE 3
Nitrogen gate balances and Nitrogen use efficiency for selected cattle farming systems in southern Chile

Farm system	N input (kg N ha^{-1})	N output* (kg N ha^{-1})	N surplus to environment (kg N ha^{-1})	N efficiency ( )
Beef, experimental farm	87	24	63	28
Milk, experimental farm	310	65	245	21
Milk, intensive housing	515	83	432	16
Milk, intensive grazing	505	87	417	17
Milk, farm survey (n=69) (range of values)	87 (8 – 236)	20 (5 – 43)	68 (0 – 193)	28 (10 – 99)

* in milk and meet

[7] Most studies and reports available on line under "Proyectos" in the FIP site, www.fip.cl

similar to those in Norway. According to Rojas and Wadsworth (2007) fish density in marine farms would be between16 and 20 kg m^{-3} and between 0.11 and 0.42 tonnes m^{-2} in farm sites averaging 15 000 m^2. Considering that the FCR are somewhat higher in Chile that in Norway, with an estimated surplus N of 60 kg per ton of salmon, the total surplus could fluctuate between 4 and 10 kg m^{-2} for normal production biomass ranging between 2 000 and 4 500 tonnes respectively. The area occupied by cages of farms on the same range varies from 10 000 and 18 000 m^2. An aquaculture farm concession will typically have 10 to 15 ha; therefore, assuming that N stays in the allotted area the surplus loading could be very high, up to 16 000 kg ha^{-1}. However, this does not consider the rate of dilution and rapid transport which could in reality reduce significantly the N surplus/load per area. Dilution could be especially relevant in some marine areas with large currents and tides while in others could be minimal (Soto and Norambuena, 2004); in turn, deep mixing in the lakes can also contribute to nutrient dispersion (Soto, 2002).

A general problem in this type of analysis is the difficulty to define boundaries for the areas impacted by salmon farming, while this is easier in the case of cattle since we refer to the area (pasture land) as being actually used by them or a specific watershed. The effect of salmon farming in cages can be referred to the whole area of a water body, lake, fjord, coastal zone, etc. It could also be delimited by the licensed area (or assigned area), or to the area being actually affected by inputs of organic matter.

Another important aspect, and often a requirement for meaningful comparisons, is that salmon farming and livestock effectively share a common physical area. In Southern Chile this takes place mostly during the salmon freshwater phase - smolt production - when their effects impact over common hydrographic basins shared with livestock. While most of the salmon grow out phase, where the largest amounts of feeds and nutrient inputs take place, is done in the marine coastal environment where cattle farming is less relevant. Therefore, regional level comparisons may be more meaningful if the impacts on the costal seas are to be included.

To illustrate the former situation at a local scale, we analysed a single watershed, the Maullin river basin which includes large Lake Llanquihue. This basin produces approximately 4 000 tonnes of salmonid smolt (2005 figures), accounting for around 30 percent of the country's freshwater production phase. This biomass should release annually 240 tonnes of N to the lake (Table 4). Meanwhile the same basin has approximately 200 tonnes in the United States of America and cattle heads (82 000 tonnes live weight biomass) with a nitrogen surplus to the environment of 2 050 tonnes of N, assuming a surplus of about 25 kg per ton of live weight cattle. For cattle systems, it is likely that a portion of such surplus could be retained by the soil and be recycled on the spot. Although the figures here are not well known, undoubtedly an important proportion could get to the waterways and finally to the lake. On the other hand, it is known that about 60 percent the N surplus from salmon farming remains in the water

TABLE 4
Annual Nitrogen surplus estimated for salmon farming at different scales and environments in Southern Chile in 2005. A fixed value of 60 kg N has been used as surplus to the environment for each ton of salmon produced

Site/spatial scale	Species/stage	N surplus kg ha^{-1}
Lakes Region, coastal marine area	Salmonids (adults)	40-50*
One typical marine farm in the X Region,	Salmonids (adults)	1145**
Lake Llanquihue basin	Salmonids (smolts)	2.7***

* Considering a salmon production of 450 thousand tonnes and an estimated total area of 600 thousand ha where salmon farming takes place in the inner Seas of the X Region, and where the surplus N can expand.

** Considering an average production of 2 000 tonnes per farm and an average farm area of 11 ha (Niklistcheck Soto and Lafon, (2006) with a dilution area ten times larger (110 ha).

*** Corresponds to an estimated smolt production of 4 000 tonnes to a lake area of 87 000 ha.

column with potential recycling by microalgae while a smaller proportion sinks to the sediments where some denitrification could also take place on site.

On a regional, much larger scale, considering a salmonid production in 2005 of 450 tonnes for the whole Lakes Region in Southern Chile (Subsecretaría de Pesca, 2005), the total N load should have been approximately 27 tonnes for that year. About 60 percent of this N had been released directly into the water, while the remaining N and most of the P had been deposited in the sediments, from where it could leach into the water at locally variable rates. Conversely, cattle biomass in the same region was around 714 tonnes with an estimated total N surplus of 17.8 tonnes.[8] Therefore, at the larger regional scale salmon input was larger. Table 4 offers different level of impacts (N surplus) when considering such different scales. It is clear that the ecosystem boundaries and spatial scales are quite relevant when achieving conclusions from such comparisons.

Consequences and environmental costs

Livestock farming, and especially poor manure management is recognized as a major source of ammonia (NH_3) emission to the atmosphere, which has been shown to cause soil acidification in Europe (Van Bremen *et al*., 1982; European Environmental Agency, 1995) and nitrate (NO_3^-) accumulation in ground and surface waters worldwide (European Environment Agency, 1995; Powlson, 2000). In addition, the importance of nitrous oxide (N_2O) emissions related to the denitrification process has become more apparent (Chadwick *et al*., 1999). It has been estimated that livestock production contributes between 37 to 82 percent of the nitrogen input, and between 27 to 38 percent of the phosphorus input, to surface waters of Western Europe (Isermann, 1990; Hooda *et al.,* 2000; Gerber *et al*., 2005). Oyarzun and Huber (2003) have also estimated relative inputs of Nitrogen to watersheds in southern Chile and the values in agriculture, cattle farming areas are significant. These can be also enhanced by fertilization practices.

On intensified cattle production systems (e.g., European cattle farming) there is an intensive use of soil and a high proportion of the farm land is contributing to animal production (Aarts, Biewinga and Van Keulen, 1992). However, especially in developing countries (e.g., Southern Chile) cattle production is based on extensive grassland systems where an important proportion of the farm could be woodland or shrub areas. On the other hand, cattle woodlands/pasture areas act as carbon "sinks", thereby generating some positive environmental effect. Nevertheless, cattle farming takes places in areas which were once covered by forest and these ecosystems have changed in significant ways (Etcheverria *et al*., 2006) with no return to the original situation as long as current farming practices continue, such costs have not been evaluated. According to this, the carbon balance in the farm should be taken into account in environmental impact assessments but even more so when using global tools such as LCA. Other effects of livestock such as soil erosion and contamination of land and water due to the use of chemicals (pesticides, antibiotics, etc.) could have important negative consequences on local biodiversity, land and water productivity, etc.; their magnitude highly depending on management practices and feeding strategies. Unfortunately, we can only address and estimate impacts (e.g., N surplus) but not the effects since there is no available information on biodiversity loses or other ecosystem services being deteriorated due to cattle farming in this region.

In the case of salmon farming it is well known that the proportion of non-consumed feed, faeces and excretes, all containing nitrogen (N) and phosphorous (P) is variable between farms, mostly depending upon the feeding technology. Automatic feeders, monitoring and feed-back devices, as well as improved feeds with higher nutritional

[8] 25 kg of N per ton of bovine live weight (Anrique, 1976).

and pellet manufacturing standards (Cho and Bureau, 2001), have been incorporated in most fish farms since the mid 1990s. Therefore, the proportion of wasted food has been declining rapidly from levels of 20 to 30 percent in the 1980s to less than 5 percent in farms using state of the art technologies in the 2000s (Nash, 2001).

All the environmental effects should be considered if using LCA; however, for practical purposes, particularly considering local effects with local environmental costs which require specific regulations and decision making, it seems more convenient to consider the involved risks of having certain environmental costs. Such costs maybe for example eutrophication, or losing water quality, or losing biodiversity due to some of the above mentioned effects from livestock or salmon farming.

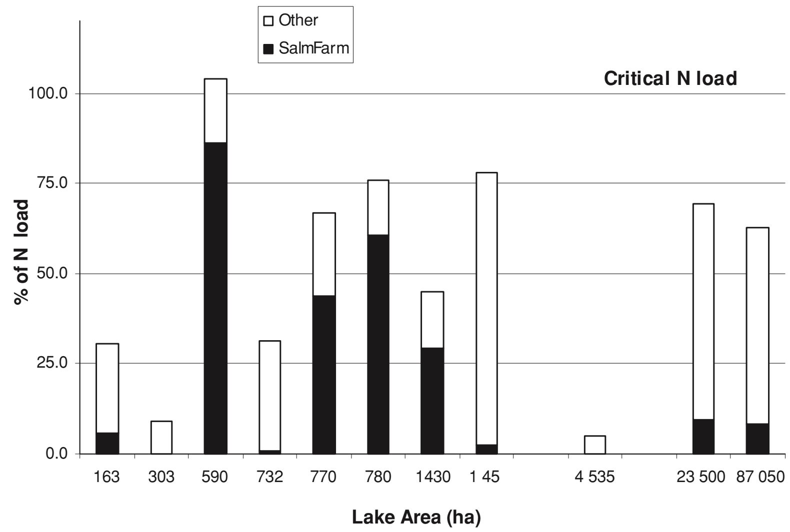
To estimate environmental costs is necessary to clearly define the effects and often we only have information on the potential impacts (e.g., the release of N ha^{-1} or the total N load to an area, etc.), but not enough information on costs to society as ecosystem services are being lost. Fore example, it is assumed that nutrient inputs as N and P will have negative consequences mainly related to eutrophication; however, concrete data on eutrophication are often lacking and risks are usually estimated through carrying capacity models more usually build for P in freshwaters under the assumption that this is the limiting factor to primary productivity.

In the case of salmon farming in marine environments, no formal effort has been conducted to assess the effects of such nutrient contributions to the water column at regional (ecosystemic) scales. Soto and Norambuena (2004) found only weak evidence for increased ammonia in production sites compared to control sites located 1-2 miles away. While those results suggest high dilution/recycling rates in assessed sites, long term monitoring programs and modelling efforts aimed to estimate carrying capacity are, probably, the most urgent research needs for this sector. Conversely, the same study showed significant losses in biodiversity below cages with a much localized effect.

Farming activities have faced different reactions when using freshwaters because of the more competing demands, and also due to higher public concerns about water quality. In search of decision making tools for potential expansion of salmon farming in Southern Chile lakes, the Chilean Fisheries Research Fund (FIP) has supported studies to evaluate trophic status and carrying capacity. These studies have mostly used Vollenweider's model for P proposed by Dillon and Rigler (1974) with a modification for the estimate of Nitrogen critical carrying capacity proposed by Jorgensen and Vollenweider (1989). The major assumption here is that loads which achieve this carrying capacity will trigger eutrophication and, therefore, they can be used to assess environmental effects in a more general way.

Based on these studies a comprehensive review was done by Soto (2000) focusing on P carrying capacity and estimated loads. We have done a similar exercise here with N, using the same information sources (FIP) plus additional data for Lake Llanquihue (Soto, 1993; 2000). In most of these lakes with salmon farming the critical Nitrogen load had not been achieved at the time of the studies (1993 to 2000). Also the proportional relevance of salmon farming is variable with higher effects on smaller lakes, most of them in Chiloe island, while in large lakes (more than 20 thousand ha) their effect is comparatively lower in relation to other load sources. Unfortunately, although these studies give specific loads for each land use in the basin, there is not enough information on livestock densities associated to river or lakes basins and therefore is not possible to identify clearly livestock inputs. However, as mentioned earlier the largest cattle production density is in the Llanquihue province and therefore is likely that a large proportion of "other sources" of N load comes from livestock in the larger lakes depicted in Figure 1. Although carrying capacity estimates can provide indications of farming effects (as contributing to filling of this carrying capacity) it is not clear that ecosystems will respond in negative ways to nutrient inputs, specially when dealing with oligotrophic systems and when society may require higher productivity for

FIGURE 1
Relative nitrogen contribution (N load) and current state (1993-1999) of Southern Chile[9] lakes regarding N carrying capacity. The 100% line represents the estimated critical N load in a carrying capacity model. Contribution of salmon farming in black, while white bars represent all other sources of N



example for recreational fishing. Also, these models are built under the assumption that one or other nutrient, or both, are limiting primary production which is often not the case. In the large lakes from Southern Chile for example, it has been proposed that productivity and biomass could be more regulated by thermal cycling and lake mixing (Soto, 2002). Therefore, we may be attempting to use the wrong impacting force (e.g., N loads) to make comparisons.

Nevertheless, this exercise using simple models for estimating carrying capacity allows for a comparison of relative inputs from different activities and could eventually be used to calculate environmental costs to society, provided that adequate information is available. Such information can contribute to regulate impacts and its effects, tools such as "load quotas/permits" can be implemented and different production sectors, such as cattle and salmon farming, can be equally evaluated and regulated according to societal decisions.

In the reported case from Southern Chile, the information from salmon farming is much more objective since salmon production per farm is well known and periodically updated according to different norms and regulations; this is mostly due to the fact that more than 99 percent of the production is for exports. While livestock production, location and management systems are less known, especially the latter. Milk and meat production are essentially for the country's internal consumption and are less regulated compared to salmon.

For all salmon farming activities which started after 1994 an environmental impact assessment (EIA) is mandatory in addition to the application of the aquaculture environmental regulation (RAMA),[10] but such is not the case and there are not

[9] Lakes included: Popetan, San Antonio, Auquilda, Los Palos, Tarahuin, Natri, Tepuhueico, Riesco, Chapo, Rupanco, all of them available as FIP Reports (www.fip.cl) while Lake Llanquihue data comes from Soto (1993).

[10] Reglamento Ambiental para la Acuicultura (Subsecretaria de Pesca, Chile, www.subpesca.cl). Also see Leon (2006).

equivalent requirements for livestock and dairy production, except when there are superficial effluents.

Other relevant considerations

The selection and use of environmental impact assessment methodologies to compare cattle and salmon production systems should first of all have a clear purpose. This is obvious in the lakes example offered above (Figure 1). In practical terms, for farmers and also for local authorities is relevant to define physical boundaries for ecosystem effects and also being able to evaluate these. The above example with lakes may seem easier to perform and to implement its results as compared to broader scale approaches, e.g., considering whole regions, countries, commodities, etc. In the latter cases it may be more difficult to establish the purpose of the comparisons.

Clearly, there are benefits and drawbacks of some methodologies for the local decision making. When the objective is avoiding eutrophication the use of some mass balance models as the one shown for N in the lakes could allow, for example, to tax nutrient inputs or even regulate maximum inputs considering social and economic benefits of the activity, which is the contribution to local economy, generation of jobs, etc. Such approach could also be possible considering the whole Lakes Region in Southern Chile. In this regard, some of the following considerations are useful; salmon farming is the activity providing most employment and generating ten times more jobs than the dairy industry in the region, while salaries are 40 percent higher than the country average for workers of farms (SALMONCHILE, 2007). In the Lakes Region salmon farming offers 11 percent of total employment with more than 35 000 job posts, creating an economic growth which generates many more indirect jobs (Niklischeck, Soto and Lafon 2006; Leon, 2006). On the other hand, agriculture and forestry represents 11 percent of the hand labour regionally (INE, 2006)[11] but with much lower salaries.

CONCLUSIONS

In southern Chile both salmon and livestock farming are competing for attention regarding policy making, where salmon farming has been attracting more attention for some of the reasons stated above but livestock production have had more support and subsidies in the past. A simple comparison of N loads of both activities at the regional scale showed a larger impact from salmon farming than from cattle farming although at local levels the latter was in some cases greater (e.g., Lake Llanquihue). Therefore, it is very important to define the spatial scale of the approach/comparison according to the impacts, effects, and mitigation possibilities, as it could be the case when dealing with lake eutrophication. Although N surplus and loads were identified as impacts, there was insufficient evidence on the magnitude and type of effects, except for the information on critical nitrogen loads or critical carrying capacity which can be used as a surrogate for environmental costs associated to eutrophication. Indeed, excess Nitrogen exports to environment has been one of the most cited causes of eutrophication in Europe and the possibility of introducing taxes over N loading is been discussed (Vatn *et al*., 2002).

The future development of the region should include considerations of comparative environmental costs along with social benefits of farming activities and therefore at this scale comparisons should include other impacts such as soil erosion from cattle farming and escapes of farmed salmon, which are regionally relevant. As mentioned earlier, more global tools such as LCA or MFA could be very useful at regional and at national levels, especially if there are ways to include the latter two types of impacts and also costs related to losses of biodiversity.

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