International Capital vs. Local Population: The Environmental Conflict of the Tambogrande Mining Project, Peru.

Draft

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Abstract: An environmental conflict between a transnational mining corporation and a rural population in Northern Peru was studied in depth. Interviews with the stakeholders and a survey of the population were carried out in order to characterise the conflict about a gold mining project, in terms of perception of environmental risks, preferred decision-making mechanism, alternative solutions, confidence in experts and government institutions, and fairness in the distribution of profits. Against the post-materialist and "risk society" theses that poor people do not worry as much about environmental hazards as the rich do, the results reveal that 85% per cent of the local population is against the project, due to environmental hazards and other concerns. Local experts were divided about the possible developmental and environment outputs of the projected mine. Disagreement between experts, the local population and the mining company, distrust in institutions, and the lack of a participatory procedure for deciding local development strategies create serious difficulties in legitimising the project assessment and approval. It seems that a "technical" solution is not possible, due to the existence of conflicting value-systems and the fact that no "value free" procedure is available. We arrive to the conclusion that, in this case, the final decision can be legitimised only if the adopted decision mechanism represents a shared meta value-principle, i.e. direct democracy. Thus, a referendum seems to be the most suitable method for deciding on the project. The definition of the most appropriate geographical scale for the voting process relies on ethical stances.

Keywords: Environmental conflicts; Governance; Environmentalism of the poor; Peru, Transnational corporations, Gold Mining.

1. Introduction

Economic growth in the industrialised countries has been accompanied by swelling energy and material consumption. Due to the existence of international trade and transboundary forms of pollution, the different stages of a material chain - extraction, processing, consumption and final disposition - and their environmental impacts can be geographically separated many thousands of kilometres. Land transformations and pollution associated with material use not only affect the functioning of natural ecosystems, but also human health and social relations. Thus, equity issues may arise as regard the spatial and temporal distribution of environmental hazards or impacts and economic benefits of international commodity-material chains. Local environmental conflicts are the consequence of disagreements between different groups within society about alternative resource uses or the allocation of environmental hazards or actual environmental impacts. In this article, equity considerations are not introduced as a charitable afterthought, but on the contrary they are seen as inextricably linked to the analysis of geographical, environmental, economic and social allocation decisions.

Mining activities, being among the most environment-intensive sectors, have generated environmental conflicts all around the world (Martinez-Alier, 2001). Mining and oil enterprises facilities face increasing opposition by local movements both in the North and the South. Mounting difficulties for exploiting public lands, decreasing prices and rising environmental concerns seem to have played an important role in “pushing back” mining activities from the industrialised world. A migration of metal production from developed to developing areas has occurred in the last decades and aggregated Northern imports of semi-manufactured metals from developing areas have witnessed a boost in the same period (Muradian and Martinez-Alier, 2001). These material flows can be conceived as North-to-South displacement of the environmental hazards associated with material extraction and processing. According to Clapp (1998), the differential in environmental costs between the operation of multinational corporations in the North and the South seems to be widening, which contributes to a North-to-South hazardous industry relocation.

Pollution from mining can be controlled by technology. However, the total amount of effluents can be reduced only if material removal diminishes. Instead, most of the time, water or air pollution is reduced by "storing" pollutants in special places. When natural environmental variations or human errors produce an involuntary spill, major ecological and health disasters may occur. Hence, akin to nuclear plants, mines are environmentally risky locations. Like landfills and incinerators, they belong to the category of economic activities commonly called "locally unwanted land uses" (LULUs). This kind of land use typically generates opposition between the "national interest" and local populations.

The concept of commodity frontier (Moore, 2000) can be useful for studying the geographic space of international commodity chains. The expansion of the commodity frontier of metals seems to be determined by the distribution and quality of actual mineral stocks, the cost of production, as well as by social and institutional aspects, like local opposition or government attitude towards mining. The migration of resource extraction towards peripheral areas may be explained by social cost-benefit considerations. Being a hazardous activity, neo-classical economic theory advises to place it in sites where the costs of ecosystem-destructive or health-impairing pollution be the lowest. Since the economic value of ecosystem functions and human health (measured through WTP or the actual market value of labour or environmental services) depends on income, environmentally risky facilities should be located in the poorest available localities. From the point of view of a cost-saving transnational mining enterprise, locational decisions should be guided -apart from the traditional factors like production costs, political stability and proximity to the final market- by the minimisation of transaction costs and compensation costs in case of accident. In peripheral areas, development decisions are usually steered by short-term income-rise considerations, regardless of long-term environmental hazards or burdens. Government are pressed by large debt-burdens and high budget deficits and unemployment rates, to undertake the privatisation of state-owned enterprises and to facilitate the entrance of any kind of foreign capital. Besides, poor and powerless people can be cheaply compensated (in comparison with more wealthy and empowered communities). Hence, at the micro-economic level is also "desirable" to site mines in peripheral regions. Given that in a globalised world economy, international capital can migrate in search of lower production costs, some authors postulate that globalisation allows businesses to have an increased capacity to externalise costs on local communities (Glover, 1999), and to displace across space and time environmental problems (Wapner, 1997). It is also argued that minorities, poor and powerless social sectors tend to bear a disproportionate burden of environmental hazards and externalities at a national and international level (Bryant, 1997; Barbosa, 1996; Adeola, 2000; Dawson, 2000; Pellow, 2000).

Despite a long history of local environmental conflicts and several cases of heavy pollution and local people intoxication due to accidents or mismanaged private or state mining enterprises (for example, the cases of La Oroya, Cerro de Pasco, Hilo, Huarmey, Choropampa, Tintaya, etc.), the Peruvian governments have adopted a strategy of economic growth based on the expansion of the mining sector. Mining production and exports have experienced a boost in the 1990's. In 1999, the mining exports constituted 52 % of total exports and the mining-oil sector represented 12 % of GDP (Ministry of Energy and Mines, 2000a). As can be seen in Figure 1, during the 1990's there was a general trend towards decreasing value/weight ratios for the most important metal exports in Peru. Due to prices decline because of oversupply, countries specialised in the mining sector are forced to export ever increasing quantities of materials in order to maintain revenues. This is striking in the case of Peru, where gold exports passed from 25.000 ounces in 1990 to 4.228.000 ounces in 1999. In the same period, tin exports mounted from 4.000 to 28.000 metric tons (Ministry of Energy and Mines, 2000b). Figure 2 shows that this trend towards increasing volume of exports is common also to copper, zinc, lead and silver. Enlarging outflow of materials implies escalating affluent production and environmental hazards. Equally, in order to broaden extraction, an expansion of the commodity frontier has to occur, increasing the probability of environmental conflicts.

It is often stated that increasing environmental awareness and local environmental movements resisting LULUs are the expression of a new value-system in modern industrialised societies, which assigns greater importance to “quality of life” issues and environmental risks avoidance then to income growth. This “postmaterialist” thesis assumes that poor people is too constrained by survival necessities to worry about environmental matters, which are seen as a luxury. According to this perspective, when basic necessities (food, shelter, cloth) are satisfied people start to be concerned by "non-material" topics, like the environment (Inglehart, 1981, 1990). Using the same arguments, the so-called economic contingency hypothesis postulates a positive relationship between income and environmental concerns (Jones and Dunlap, 1992, Albrecht, 1994). The paradigm of *risk society*, which has had a great leverage in current environmental sociology, also stands on the idea that burgeoning concerns for environmental risks is basically a phenomenon restricted to societies in a "late stage of modernity" (Beck, 1996). Beck (1992) thinks that "a change from the logic of wealth distribution in a society of scarcity to the logic of risk distribution in late modernity...occurs where and to the extent that genuine material need can be objectively reduced". Industrial society, a previous state of modernity where these material needs are not completely fulfilled, is characterised by a consensus for progress and the abstraction of ecological affects and hazards. The basic conflicts of this period deal with the distribution of "goods" (income, jobs, etc.). The risk society designates a "stage of modernity in which the threats produced so far on the path on industrial society begin to predominate". In this reflexive stage, conflicts over the distribution of "bads" (Pollution, risks, etc.) tend to prevail (Beck, 1994). There are similarities between this rather linear approach for understanding the evolution of society-nature relations and that proposed by Nash to explain increasing social value for wilderness in the US. Nash (1982) proposed that whereas the marginal value of "civilisation" tends to decrease across time, the marginal value of nature tends to increase. Consequently, in the latest stages of development there is a "nature appreciation" process. This vision is shared by some economists, who state that the income elasticity of demand for environmental amenities is larger than one. This means that when people become richer, the demand for environmental quality increases more than the level of income. This has been proposed as one of the mechanisms underlying the environmental Kuznets curve (Barbier, 1997). Environmental movements leading current environmental protests are often classified as New Social Movements (NSM). NSM are conceived as a new form of organisation, different from social movements of the industrial age (focused on workers and class struggles), and as a product of the shift to a postindustrial economy. It is argued that the demands of NSM moved away from the distributional issues of industrialism to the quality of life concerns of a postmaterial age (Pichardo, 1997).

Such views cope with serious difficulties when trying to explain the emergence of local resistance environmental movements in poor and peripheral areas, in developed countries or at a global scale. This approach fails to explain why poor people are permeable to so-called postmaterialist values, when they are confronting more urgent problems. An alternative perspective on the issue postulates that as environmental problems become more obvious and ubiquitous, awareness tend to spread across any sector of society, independently of their socio-economic characteristics. This has been called the broadening base hypothesis (Jones and Dunlap, 1992). This idea has been supported by some empirical findings showing that environmentalism has become a world-wide phenomenon (Adeola, 1998). Some authors adopt a third viewpoint on the evolution of society-environment relationship and the nature of environmental protests movements in peripheral communities. Norgaard (1994, 2001) for example, claims that society-environment relations do not evolve in a linear and predictable way, but rather co-evolve in a complex and culturally bounded manner. Therefore, the meaning of concepts like “environment” changes importantly through time and space (Redclift and Woodgate, 1994). Concerning environmental movements, Taylor (2000) conceives the history of the environmental justice movements in the US (fighting the siting of hazardous facilities in people of color neighbourhoods) as independent from the evolution of male-white-dominated, middle-class, and wilderness-oriented mainstream environmentalism. Whereas the origins of the mainstream environmentalism are in the romantic worship for wilderness, the dawn of the environmental justice movement is traced back to the long history of social justice and civil rights struggles in Afro-American communities. In the same line of reasoning, some authors state that the “subaltern environmental struggles”, also called “environmentalism of the poor”, are founded on different values from western mainstream environmentalisms (Guha, 1991; Pulido, 1996, Guha and Martinez-Alier, 1997). It is argued that for many marginal groups, the environment is a matter of livelihood, not of enhancing the quality of life. According to this approach, current environmental struggles are linked historically to the long history of oppressed groups resistance to the forces that seek to undermine them (Fuentes and Frank, 1989). Therefore, they are not qualitatively different from the distributional movements of the “industrial age”. Whilst “sustainability” or “conservation” are key words for the mainstream Western environmentalism, “justice” or “rights” are instead key words for peripheral environmental movements. From this point of view, environmental movements in the periphery are not the ouput of spreading environmental values from the core, but rather a new representation of old ways of resistance.

This paper analyses some social aspects of an environmental conflict between a transnational mining corporation and a rural population in Northern Peru. The next section describes the area of study and the scope of the project. Later, the conflict is schematised in terms of stakeholders’ perceptions and languages used. This is followed by a discussion on, *inter alia*, the nature of “peripheral” environmental movements and the role of experts and science in environmental conflicts.

2. The Tambogrande project

Tambo Grande is a town of around 16.000 inhabitants (mainly mestizos). It is the capital of the Tambo Grande district and the most important urban settlement of the San Lorenzo Valley, a farming area of roughly 60.000 ha. Tambo Grande belongs to the province and department of Piura. Piura is also the name of the department’s capital city, which is about 75 kilometres from the town of Tambo Grande. Tambo Grande is 100 kilometres away from the Paita seaport. The San Lorenzo valley belongs to the same department, but its area is shared among the provinces of Piura and Las Lomas. Current cultivated area is the result of a successful colonisation and irrigation program started in the 1950s, which was financed mostly by the World Bank and the American co-operation agency USAID. The valley is specialised in the production of fruits, particularly mango and lemon. Local factories elaborate part of the mango production for exports. The Tambo Grande district is essentially a rural area, characterised by low income and a lack of services. In 1993, date of the last national census in Peru, adult illiteracy in the Tambo Grande district was 25.5 %, higher than the national and departmental rates. In the same year, only 16 % of the population had access to potable water, 66 % did not have hygienic service and 87 % lacked a connection to electric national network. The Tambo Grande district was classified as *poor* in a recent government evaluation of districts conditions at a national level (FONCODES, 2000).

Like almost all the Peruvian coast, the Tambo Grande district is very arid. Agriculture was possible due to dams and irrigation canals. Original vegetation is classified as a dry tropical forest, dominated by tree species of the genus *Prosopis* (Algarrobo). This kind of vegetal cover is very fragile to human perturbations. Consequently, desertification is likely to occur. The area is also characterised by recurrent periods of extremely high precipitation, called El Niño, produced by appearance of atypical, hot and superficial waters in the South Pacific. The late twentieth century witnessed two extreme El Niño phenomena, one in 1983 and the other in 1998. They produced considerably economic costs (and some benefits) in the department of Piura, and altered significantly the landscape of the region.

The Tambogrande project comprises 97 mining concessions and has an extension of approximately 87.000 ha, which are mainly situated in the Tambo Grande district. A concession is a land area that is transferred by the state to an enterprise for exploration. The company has the option to exploit it, after accomplishing legal requirements. The project has been divided in three sub-projects (depending on the geographical distribution of concessions): Tambo Grande, Lancones and Papayo. Works on exploration and planning are concentrated on the Tambo Grande area, having nearly 10.000 ha. These concessions belong to Mineru Peru, a state mining enterprise, which has granted the right of exploration to Manhattan Minerals Corp., a relatively small Canadian transnational corporation registered in the Toronto stock market. Tambogrande is the only project of this company. Manhattan has some years for carrying out a feasibility study, as well as an environmental impact assessment. If the project is approved, Mineru Peru will transfer to this company 75 % of the project's property. Thus, the state will be a shareholder of the project with 25 % of participation on benefits.

There are three major mineral deposits where mining is envisioned, called TG1, TG3 and B5. For the moment, the project is restricted to study the feasibility of exploitation in TG1, which coincides with 1/3 of the area of the Tambo Grande town. TG1 comprises two distinct deposits, the most superficial is an oxide gold/silver sulphide deposit, hosting 8.75 million tonnes of mineralisation grading 3.7 g/t gold (1.0 million contained ounces) and 70 g/t silver (20 million contained ounces). The other is a copper/zinc/gold/silver massive sulphide deposit, hosting 56 million tonnes of mineralisation grading 1.6% copper, 1.1% zinc, 0.6 g/t gold and 28 g/t silver. During the first years of exploitation, TG1 is planned to be a gold and silver mine. After, copper would be the main mineral extracted. TG1 mining would last about 10 years. Because the deposit is rather superficial, mining would be made through an open pit. The pit would have an area of 27 ha and a depth of 250 meters. An important part of the town has to be relocated and the riverbeds of the Piura river (the larger in the district) and the Carmelo stream have to be canalised and moved away. The company plans to build 16.000 new houses in 20 ha to lodge the displaced people. Gold will be extracted using the method of lixiviation in tanks, which involves the use of cyanide to separate gold from non-useful materials. Wastes from the concentration process would be accumulated in a "tailings field" (cancha de relave) and sterile material would be stored in an "overburden landfill" (cancha de desmonte) of approximately 90 ha. The project plans to create 600 direct jobs, and from 4 to 5 indirect jobs by each direct employee. According to the last projections we got, the mining and concentration processes at the maximum level of production (20.000 tons/day) will use 551 liters/second of water. 485 L/s are expected to be re-circulated. Therefore, it would need a net water input of 66 L/s. 44 L/s would be obtained from subterranean water emerging from the pit and the rest would be taken from the Piura river.

Operations of Mahattan in the Tambo Grande district started in 1996. The company states that a community-relations program has accompanied the feasibility and environmental baseline studies. It included a house-by-house information campaign and public information meetings in Tambo Grande, as well as agriculture professional advise to farmers in the rural area. In the village Apostol San Juan Bautista de Locuto (where the B5 deposit is situated and one of the two possible sites to locate the overburden landfill), Manhattan reached an agreement with local dwellers. Local drilling for exploration was allowed in exchange for the construction of chapels, classrooms, medical dispensaries, and a program of alphabetisation and technical-skills training. The project explicitly plans to implement social programs for ameliorating local education and health conditions. Moreover, agriculture diversification has been proposed as one of the project's objectives. Currently, the company is dedicated to elaborate the environmental impact assessment for mining the TG1 deposit.

The Peruvian constitution does not allow foreigners to own mines, lands, forests, water or energy sources in areas located less than 50 km from the frontier, unless the activity is declared as "public necessity" through a supreme decree approved by the president and the ministers council. As the Tambo Grande project is close to the Ecuadorian frontier, the ministers' council declared, in May 1999, "public necessity" private inversions in mining activities, in order to allow Manhattan Minerals Corp. to operate in the region. In November 1999, the municipality of Tambo Grande emitted a decree permitting Manhattan to undertake exploration studies inside the town of Tambo Grande. In March 2001, the municipality, pressed by the local population, annulled this decree and request the national government to reconsider national laws about mining exploitation in urban areas.

The Peruvian state has a trans-sector organisation to deal with environmental issues. General environmental policies and guidelines are given by the CONAM (National Environmental Council), but there is no a ministry of the environment or other organism centralising environmental decisions or in charge of environmental auditing. In the case of mining, environmental policies are decided and assessed by the Ministry of Energy and Mines. This ministry is also the organism in charge of evaluating environmental impacts assessments (EIA) presented by mining companies. Private companies, hired by mining enterprises, make the EIA. Private companies also audit the environmental performance of mining activities. The mining company whose environmental performance is to be evaluated pays these enterprises, and the results are reported to the Ministry of Energy and Mines. The EIA is the most important step mining enterprises have to pass in order to get the final approval from the state. Some months after the submission of the EIA to the Ministry of Energy and Mines, a public audience has to be done. In this audience, any person has the right to express criticisms, comments and suggestions on the study. Taking into account this feedback, the ministry arrives to a final decision as regard project's feasibility. This is the only participatory step in the decision-making process about mining projects.

3. The conflict

In the town of Tambo Grande, a grassroots organisation has opposed the mining project and the community-relations strategy of the enterprise. It is called the "Frente de Defensa de Tambo Grande y el Valle de San Lorenzo" (hereafter Front of Defence). It is constituted mainly by farmers and peasants. Members of the directing council were elected in a public assembly and the president is the director of one of the local schools. The Front of Defence organises local protests, as well as manifestations in the near city of Piura. The conflict witnessed a violent turning point on February 27 and 28, 2001, when a massive demostration took place in the town. These days, some 5,000 local residents stormed the company's premises, burning machinery and destroying scale-models of houses for future relocated people that Manhattan has built as part of the community-relations program. The second violent twist to the conflict occurred on March 31, 2001, when Godofredo Gracía Baca - a local farmer, agronomist and one of the main leaders opposing the project – was shot dead by a hooded gunman, who ambushed he and his son as they drove to work. On May 2001, a forum for dialogue was established between the company, the Ministry of Energy and Mines, the Archbishop of the Diocese of Piura and the Front of Defence. Dialogue did not las much, because the Front pointed out that a compromise solution was not possible. They opposed any mining involving dismantling part of Tambo Grande. Later, on July 2001 the clergy of the Archdiocese of Piura published a manifest against the mining project. Thus, the Archbishop could not play any more a neutral role. Public opposition of the Archdiocese to the project was an important point in the evolution of the conflict because the Catholic Church has a significant moral influence on the population. In October 2001, dialogue was re-launched, mediated by the Defensoría del Pueblo (a Peruvian Ombudsman group funded by and reporting to congress). At the moment of writing, no agreement has been reached.

Debate about the project not only happened in Tambo Grande, but also in Piura city. The Front of Defence has received support mainly from NGO's from Piura, whereas the mining project is defended mostly by the entrepreneurial sector (represented by the Commerce Council of Piura). Some professionals from Piura, including lawyers, economists, engineers and biologists, have constituted a voluntary and non-formal association (working group) for elaborating technical arguments against the project. It was named Piura Life and Agriculture (Piura Vida y Agro). Due to the polarisation of positions, the occurrence of violent events and the participation of multiple and influencing stakeholder, the conflict has taken a dramatic significance in the region. It also attracted the attention of the international press (see for example The Economist Jun 23rd 2001). Opposing stakeholders have received directly or indirectly financial and logistic support from international NGO's, like Oxfam America, the Mineral Policy Center, the Environmental Mining Council of British Columbia, and Friends of the Earth Costa Rica.

In order to make a conflict analysis in terms of stakeholders' perception of environmental risks, confidence in institutions, preferred decision-making mechanism, etc., we carried out a population survey in Tambo Grande, as well as structured interviews to the main actors involved in Tambo Grande and Piura city. Information was also gathered in several debates with technicians and stakeholders organised by the University of Piura. Other sources of information were local newspapers and public manifests. The interviews were undertaken between July and August 2001. The interviews were made with a representative spokesperson for the each organisation. The population survey was carried out (from August 2 to 5) in the rural area of the San Lorenzo Valley and the urban area of Tambo Grande. 18 students from the National University of Piura surveyed people. The sample was chosen randomly, from a list of households of farmers in the case of rural surveys, and asking systematically passing pedestrians in 18 randomly chosen points in the town, for urban surveys. Table 1 summarises background characteristics of the sample. Males are predominant in the sample, probably because there is a higher probability to find a man as a pedestrian. Women are predominantly housewives. Men also are more prone to be respondents when doing household surveys in the rural area. The exchange rate at the time was approximately 3.45 soles per dollar.

Table 2 reviews results of the survey. A large percentage of the sample was against the project (85 %). Opposition is reduced to 76 % when local population is proposed as shareholder of the project. Among women, the percentage of opposition was very similar (86%, result not published in the table). The majority of the population prefer a referendum at the level of San Lorenzo Valley to decide whether the project should be developed. Most of the people believe that mining and agriculture cannot coexist in the area, feel that more information about the project is needed, are not willing to be relocated (even if monetary compensation is made), think pollution would be high or very high (72.5 % of women also believe that pollution would be high or very high), don't trust in Manhattan's experts and declare that the main beneficiary of the project would be the company. The fact that people acknowledge a lack of information may imply that current positions and perceptions can change when information be provided. As regard confidence in the government for assessing and auditing the project, the population is divided, although a considerable proportion (about 40 %) does not trust in government institutions. This result is probably explained by the democratic transition Peru was living at the time. Due to corruption scandals, the president A. Fujimori left the country after a long period of autocratic practice. A transient government was established and a new president (A. Toledo) and congress were elected, which took the power in July 2001. The new government raised optimistic expectations in the population in relation to governing rules and ethics. The population is also divided as regard the company's involvement in the murder of García Baca. A large proportion does not have a position towards this delicate theme. This question was formulated because people's perception on the company's ethical behaviour is crucial for mutual relations.

Table 3 schematises the stakeholders' position towards different relevant issues in the conflict. Numbers in Table 4 resulted when a spokesperson of each institution was asked to rank macro productive alternatives according to the organisations' preferences, using a scale from 1 (not desirable) to 5 (very desirable). Alternatives were suggested by the stakeholders. Table 5 results when stakeholders were asked to assign a number to express their subjective perception about the economic feasibility of the same alternatives, given current institutions and available economic resources. They also used a numerical scale from 1 (hardly feasible) to 5 (very feasible). The difference between both tables can be described as the psychological difference between desirability and feasibility. Mining and business as usual were assumed as very feasible by definition. Note that alternatives are not necessarily excluding. This exercise was thought as a general tool to characterise stakeholders' position towards alternative allocation of resources in productive activities. The municipality of Tambo

Grande does not appear in Table 4 and 5 because the major refused to evaluate alternatives. Aider, Centro Ideas, Pidecafe and Cipca are NGO's from Piura city, dedicated mainly to rural development issues. Aider works on productive activities for the sustainable management of the dry forest. The Centro Ideas has worked, among others, on citizenship, watershed management and agroecology. Pidecafe supports peasants for organic coffee production and integration in fair-trade networks. Cipca is the larger NGO's in the region. It is devoted to research in social science and to implement development programs in rural areas. Diaconia is the office of public relations of the Archdiocese of Piura.

As can be seen in Table 3, there are three kinds of general positions toward the project: in favor, against and "non-aligned". The latter category arises because, according to the stakeholder, not enough information is available to take a final position. Nonetheless, this can be also a "strategic" answer, to avoid the consequences of taking a stance on the conflict. All the stakeholders against the project share some common perceptions. They prefer the referendum as decision-making mechanism, they think that the voting process should be done at a very local level (S.L. Valley or district), they also state that mining and agriculture are not compatible in the region and the probability of ecological disaster is high or very high. They share low confidence in governmental institutions, Manhattan's experts and the enterprise doing the EIA. As can be noted, the position of these stakeholders is very close to that of the majority of the population. On the other hand, stakeholders in favor of the project (Manhattan and the Commerce Council) trust in governmental institutions and the enterprise doing the EIA. They also share a preference towards EIA, though Manhattan does not oppose the referendum as decision-making mechanism. However, it differs substantially with opposing groups, preferring the department as the most suitable scale to undertake the voting process. According to the Commerce Council, experts in the private and public sectors should assess critical issues in the project, as environmental hazards. Its general position is that if experts determine that the mine will not harm the environment, it should be built. The NGO's "not aligned" only share indeterminacy as regard compatibility between agriculture and mining. For the rest of subjects analysed, they have different perceptions. Cipca have the same perception as "opposing" stakeholders, except for the compatibility between agriculture and mining.

Stakeholders' perceptions regarding productive alternatives were stated in order to envisage positive solutions to the conflict. That is, the current environmental conflict can be useful to foresee alternative economic activities, to be developed in the area in order to increase local income. A general participatory development plan can be one of the outputs of the conflict. We deal here only with the economic aspects of this possible plan. The alternatives can be seen as very general choices to invest local scarce resources. As regards stakeholders' preferences, Table 4 shows that almost all agree that the current situation is not very much desirable. Mining activities to resolve existing problems is very desirable only for Manhattan and the Commerce Council. Alternatives where a high degree of convergence is found are: agroindustry intensification and present agriculture intensification. This distinction is made because not all the current production is dedicated to exports. There are limits imposed by the capacity of the processing plants and the quality of the production. For these activities, there is also a high degree of convergence on the subject of subjective economic feasibility, except for Cipca. This organisation is rather pessimistic about the productive alternatives actually available. Cipca also considers current situation as not desirable. This combination of perceptions likely explains why Cipca does not hold a clear position on the conflict. Probably, the argument handled by this institution is that mining, though environmentally harmful, is the only option the region really has for income increase. Cipca and Aider position can be also explained by their role as "third party consultants", as were called by Manhattan. These NGO's were hired to implement some parts of the consultation process and community relations program developed by Manhattan in the region, with aid of "independent" organisations.

3. Languages of the conflict

The vision of Manhattan and the Commerce Council on project assessment can be defined as sheer technical. For these groups, the EIA will allow taking into account local environmental conditions in designing the mine. Using updated technologies, environmental risks will be minimised. According to this view, any engineered project has an implicit risk. The task of engineers is to minimise these risks, through technology and a good design. In this sense, any accident would be an “act of god”, because all precautions would be taken. Engineers assure that El Niño is included into the project design. For example, the tailings field is planned to withstand precipitation levels equivalent to the most dramatic El Niño experienced in the past. According to the public relations department of Manhattan, there exists a contingent plan is case of ecological disaster and monetary compensation for the affected population is envisaged. The underlying assumption of this approach is that environmental impacts can be economically compensated. This stance, shared by many environmental economists, rests on the idea that, since pollution cannot be indefinitely reduced, populations bearing pollution burdens must be compensated through Pigouvian taxes or direct Coasian negotiations in order to guarantee a socially “optimal level of pollution” (Munasinghe,1993; Boerner and Lambert, 1995).

In this case, economic compensation is planned through an agreement between the corporation and the local population. This agreement and all the social aspects of the project are managed by a public relations department. The company has a “paternalistic” approach on this issue, in the sense that it plans to surrogate state’s functions in the region, like education or health assistance. For Manhattan, coexistence between agriculture and mining and co-operation with local farmers is desirable and totally feasible. In fact, agriculture diversification is proposed as an explicit objective of the project. For the enterprise, the “social cost” of the project is the monetary value of the community-relations program and the cost of building a “new” town to relocate people, coinciding with the World Bank statement that the cost of involuntary resettlement should be calculated as the replacement cost. Local resistance is conceived by the company as the result of a stigmatisation of mining activities due to past mismanagement, a local lack of information and people manipulation by regional and international groups with “political” interests. The company expresses publicly its commitment with democracy and consultation. In its webpage, it points out that the consultation process for people relocation “stopped when a group of politically motivated people attacked and damaged the Manhattan office facilities in Tambo Grande”. This document also states “Manhattan is committed to applying the principles of sustainable development”. For an important manager of the company, some of opposing groups are interested in exacerbating the conflict as a way of getting foreign resources. Thus, mistrust is mutual. Summarising, for the enterprise, social sectors are intending to make the project a political issue, when it is basically a technical matter.

The opposing NGO's and the work group Piura Life and Agriculture share a common language on the conflict, which is dominated by an environmental-hazards-averting and local-democracy discourse. Technical counter-arguments are given to emphasise the environmental risks of the project. The main are: a) Deforestation for building the open pit may cause local temperature changes, modifying wind patterns, which may produce dust and polluting spread; b) Water withdrawal from the pit and possible pollution (acidification) may change the quantity and quality of subterranean water flows, affects surrounding dry forests ecosystem and plantations. As the dry forest ecosystem is very sensitive to human perturbation and local weather conditions, desertification is likely in case of water lack or pollution. The magnitude and occurrence of El Niño phenomenon is basically unpredictable. The event of 1983 was thought to be the strongest of this century. However, some years later, in 1998, el Niño was even more destructive. Moreover, it is pointed out that the Tambo Grande district is one of the places with lowest no-Niño/Niño precipitation ratios in the department of Piura and the Peruvian coast. It was around 1/30 in 1983 (Rodriguez et al., 1994). Additionally, extraordinary one-event rains may occur is this area during El Niño. c) Historically, the riverbed of the Piura river has been very mobile. Therefore, canalisation of it (as the project plans) would be very risky, especially because this river is extremely affected by El Niño. For example, its volume increased as much as sixteen times in 1998 (van der Veen, 1999). For the opposing groups, all the above-mentioned factors make environmental hazards too high to be "acceptable". On the other hand, these groups claim that local development decisions have to be taken in a participatory way. After an almost dictatorial regime, there is a general plea for democratisation and decentralisation in Peru. Furthermore, they assert that no EIA in the history of Peru has been rejected (therefore it is only a bureaucratic formality) and that the centralised decision-making procedure is very permeable to corruption and dominated by a reduced group of influencing people. Many opposing stakeholders mentioned that the president of the company elaborating the EIA in Tambo Grande is also the representative of the mining sector in the National Environmental Council (CONAM). The Front of Defence' discourse is based on the idea that the local population has the right to decide about its future. They consider unfair the imposition of activities that are potentially harmful for local sources of income and human health, would disturb local social organisation, won't create much local employment and would favour principally to foreigners and the (traditionally corrupt) central government.

In a letter dealing with the Tambo Grande conflict (that became unintentionally public) the minister of energy and mine of the transitory government wrote to the general secretary for the Flemish North-South co-operation that "the presence of certain NGO's in the area of influence of mining enterprises may be an unnecessary perturbation factor, hampering communities-company relationships". Of course, this letter raised strong reactions from the opposing groups. In general, the official position of the government has been that the conflict has to be resolved by legal ways and that regular procedures have to be followed. However, opposing stakeholders claim that the government has had "hidden" positions and a double discourse, and the previously mentioned letter is cited as an example. The government has difficulties for justifying its position as "judge and part" in the conflict.

4. Discussion

The case here described is an example of a distributional environmental conflicts between international capital and local "peripheral" populations. In search of lower costs, and facilitated by government pressures to attract foreign investment, transnational corporations in environmental hazardous sectors, like mining, are prone to locate facilities in peripheral (poor and powerless) areas of the world economy. Unexpectedly for many, local populations are increasingly opposing export-oriented environmentally and health risky economic activities, despite their possible short-term economic benefits. In our case, local opposition is based on the stance that the economic benefits and environmental burden of the mining project are unfairly distributed between the mining corporation, the central government and the local population. Since local-people's resistance to the project still remains very high (although diminishes) when a participation in the project as shareholder is (virtually) offered, environmental and self-determination concerns seem to overcome "equity" and "income growth" considerations. It is worthy to note that "environment" is this rural locality may have a totally different meaning than for an urban middle-class person. Here, “environmental quality” is rather a matter of survival and a condition for economic revenues (agriculture). This is a different perception from many environmental movements in the US for example, where wilderness takes a crucial significance. A third important aspect of the studied conflict is the notion of “rights” to decide local development strategies. The project is seen as an “imposition”, hindering local empowerment. This point is very important in a period “democratic transition” in Peru, and Latin America in general. Mobilisation around issues defined in terms of rights is very common in environmental conflicts (Cormick, 1992; Bapat, 2000). Another critical issue in the conflict is people’s confidence in experts and institutions. Common distrust is also a characteristic feature of local environmental disputes (Smith and Marquez, 2000). Our analysis suggest that people mistrust in the company and the central government, not only because people’s perception on environmental risks differ from experts, but also because of a long tradition of corruption at any level in the Peruvian society. Corruption was aggravated during the Fujimori’s administration, which blended illegality with authoritarianism. Authoritarian regimes in Latin America have been traditionally guided by delivering economic benefits in exchange for taking away political rights and dismissing environmental problems (Alario, 1992).

The qualitative and subjective analysis of preferences and economic feasibility perceptions toward alternative productive activities reveal that agroindustry and agriculture intensification are seen by almost all the stakeholders as both desirable and feasible. The only exception is Cipca, which is pessimistic about local capital availability. Apparently, social acceptability on the development of these activities can be easily reached. Active planning and promotion of these activities is likely to be successful as a strategy of mining opposition. The current situation is considered by many groups of society as “undesirable”. Therefore, the protest movement would gain if opposing groups complement their grievance with the elaboration of alternative development strategies for the region.

“Right”, “risk”, “trust” and “equity” are key words for local people as regards resistance to the mining project. Apparently, these words have more weight than “economic growth”, “progress” or “modernisation”. Is this a *risk society* or postmaterialist value-system?. Tambo

Grande socio-economic conditions reveal that the region is far away from having resolved urgent material necessities of the "industrial period", like child mortality or malnutrition. In this place, conditions are not given for the emergence of "quality of life" concerns. Therefore, our results are against the posmaterialist thesis. On the other hand, we think that the people's position is neither explainable by an international broadening of environmental values, although it could have played a role. Local grassroots movement is closer to the environmental justice movements in the US than to traditional environmentalism in developed countries. For local people, the environment is not a matter of "quality of life", but just of "life". We argue here that the environmental values of rural dwellers in peripheral areas of the world economy differ from those of "green" middle-class-urban inhabitants in core (industrialised) countries, e.g. traditional environmental movements in the US or green parties in Europe. Local resistance in Tambo Grande can be conceptualised as a civil rights movement dealing with environmental issues, instead than as an environmental movement (at least, as it is conceived in the North). However, both visions are not incompatible. On the contrary, coalitions with Northern environmentalism are likely to arise, and has actually occurred in this case. It must be said that there are varieties of Northern environmentalism. For example, there are significant differences in the approaches of species-conservation-oriented organisations like Nature Conservancy and equity-oriented ones like Oxfam. In the Tambo Grande project there is a strong partnership between the Front of Defence and development NGO's from Piura city. These NGO's are strongly coupled with Northern counterparts. Actually, most of them depend on foreign resources for functioning. Alliances between rural grassroots organisations and urban NGO's are very common in Latin America (González, 1992). These rural-urban-international coalitions tend reinforce the strength of environmental protests (Haynes, 1999; Kousis, 1999). International civil society financial resources have supported the local campaign against the mining project. In this case, we are confronted by a double-faced globalisation process. Paradoxically, this peripheral area is witnessing the influence of two opposing globalisation forces.

As was made clear before, multiple languages are used in the conflict and, not less remarkably, some languages are absent (indigenous territorial rights, endangered endemic species, sacredness, nationalism although not regionalism, environmental racism...). In the

Tambo Grande environmental struggle, two communities of experts interplay adopting similar technical languages, but dissimilar value-systems and having unlike perceptions on environmental risks. The risk-averting approach of opposing experts is close to lay-people standpoint. The value-system of the enterprise's experts is nearer to that of the entrepreneurial class from Piura and the decision-makers at a national level. This division is common in highly uncertain environmental issues, like global warming, nuclear power, GMO's, etc. The distinction between experts and lay people perception of environmental risks has been largely addressed in the environmental sociology literature. Experts tend to deal with risk in terms of probability of the undesirable event and the quantitative magnitude of the consequences (number of dead or sick people, economic costs, etc.), while lay people tend to address risk from a more complex viewpoint, including considerations on qualitative aspects, like the distinction between personally-chosen and "forced" risks (Kasperson, 1992). According to Percival (1992), moral outrage against involuntary exposure to risk, particularly when the exposure is a result of actions that provide economic advantage to others, may explain why the general public's perceptions of the significance of environmental risks appears to differ systematically from those held by the elites. If the gap between those whose make decisions about risks and those affected by risks is large, conflicts are likely to emerge (Ali, 1999). Disparities on risk perception not only occur between experts and lay-people, but also within the expert-community. Different groups in society may compete with one another over the scope and degree of environmental dangers and hazards (Freudenburh, 1997). In siting disputes, it is common that local groups ascribe to radically different assumptions about the kind and degree of danger besetting their community (Cough and Kloll-Smith, 1994). If risk perceptions vary among individuals and groups, a pertinent question is "whose perceptions should be used to make social decisions on risk?" (Renn, 1992).

Most of resources disputes are interest-centred and value-centred conflicts, which are best addressed through a political process rather than from a sheer technical approach (McDonnel, 1988). When uncertainty and the decision's stakes are high, disagreements on the perception of risks are frequent, and typically a "legitimisation" problem takes place. According to Funtowicz and Ravetz (1992), these situations lead to questioning the role of traditional science and experts in the decision-making process. In these cases, the scientific realm cannot be dealt with as an objective "black box" separated from moral and social issues (Wynne, 1996). Funtowicz and Ravetz (1994) propose that, as differing propositions about uncertain events cannot be tested or validated *a priori*, an extension of the peer-community has to be done in order to improve the *quality* of the decision-making process. A change from "optimal" to "quality" decisions is required. This is the aim of a *post-normal* science. That is, a practice intending a democratisation of the traditional role of experts in decisional frameworks, what has been also called a "de-monopolisation" of science (Beck,1994).

Our case seems to be a value-system contest (O'Connor, 1993), where diverse groups within society fight to impose their stance over access to depletable natural resources, and a certain conception of "legitimate" decision-making process. In this (democratic) context, legitimisation is achieved when the rules (algorithms) for decision-making are accepted by the majority, even when the majority could differ with the final output when following the rules. Legitimisation is a precondition for social stability. When a social decision is taken through a legitimate way, people are able to *accept* it, even when they don't *agree* with it. Modern democracies have relied largely on science and experts for legitimising decisions, at least at the project-level. Science is traditionally looked upon as a source of authority (Ozawa, 1996). Power-knowledge relations have been characterised by invisible cultural constructions of the human subject and unspoken social and moral commitments (Goldman and Schurman, 2000). Current environmental problems have raised these hidden assumptions to the public debate. The traditional power-knowledge alliance has been questioned because, on the one hand, some experts' previsions have failed loudly in the past (the Chernobyl-Exxon Valdez syndrome). On the other, high uncertainty, common to many environmental issues, reduces substantially the predictive capacity of scientist and engineers. If predictability (and explanatory capacity) is lacking, experts' stance is not necessarily more suited than any else to resolve problems. Furthermore, strong disagreements inside the expert community may come up, complicating rather than simplifying the task for decisions-makers, who have to choose between *a priori* equally well-founded technical perspectives. Paradoxically, in spite of the trust-crisis on scientific institutions, lay people and grassroots organisations are increasingly aware of their dependence on expert knowledge to address environmental decisions (Couch and Kloll-Smith, 1997). Coalitions between protest groups and “alternatives” technical visions (to the conventional power-knowledge alliance) are frequent to occur. This has happened in the case of Tambo Grande, where “science” has been used by two communities of experts to support different value-systems. These value-systems differ on the perception of environmental risks, the confidence in institutions, the preferred decision-making mechanism and the conception of legitimacy. This example supports the idea that, in most social decisions, science is “merely an ammunition” in a larger debate about values, interests and power (Otway, 1992).

For Manhattan, the entrepreneurial class of Piura and the government, the assessment of the project should be legitimised through an expertise system, where experts from the private sector make an EIA and “public” experts evaluate it. In this system, participation requires expertise and resources, because the only legal space for feedback envisioned in the process is a public audience for discussing technical aspects of the EIA. Thus, project approval is restricted to a dialogue between experts. Experts legitimise the decision because lay people should recognise their inability to state opinions on a technical matter. The alternative value-system of opposing groups is based on four main milestones: distrust in government institutions, unacceptability of environmental hazards, rights to decide and equity. A combination of several aspects are involved in “unacceptability of environmental hazards”, some of these are the occurrence of severe stochastic natural variations in the area and distrust in mining experts. The Peruvian history is full of examples of experts and institutions fallibility as regards mining projects. From this view, the problem, although having an important technical component, is also ethical and necessarily political. Hence, legitimisation of the decision can be only reached through direct consultation to the population. The clash between both value-systems is revealed when Manhattan disqualifies the position of some opposing groups, blaming them as “political”.

Funtowicz and Ravetz (1992) schematise decisional conditions in a diagram plotting “uncertainty” against “decision’s stakes”. When both variables are low, they argue, decisions belong to the domain of applied science. When both variables are intermediate, the decision is in the realm of professional consultancy. Finally, as was stated before, when uncertainty and the decision's stakes are both high we are in the sphere of post-normal science. Uncertainty and the decision's stakes are value-dependent and, to a certain extend, a social construct. When talking about uncertainty, one should keep in mind the questions "according to which value-system or according to which expert community?". The same holds for decision's stakes. In this case, "for whom?" is the following relevant question. In the Tambo Grande project, the pro-project stakeholders conceive the decisional conditions as belonging to the "professional consultancy" domain, whereas the opposing groups envision it as fitting in the post-normal science category (although not using explicitly this category). We argue here that there is not an objective means to decide where to locate the decision in the above-described scheme.

Project assessment could be done following different methods: EIA, cost-benefit analysis (including economic risk analysis), multicriteria analysis, voting process, etc. Choosing the most suitable method would depend on the value-system adopted, including the notion of legitimacy. Any social decision cannot be "objective" or "value-free", however these categories can be used by stakeholders to support a certain value-system. These words traditionally have had an ethical weight in western culture. Habitually, the technocratic culture has disguised the political nature of planning decisions appealing to "objective" scientific methods, which are supposed to give respectability to decisions (García and Groome, 2000). In many environmental issues, technocratic views have been systematically privileged (Taylor and Buttel, 1992). Conflicts come to pass when stakeholders cannot agree on a common way for legitimising decisions. In this sense, conflict "resolution" is reached when a part of society have the power to impose its own value-system on others. That is, it is able to "legitimise" decisions. When a decision method is imposed, but it is still deemed as "illegitimate" by the majority, we can say a decision can be taken, but the conflict persists. It seems clear that a key aim for a democratic state should be to achieve "legitimate" decisions.

In our case study, the final decision can be legitimised only if the adopted decision mechanism represents a shared meta-value-system, e.g. democracy and the voting system. However, in implementing this idea, the government may face important problems, especially because of different geographical scales within the state. The concept of "majority" must have a geographical reference. In the Tambo Grande case, a serious problem is the definition of the geographical scale where legitimisation is intended. This would depend on ethical principles. One position on the subject is that local empowerment should not be hampered for the sake of the “national” interest. This stance can be defended using different languages; one of them is the plea for the human right to a “safe environment” (Nickel, 1993; Adeola, 2000). From such stand, any method for decision-making not relying mainly on local lay people participation would likely prolong the conflict across time, increasing the probability of local violent protests and weakening the pretended democratisation and renovation of the state. Real public participation will require strengthening democratic processes and institutional innovation (law reform). The Tambogrande conflict, coinciding with a supposed democratic transition, is an opportunity to trigger a debate about public participation and governance in Peru.

Governance is defined, in this (democratic) context, as the rules and institutions framing the social decision-making process. Legitimisation occurs when governance rests on a value-system shared by the majority, which does not jeopardise the value system of “minority” groups. If legitimacy fails, a conflict between decision-makers and lay people is likely to happen, and a typical “revolutionary” situation arises. In these situations, groups within society have the possibility to consolidate social support by pursuing radical changes in decision-making rules and institutions. From this point of view, legitimisation assures social stability and power consolidation. As it was stated before, struggles between different groups to impose a certain value-system (a decision-making method, a notion of legitimacy, etc.) often take the form of discourse-contests, but ultimately are power conflicts. In the present paper, we argue that, in environmental conflicts, environmental valuation procedures are an outcome of such language(value)-contests. Hence, they are essentially “fighting-tools” in power confrontations.

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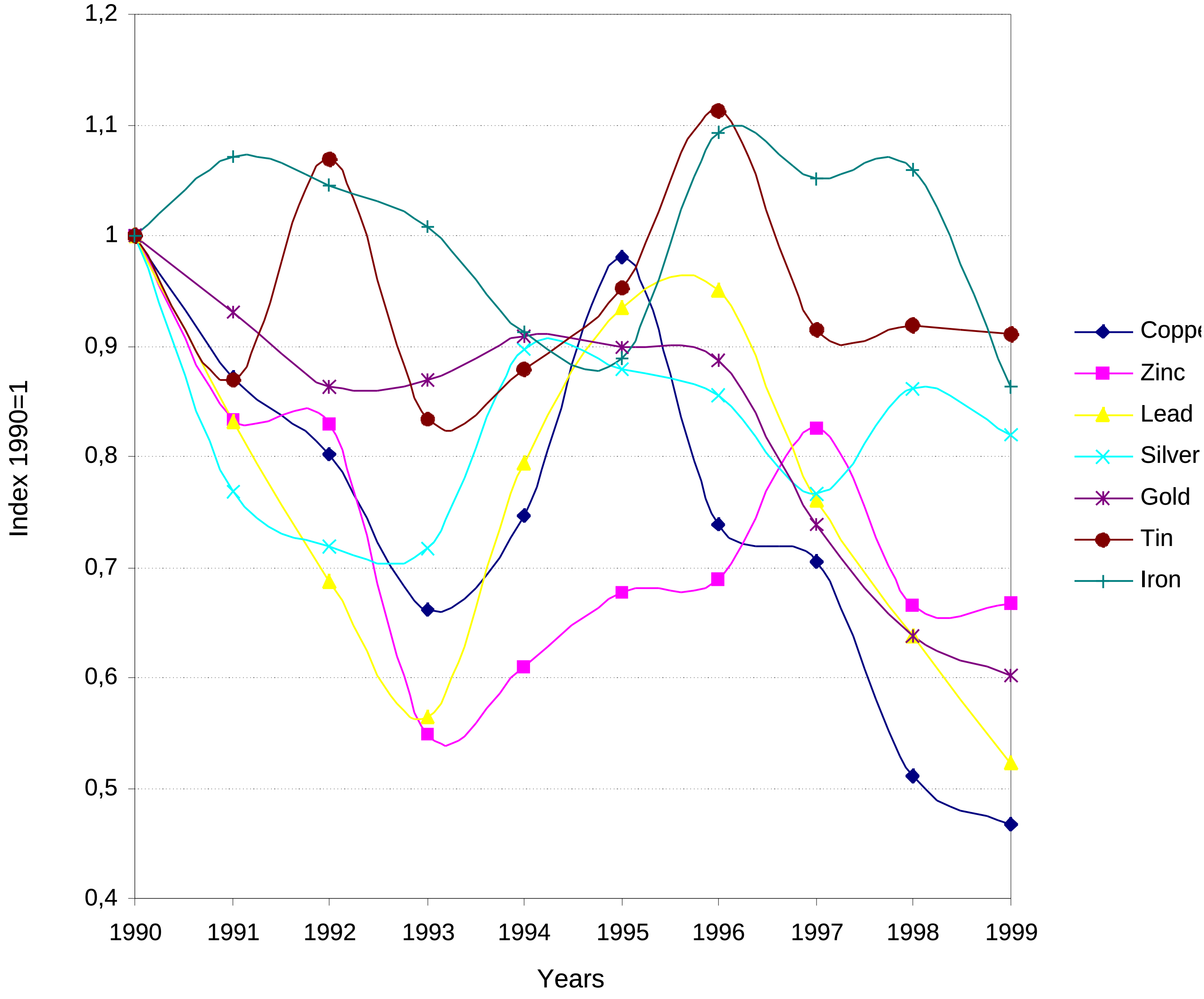
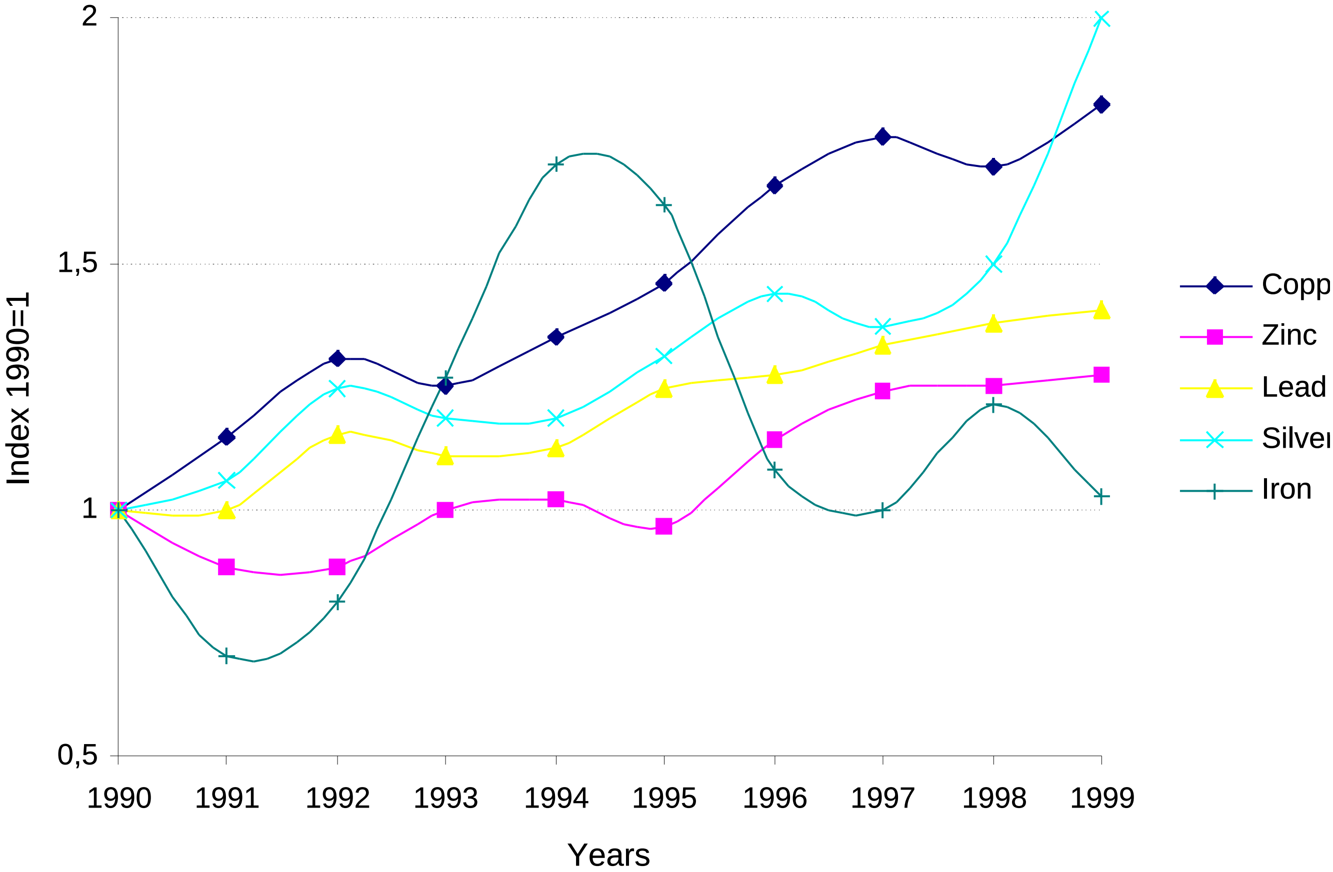
Figure 1. Peruvian Exports. Value / Weight. Constant $ 1987



Source: Authors calculations based on Referential Mining Plan 1999-2008. Ministry of Energy and Mines, Peru.

Figure 2. Peruvian Exports. Weight



Source: Authors calculations based on the Referential Mining Plan 1999-2008. Ministry of Energy and Mines, Peru.

Table 1
Background Characteristics of the Sample

Variable	N	Percentage
Sex		
Female	160	27,8
Male	415	72,2
Place of residence		
Rural area	163	28,3
Urban area	412	71,7
Level of education		
Without formal studies	41	7,1
Some primary school	70	12,2
Completed primary school	77	13,4
Some high school	63	11
Completed high school	173	30,1
Some Technical or university studies	151	26,3
Main economic activity		
Agriculture	225	39,1
Commerce	91	15,8
Housewife	63	11
Construction	16	2,8
Other non-professional services	37	6,4
Professional services	88	15,3
Student	41	7,1
Unemployed	11	1,9
No answer	3	0,5
Total household income (month)		
Less than 100 soles	26	4,5
100 to 200 soles	54	9,4
200 to 300 soles	100	17,4
300 to 450 soles	127	22,1
450 to 600 soles	75	13
600 to 1000 soles	86	15
More than 1000 soles	74	12,9
Without income	28	4,9
No answer	5	0,9
Average age: 36,9 years		

Table 2. Population survey. Summary of the results

ITEM	%	ITEM	%
Position as regard the mining project		**Do you trust in the government for assessing the project?**	
In favor	7.1	Yes	33.5
Against	85	No	43.5
Don't have enough information	6.6	Don't know	19.8
It is not interested			
Preferred decision-making mechanism		**Do you trust in the government for verifying mining enterprises comply with environmental laws?**	
EIA	16	Yes	33.6
Referendum	58.3	No	46.3
Negotiation	9.7	Don't know	18.4
EIA and referendum	4.2		
Other combinations	1.7	**If your house and belongings are well paid, are you willing to be relocated?[1]**	
Indifferent	2.6	Yes	19
Don't know	6.8	No	76.4
		Don't know	5.7
Preferred scale of the referendum[2]			
Town of Tambogrande	11.6		
San Lorenzo Valley	35.5	**How would be pollution if the project were undertaken?**	
S. L. Valley and Tambo Grande district	10.5		
Tambo grande district	19.5	Very high	46.6
Piura Province	2.7	High	27.8
Piura departament	15.7	Moderate	5.4
Peru	4.6	Low	3
Don't know	1.6	Very low	0.5
		Don't know	16.7
Can agriculture and mining coexist?		**If Manhattan guarantees you that pollution will be very low, would you be in favor of the project?**	
Yes	7.1	Yes	12.9
No	87.8	No	80.9
Don't know	4.3	Don't know	5.6
Is needed more information about the mining project?		**Do you believe Manhattan is involved in the murder of G. García Vaca?**	
Yes	73.2	Yes	30.8

[1] Percentage to the total surveyed people living in the urban area
[2] Percentage to the total surveyed people choosing "referendum" as the preferred decision-making mechanism

No	20.2	No	18.9
Don’t know	6.3	Don’t know	49
ITEM	%	ITEM	%
If the project were undertaken, who would be the main beneficiaries?		**If the dwellers of the San Lorenzo Valley were shareholders of the project, would you be in favor of it?**	
Dwellers of the San Lorenzo Valley	3.3	Yes	14.4
Central government	12.4	No	75.8
Municipality of Tambogrande	3.0	Don’t know	8.5
Manhattan	51.7		
Dwellers of Piura city	1.2		
All the Peruvians	5.0		
Central government and Manhattan	13.2		
Other combinations	4.7		
Don’t know	2.6		

Table 3. Stakeholders' Position on Different Issues

Stakeholder	Position towards the project	Preferred mechanism for decision-making	Preferred scale for the referendum	Compatibility between agriculture and mining	Trust in government for assessing the project	Probability of ecological disaster	Trust in Manhattan's experts	Trust in the enterprise doing the EIA
Piura life and agriculture	Against	Referendum	District	Very low	Low	Very high	Low	Low
Commerce council	In favor	EIA, negotiation	-----	NEI	High	NEI	Moderate	High
Front of Defence	Against	Referendum	San Lorenzo Valley	Very low	Low	High	Low	Low
Municipality Tambogrande	NEI	Referendum	District	NEI	Low	NEI	Low	Low
Aider	NEI	EIA	-----	NEI	Don't know	NEI	Moderate	Don't know
Centro Ideas	Against	Referendum, negotiation	District	Very low	Low	High	Low	Low
Manhattan	In favor	EIA, negotiation, referendum	Department	Very high	High	Very low	Very high	Very high
Pidecafe	Against	Referendum	S. L. Valley, district	Very low	Low	Very high	Low	Low
Cipca	NEI	Referendum	S. L. Valley	NEI	Low	High	Low	Low
Diaconia	Against	Referendum	District	Very low	Low	High	Low	Low

NEI: Not enough information

Table 4. Matrix of Preferences

Stakeholder	Macro productive Alternatives: Business as usual	Mining	Agriculture diversification	Forest management	Aquaculture	Cattle-ranching	Handicraft	Agroindustry intensification	Agriculture intensification	Ecotourism
Piura Life & Agri.	2	1	4	5	4	3	4	5	5	5
Commerce council	1	4	4	4	5	3	2	5	5	2
Front of Defence	4	1	5	5	4	4	4	5	5	5
Aider	2	2	3	5	4	2	3	5	5	3
Ideas	2	1	5	4	2	4	4	5	5	3
Manhattan	1	5	5	5	4	5	4	5	5	4
Pidecafe	1	1	3	3	2	2	2	5	5	4
Cipca	1	2	5	5	4	5	3	5	5	3
Diaconia	3	1	4	4	2	3	3	4	5	5

Table 5. Matrix of Economic Feasibility Perception

Stakeholder	Macro productive Alternatives: Business as usual	Mining	Agriculture diversification	Forest management	Aquaculture	Cattle-ranching	Handicraft	Agroindustry intensification	Agriculture intensification	Ecotourism
Piura Life & Agri.	5	5	3	5	2	2	3	3	4	3
Commerce council	5	5	3	3	5	4	2	5	5	2
Front of Defence	5	5	3	4	2	3	3	5	4	3
Aider	5	5	3	5	4	3	5	5	5	5
Ideas	5	5	4	5	2	4	3	5	5	3
Manhattan	5	5	4	4	3	5	3	5	5	3
Pidecafe	5	5	1	1	1	1	1	5	5	3
Cipca	5	5	1	2	1	1	2	1	1	2
Diaconia	5	5	3	4	3	3	2	5	4	4

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SURVEY

Ecological thresholds: a survey

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Abstract

The existence of ecological discontinuities and thresholds has been recognised by ecological economics as a key feature to take into account in the study of environment–economy interactions. This paper reviews some theoretical developments and empirical studies dealing with ecological phenomena involving non-linear dynamics. The literature about this issue reveals that there is abundant evidence of discontinuities and threshold effects as the consequence of human perturbations on ecological systems. However, due to the complexities involved, the predictive capacity of ecology is limited and large uncertainties still remain. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: Ecological discontinuities; Ecological thresholds; Alternative states; Ecosystem resilience; Stable equilibria

1. Introduction

Vis-a-vis neo-classical economics, ecological economics has paid more attention to feedback effects, self-organisation, uncertainties and non-linear dynamics. From a co-evolutionary and adaptive perspective (Norgaard, 1992), ecological discontinuities, being an important source of uncertainty, are key to understanding the complex interactions between the economy and the bio-physical environment where it is embedded. The ecological economics literature about this topic has put emphasis on the formulation of environmental policy or management models as well as the proposition of adequate institutional responses when ecological thresholds are likely. However, there have not been many attempts to integrate the ecological literature into this discussion. This paper tries to review the theoretical developments and empirical studies dealing with discontinuities in the ecological science. It pretends to systematise useful ecological information for economists and policy-makers interested in non-linear economy–environment interactions. How often do ecological discontinuities occur in nature? How uncertain are they? Are the underlying mechanisms well understood? Around these kinds of questions some fruitful dealings between economists and ecologists may arise. The present paper pretends to collaborate with this dialogue.

Ecological discontinuities can be defined as a sudden change in any property of an ecological

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system as a consequence of smooth and continuous change in an independent variable. Ecological discontinuities imply critical values of the independent variable around which the system flips from one stable state to another, that is, ecological thresholds. The definition of thresholds can be arbitrary in the sense that it depends on the temporal and spatial scale adopted. In general, no description in ecology makes sense without reference to particular temporal and spatial scales (Levin, 1992). Ecological discontinuities are phenomena of special interest for economics when they comprise abrupt changes in the ecological services provided by the involved ecological system (Myers, 1996; Daily, 1997; Pimentel et al., 1997). These services can be supplied by different levels of the hierarchical organisation of ecological systems. Therefore, the most suitable temporal and spatial scales to address ecological discontinuities can vary broadly depending not only on the ecological system involved but also on the dynamics of the environmental services provided by it.

For the current theory of environmental externalities, to take into account ecological discontinuities is not a serious problem. Although present models usually assume a continuous relationship between production and environmental externalities (Turner, 1999), it would be not complicated to relax this assumption and to assume the possibility of discontinuities in the external cost curve. Rather, ecological discontinuities are a challenge for the economic theory because the threshold values and the magnitude of the change are generally uncertain. Perrings and Pearce (1994) deal with this problem by proposing a penalty to be paid when an environmental standard is surpassed. This standard would be a value of the independent variable that imposes a limit to the economic activity in order to maintain the risk of surpassing the ecological threshold at acceptable levels. The aim of this measure is to assure the resilience of ecosystems, which allows statistically predictable environmental externalities. This notion is based on the belief that once the ecological threshold is surpassed, the general equilibrium effects cannot be estimated ex ante. This kind of approach, based on safe-minimum standards, seems more suitable than market mechanisms to deal with complex economy–environment interactions because prices are unable to detect when a system is approaching a threshold (Bishop, 1993; Folke, 1999). However, if the critical values which may trigger an ecological discontinuity are basically uncertain, the safe-minimum standards can be very elusive and the process of standard-definition very conflictive. Hence, ecological research on the occurrence and predictability of ecological discontinuities can be a crucial input to the environmental decision-making process. The 'quality' of this input will determine the relevance of scientific advisory in the decisional process (Functowicz and Ravetz, 1994). Because of that, it is helpful to check the 'state of the art' of the ecological science on this subject.

The existence of ecological discontinuities is also a key feature to take into consideration for designing environmental management regimes. Perrings and Walker (1997) have shown, at a theoretical level, how different management strategies may influence non-linear and long-term dynamics of ecosystems. They point out that optimal management should be sensitive to key variables conditioning long-term resilience of the system, like fire frequency in grasslands (see below). Gunderson et al. (1997) introduce a set of propositions about the adequate scales of analysis, institutional strategies and variable characterisation and monitoring to deal with uncertainty linked to non-linear behaviour in natural systems. The aim of the present paper is not to go further on these issues, but instead to visualise the current state of knowledge about ecological discontinuities.

The second and third sections of this paper link the concept of ecological thresholds to the debate on the relationship between biodiversity and (a) resilience and stability of ecosystems and (b) ecosystem functions, respectively. Then, a fourth section reviews some examples of multiple states and ecological discontinuities in each of the current most important human impacts on ecosystems, population harvesting, pollution, habitat fragmentation, ecosystem management and biological invasions. The paper finishes with some concluding remarks.

2. Biodiversity, resilience and stability

The reference to the maintenance of ecosystems' resilience as a key aspect of sustainability and as a crucial aim for environmental policy has been very present in the ecological economics literature (Common and Perrings, 1992; Arrow et al., 1995; Folke et al., 1996; Ludwig et al., 1996; Berkes et al., 1998; Levin et al., 1998). According to this position, ecological discontinuities may arise when human intervention has reduced enough of the ecosystem's resilience. When the system loses resilience it becomes vulnerable to perturbations that earlier could be absorbed without structural change (Gunderson et al., 1997). Resilience has been defined in very different ways, but mainly two connotations dominate the ecological literature. The most common definition is related to the capacity of ecological systems to recover from a disturbance (Walker, 1995). Under this point of view, resilience can be measured by how fast the variables of the system return to their equilibrium following a perturbation (Pimm, 1984; MacGillivray et al., 1995). The second definition (the most used in the ecological economics literature) emphasises the existence of alternative states in ecosystems. Under this approach, resilience is the ability of the system to maintain its structure and patterns of behaviour in the face of disturbance, that is, the capacity to absorb perturbations and still persist (Holling, 1973, 1986). In this context, resilience can be measured by the magnitude of disturbance that can be absorbed before the system flips to another state — before an ecological threshold is reached. The former definition requires that at least one equilibrium or stable state exits and can be recognised. The second one assumes a set of variables through which persistence can be assessed.

In the ecological literature, resilience traditionally has been associated with the concept of ecological stability, one of the most nebulous terms in ecology. Grimm and Wissel (1997) have counted at least 163 definitions of stability concepts. However, despite this prolific production, only few basic components of the concept can be recognised. One is related to the permanence of a certain reference state, dynamics or a set of properties that identify the system. The other component is associated with the capacity to return to the reference state (or dynamics) after a disturbance. Thus, each of the previously mentioned concepts of resilience tackles complementary aspects of stability. Therefore, in order to avoid confusion, they should be renamed. Grimm and Wissel (1997) try to do this, proposing a new vocabulary in the ecological stability discussion. Under their classification, persistence and elasticity can be new names for the previous definitions of resilience.

Currently, the Earth is witnessing one of the highest rates of species extinction in its history (Ehrlich and Wilson, 1991; Myers, 1993; Smith et al., 1993; Hughes et al., 1997). Biological diversity decline has been proposed as a cause of ecosystem resilience loss (Walker, 1995). Starting from this proposition, for some authors, there exists a threshold of biodiversity below which ecosystems lose the self-organisation that enables them to provide ecological services (Perrings and Opschoor, 1994). From this viewpoint, it is crucial for environmental policy to understand the relationship between species richness and ecosystem stability. This theme has been a topic of interest for ecology since the 1950s. At that time, Elton (1958), based on some empirical evidence, naturalist observation and mathematical models, stated that complexity, measured as species richness, number of biological interactions (connectance) or interaction strength, begets stability. Moreover, MacArthur (1995) pointed out that the more possible energy pathways in a community, the less likely would be a change in species density as a consequence of other species outbreak or deletion. Elton's and MacArthur's propositions were adopted as 'conventional wisdom' until the 1970s, when theoretical models found that food web local stability[1] was inversely correlated to the number of species in the community (May, 1973). However, mathematical models dealing with the return speed to initial equilibrium did not show a unique relation between this variable and a community's complexity (Pimm, 1984).

[1] A system is locally stable when it returns to the initial equilibrium following a small perturbation.

After the mid 1970s, research on the diversity–stability problem started to be empirical (rather than comparative or theoretical). Usually, the strategy of these studies has been to estimate the ability of different plant communities (varying in species richness) to maintain total plant biomass after a perturbation. In general, biomass stability is positively correlated to species richness, which apparently contradicts the theoretical results. This relationship has been found for different kinds of natural perturbations, highly variable rainfall (McNaughton, 1997); grazing (McNaughton, 1993) and severe droughts (Leps et al., 1982; Frank and McNaughton, 1991). A positive relationship between stability and diversity also has been found for other communitys' aggregated parameters. For example, McGrady-Steed et al. (1997) report that for nutrient uptake and community respiration, variability decreases as biodiversity increases. Nonetheless, not all empirical studies have found a positive relationship between species richness and ecosystem stability. Some works show that the positive relationship between biomass stability and diversity is not maintained at trophic levels higher than primary producers (Hurd and Mellinger, 1971). On the other hand, using artificial seeded plant communities and based on measurements of the soil microbial respiration:biomass ratio, Wardle and Nicholson (1996) concluded that stability does not respond predictably to shifts in species richness.

Tilman (1996) reports empirical results where species diversity was positively correlated to stability of total plant biomass, but negatively correlated to the stability individual species' biomass. McNaughton (1997) also found that more diverse plots in African grasslands showed greater variation in species richness in response to perturbations than the less diverse ones. Equally, Sankaran and McNaughton (1999) report experimental evidence supporting the hypothesis that diversity promotes instability at the species abundance level. Frost et al. (1995) found that zooplankton biomass was more resistant to acidification than species-level properties. Schindler (1990) also reports that species or population level indicators were more sensitive than ecosystem-level properties in response to acidification and eutrophication in fresh-water ecosystems. Thus, the empirical evidence seems to reveal that increasing diversity beget stability at the community level but instability at the population level. According to Tilman (1996), these apparently contradictory results can be linked, taking into account the effect of inter-species competition (which is larger in more diverse communities). This effect may resolve also the apparent contradictions between empirical and theoretical results. This 'compensatory' mechanism predicts that ecosystems will be more robust to perturbation than their components. Thus, environmental services relying on ecosystem-level properties would be more reliable (more stable) than those depending on particular species or populations.

None of the previously mentioned studies show a clearly discontinuous relationship between stability and diversity. Nevertheless, McNaughton (1993) points out that "it is probable that there is a threshold of change that will overcome the damping effect of biodiversity, with an associated break point of ecosystem function to quite different levels". Thus, he suggests an ecological threshold depending on the magnitude of the perturbation (species deletion). On the other hand, Tilman and Downing (1994) found a convex (logistic-like) curve for the relationship between species diversity and community resistance (measured as total biomass). Although the experimental design of this study has been criticised (Givnish, 1994; Huston, 1997; Chapin et al., 1998), this result may indicate that above a certain diversity level there are no additional increments in ecosystem stability when biodiversity is increased. That is, there seems to be a certain 'saturation point'. If this pattern can be generalised, then there could be species diversity thresholds below which the system becomes increasingly unstable as diversity decreases. Above this 'stability' threshold, the system is robust enough to resist species deletion without undermining its capacity to absorb perturbations. However, with the current knowledge, it is not possible to propose this kind of pattern as a universal phenomenon across different systems, parameters and perturbations.

To study empirically the stability of real ecosystems requires identification of stable equilibria or persistent structures. This is undoubtedly a hard task for ecologists because events occurring during community succession or assembly can lead to important differences in community structure (Connell and Sousa, 1983; Drake, 1991; Dodd et al., 1995). Stochastic events, like the sequence of species invasion, can result in vast differences in community organisation after a perturbation. This makes the notion of equilibrium or attractor state very evasive and difficult to determine in real ecosystems (Jhonson et al., 1996). Because of that, for some authors, to look for a relationship between diversity (or complexity) and stability is pointless (Sutherland, 1981). Due to these inconveniences, despite the relatively abundant ecological literature dealing with this issue, it is still very difficult to arrive at some universal proposition. Perhaps the emphasis on these sweeping generalisations is the consequence of a tradition of analysis in ecology that has been inherited from classical physics (Holling, 1973), which does not take into account in an appropriate way the inherent complexities of the ecological systems and their relevant qualitative properties. On the other hand, for some ecologists the abstract nature of the resiliences concept and the lack of a generally accepted resilience measurement limits its applicability for guiding policy (Orians, 1996; Risser, 1996). From an operational perspective, the two most used connotations of resilience face difficulties because of the complications of identifying stable equilibria. Thus, the common statement in ecological economics, that environmental policies have to be designed intending to maintain ecological resilience, is helpless unless resilience is operationally well defined.

3. Biodiversity and ecosystem function

3.1. Theoretical hypotheses

Environmental services may rely upon the functioning of ecosystems, that is, on aggregated ecosystem processes involved in the flux of mass or energy. The relationship between biodiversity and ecosystem processes is a topic receiving increasing interest in the ecological literature. Vitousek and Hooper (1993) recognise three possible relationships between biological diversity and ecosystem-level biogeochemical functions, (1) linear; (2) asymptotic and (3) non-existent. Case 1 is also called the 'diversity–stability hypothesis' (Jhonson et al., 1996; Boucher, 1997). Lawton (1994) makes a classification where case 2 is called 'redundant species hypothesis'. He also identifies an 'idiosyncratic' hypothesis, which proposes a basically unpredictable relationship between diversity and ecosystem function. Another kind of relationship, comprising a range of possible responses, may appear if all species make a contribution in an additive but unpredictable way. This has been called the 'rivet hypothesis'.

The 'redundancy hypothesis' (Walker, 1992; Lawton and Brown, 1993; Walker, 1995) suggests that in terms of ecosystem function, there is a certain degree of substitutability between individual species, especially if they belong to the same ecological functional group. The term functional group refers to species of the same ecosystem, which share common features determining ecosystem functioning and organisation (Schulze and Mooney, 1993). The 'rivet hypothesis' (Ehrlich and Ehrlich, 1981; Lawton, 1993) suggests that the function of species in ecosystems can be analogous to the function of rivets in an airplane. Both systems can afford continual extraction of its constituent components without experimenting a loss of function. However, after a certain point this capacity is lost and only one additional species extinction (rivet popped) may cause a collapse in the functional properties of the system. This vision makes emphasis on the unpredictability of this possible threshold. Ehrlich and Walker (1998) note that there is no essential difference between the rivet and the redundancy hypothesis. The rivet hypothesis recognises the existence of redundancy in ecosystems but emphasises the ignorance of which species we can afford to lose. The redundancy hypothesis points out that deletion of some species may have no immediate significant impact on ecosystem function, but redundancy is likely to be key in the long run because it allows the system to have a higher capacity to afford perturbations

(Grime, 1998). That is, species redundancy contributes to ecosystem resilience (Walker, 1995).

According to the rivet-redundancy hypothesis, a 'non-linear' relationship between biotic diversity and ecosystem function may occur (Solbrig, 1993; Naeem, 1998). Starting from this proposition, Carpenter (1996) suggests a pattern for the relationship between diversity and ecosystem processes rate very similar to that schematised in Fig. 1, when 'biodiversity' is in the abscissa axis and 'ecosystem function' in the ordinate axis (case a). This is clearly a hypothetical case of ecological thresholds with a high degree of uncertainty because the threshold of diversity is basically uncertain with our current knowledge of ecosystems behaviour (Gitay et al., 1996; Levin, 1993).

3.2. Empirical evidence

Although the discussion about the relationship between species diversity and the rate of ecosystem processes started with Darwin, empirical tests intentionally addressing the issue are relatively recent. The debate started to receive considerable scientific attention (Baskin, 1994; Lacroix and Abbadie, 1998) after a laboratory study made by Naeem et al. (1994). They manipulated plant and animal community diversity to create low, intermediate and high diversity microcosms. The rates of five ecosystem processes were measured, community respiration; decomposition; nutrient retention; plant productivity and water retention. They found that CO_2 community consumption and plant productivity were positively related to species richness. For the rest of the processes there were no consistent patterns of variation. Tilman et al. (1996) made a field experiment where plant species diversity was also intentionally manipulated to address its effects on ecosystem function. They found that both plant productivity and resource utilisation were significantly greater at higher diversity levels. The pattern that better described the relationship between productivity and diversity was again a convex (logistic-like) curve. This pattern is consistent with the redundancy hypothesis and it is predicted by mathematical models assuming increasing competition and increasing chance of having better competitors as species richness increases (Tilman et al., 1997b). As it was stated above, a logistic-like curve rela-

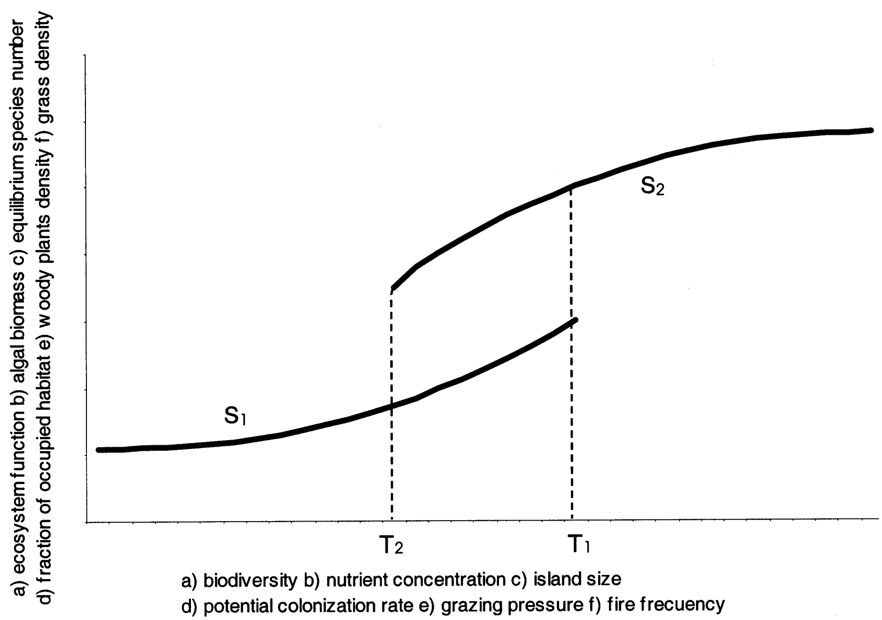

Fig. 1. Hypothetical ecological threshold.

tionship may encompass a certain 'saturation' (redundancy?) point (Tilman, 1997). This may imply also a diversity 'threshold', below which the system starts to experience increasing variability in its processes (McGrady-Steed et al., 1997) and increasing susceptibility to perturbation (in this case, to species deletion).

3.3. Species or functional diversity

Tilman et al. (1997a) made a field experiment, methodologically very to similar to Tilman et al. (1996), but this time controlling, in addition to species number, the functional diversity and the functional composition in a grassland plant community. They measured the rate of six ecosystem processes and concluded that functional group diversity may be a stronger determinant of ecosystem processes than species richness, per se. Hooper and Vitousek (1997, 1998) manipulated a grassland plant community to have a gradient of functional groups diversity, controlling also for composition (the identity of the functional groups). They also concluded that functional composition can have a larger effect on ecosystem processes than functional richness, per se. Wardle et al. (1997a) arrived at similar conclusions studying several ecosystem processes along a gradient of island-area (diversity). Wardle et al. (1997b), Nilson et al. (1999) studied the effect of litter composition manipulation and diversity on below-ground processes. They found that the nature of effects tended to be idiosyncratic — dependent on the particular species composition of the litter mixture.

For some authors (Grime, 1997; Wardle, 1999), these studies disclose a debate about the relative importance of biodiversity and species composition (functional types or species identity) in determining ecosystem function. According to Lawton et al. (1998a,b), this debate misses the point because no two plant species within a functional group have identical niches and because functional guilds are arbitrary divisions of continuous niches. Therefore, the results showing that functional composition explains more than species diversity about ecosystems functioning may be tautological, because by definition different functional guilds embrace species groups with large niche differences (Chapin et al., 1997). This implies that functional groups deletion necessarily will produce a great impact on ecosystem processes.

Studies dealing with the relationship between diversity and ecosystem function have been criticised because, according to some authors (Huston, 1997; Hodgson et al., 1998), they do not control in an appropriate way the functional traits of involved species. This does not establish an unequivocal relation between species richness and ecosystem function. However, these criticisms fail to recognise that it is impossible to manipulate species number without changing the species composition of the sample. For environmental policy, the relationship between species number and ecosystem function in abstracto is not as relevant as knowing which species are prone to become extinct, and to have the possibility to forecast the ecosystem-level consequences of their loss. Nonetheless, the ecological science is far from having this predictive capacity and much more research is needed in this area. Some issues where special attention should be paid are, (a) ecosystem consequences of natural extinction; (b) down and up food web effects of changing diversity at one trophic level; (c) the functional effects of diversity diminution on different marine ecosystems (Covich et al., 1999); (d) the ecosystem repercussion of below-ground biodiversity change, including microbial deletion, in terrestrial (Brussaard et al., 1997; Wall, 1999; Wall and Moore, 1999), fresh-water (Palmer et al., 1997) and marine ecosystems (Snelgrove et al., 1997); (e) the ecosystem role of individual species or processes, specially those being prone to be lost by human impacts (Risser, 1995); (f) the effect of different temporal and spatial scales, and the possibility of discontinuities' occurrence on the relationship between diversity and ecosystem function.

4. Examples of multiple states and thresholds

The existence of multiple stable states or alternative equilibria in ecological systems has been recognised since the 1960s in the ecological literature. Multiple stable points may occur at different levels of the ecological organisation, but usually they are defined as alternative species assemblage in a community (Scheffer et al., 1993) or multiple possible stable densities of a population (Loenardsson, 1994). Alternative assemblage of species may arise due to differences in the historical development of the community, like a different sequence of recruitment (Sutherland, 1974; Drake, 1990), or due to the effects of physical or biological perturbations, like changes in nutrient concentration or species deletion or invasion (Barkai and McQuaid, 1988). The flip from one possible population density to another may also be triggered by perturbations (May, 1977). Very often, the shift between alternative states in ecological systems occurs suddenly and comprises the existence of threshold effects (Wissel, 1984). This section will continue making reference to some of the already described examples of ecological discontinuities as a consequence of anthropogenic perturbations. The current five most common ways of human impacts on natural ecosystems are considered separately.

4.1. Population harvesting

Species abundance diminution as a consequence of harvesting is one of the most common perturbations on ecosystems. Humans can directly take away individuals for consumption (like in a fishery) or may induce indirectly higher rates of mortality in a certain population due to changes in its biological or physical environment. Multiple stable population densities are predicted by one of the simplest predator–prey models. Fig. 2 illustrates one of the possible outputs of this model where alternative states are present. P represents prey abundance. Following May (1977), the prey population rate of growth $G(P)$ in the absence of predation follows the solid curve. The rate at which P changes is given by $\mathrm{d}P/\mathrm{d}t = G(P) - C(P)$. The prey's abundance will tend to reach an equilibrium level when the natural growth rate equals the loss rate. Thus, at intermediate levels of predator's density, the points A and C correspond to locally stable states, whose domain of attraction are divided by the unstable equilibrium point

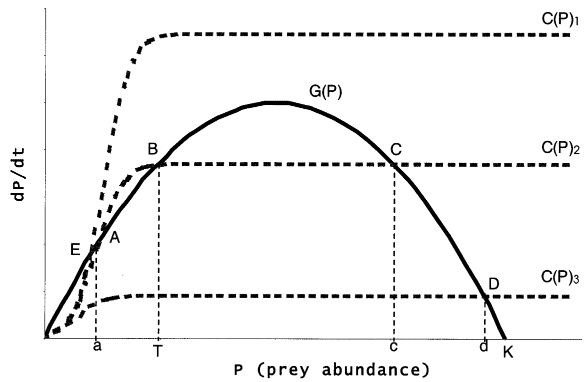

Fig. 2. Multiple states in predator–prey interactions.

corresponding to B. For high and low predator's density, the system will have only one equilibrium (D and E, respectively). If the predator's abundance is intermediate and the system is in the point C, prey population harvesting can move P to levels lower than 'T' (threshold). After that, the system may change to another domain of attraction given by A. On the other hand, if the predator's population is harvested, the rate of consumption may change, for instance, from $C(P)_2$ to $C(P)_3$ and the system may flip from C to D. This kind of model has been used to conceptualize alternative equilibria in, among others, grazing systems (Noy-Meir, 1975), zooplankton communities (Steele and Henderson, 1981), marine fish populations (Steele and Henderson, 1984), coral reefs species (Knowlton, 1992) and the ecological effects of introduced animals in islands (Sinclair et al., 1998).

Alternative species composition of Alaskan subtidal and intertidal kelp and sea grass beds associated with the presence or absence of sea otters have been one of the most cited examples of community alternative stable states. Otters have been hunted for trade and local consumption. Due to over-harvesting, in some locations they have completely disappeared. Presence of sea otters is correlated to communities with high macrophyte productivity, high fish diversity and density, low invertebrate productivity and high abundance of harbor seals. On the contrary, when sea otters are absent these communities are characterised by low macrophyte productivity, low fish diversity and density, high invertebrate productivity and low abundance of seals (Estes and Palmisano, 1974). Intermediate states between these two community configurations are reported to be both rare and transitory. Moreover, abrupt transitions between forested and deforested kelp forests have been reported worldwide (Estes and Duggins, 1995). The effects of otter disappearance on kelp forests are equivalent to overfishing in coral reefs, but in the latter case urchins blooms destroy the reef framework (Kaufman and Dayton, 1997).

Although the alternative states in the kelp forests comprise different species assemblage in a community, the above-mentioned population model has been used to explain this phenomenon. Estes and Duggins (1995) propose this kind of predator-prey model to describe the top–down control that otters exercise in kelp forests. In this community, otters are top predators who consume marine invertebrate, mainly sea urchins. Sea urchins graze on macrophyte, which are the most important source of food and shelter for fishes. Finally, fishes are the main food source of harbor seals. Thus, a radical change in the community species assemblage can be triggered by changes in the domain of attraction in predator–prey (otters-urchins) dynamics. In this case, it is a consequence of predator population harvesting. Graphically (in Fig. 2), this would mean, for instance, a change in the prey loss rate from $C(P)_2$ to $C(P)_3$. This involves a drastic increase in the prey (urchins) population (from 'a' to 'd'). This change of attractor encompasses a 'cascade' effect.

Interpreting paleoecological evidence in the Aleutian islands, Simenstad et al. (1978) suggest that overexploitation of otters is not a recent phenomena, but even occurred many thousands of years ago by the action of aboriginal Aleuts. This led the nearshore community to alternate between one state dominated by macroalgae and any other dominated by epibenthic herbivores. In response to this switch, human populations had to shift their diet from one based on fishes and marine mammals to an alternative one based on marine invertebrates. This is a clear example of how ecological discontinuities may comprise changes in environmental services. In this case, environmental services disruption was afforded by human population with an adaptive strategy, to shift the diet. Human adaptive strategies may reduce social costs associated with environmental (discontinuous) change. This can be a further complication to the already hard task of estimating social costs of environmental changes, because human adaptive responses to ecological discontinuities are difficult to predict. The adaptive strategy of human societies to cope with environmental change is one of the main arguments for adopting a technological-optimism perspective and to assume substitutability between natural and human-made capital in environmental economics.

The effect of otters on kelp forests is a typical example of keystone species. This term was proposed by Paine (1969) to emphasise the importance of top predators in determining the structure of animal communities by keeping low the density of prey species, which in the absence of the predation would dominate the system. Later, the term was extended to include those low abundance species whose deletion produce important effects on certain traits at the community level (Power and Mills, 1995). Keystone species may exercise great influence on the structure of communities through consumption, physical or biological habitat modification, mutualism, diseases, seed dispersal, resource provision, etc. (Power et al., 1996). Species which exert a great influence on the ecosystem level modulating the availability of resources to other species by physical state changes are also called ecosystem engineers (Jones et al., 1994; Lawton and Jones, 1995). The term keystone species has been useful in emphasising the inequalities in the relative importance of species in maintaining ecosystem function (Grimm, 1995). However, its application in environmental policy and conservation strategies can be less than straightforward because the label 'keystone' is relative to a temporal and spatial scale and also to certain assemblage of species and physical conditions, that is, it is context-dependent (Bond, 1993; Navarrete and Menge, 1996). This makes it difficult to confidently define a priori which species, and under which circumstances, are keystone (Mills et al., 1993; Paine, 1995). However, despite its operational difficulties, the keystone species concept may be very helpful to understand the mechanisms underlying ecological thresholds and discontinuities. It exemplifies very well how relatively small changes in the components of an ecosystem, like harvesting on a keystone species, may involve sudden and far-reaching shifts in the system's properties.

Overharvesting of natural populations may also influence natural populations cycles, making less predictable and more frequent natural fluctuations (Steele and Henderson, 1984). These fluctuations involving drastic, uncertain and rapid changes in population abundance, although not equivalent to shifts between alternative states because they are transitory, may encompass important economic costs. Classic examples of abrupt population abundance changes linked to overharvesting of fish populations are the sardine stock collapse in California and Japan in the late 1940s and the anchovy disappearance in Peru and Chile in the 1970s (Botsford et al., 1997).

4.2. Pollution

Changes in the physical environment of ecosystems may produce a switch between alternative equilibria. These cases are proposed as examples of alternative stable states because once the switch has already occurred, it is not enough to recover the initial physical conditions to return the system to the original state. One of the most cited examples of ecological discontinuities linked to changes in nutrients concentration is the existence of alternative equilibria in temperate shallow lakes. It has been reported that shallow lakes in temperate zones can present two basic organisations, a clear state dominated by aquatic vegetation (macro-algae) and a turbid state characterised by high algal (phytoplankton and cyanobacteria) biomass (Weisner et al., 1997). Continual nutrient loading to clear lakes can be afforded by the system until a certain nutrient concentration level is reached (threshold). After this point, a shift to the turbid state is highly probable. Once the system has switched to a turbid state and cyanobacterias become the most important primary biomass, it takes a strong nutrient reduction (much below the value of the initial threshold) to enable macrophyte recolonisation (Moss et al., 1996; Scheffer et al., 1997). This phenomena, as it is suggested by Scheffer et al. (1993), can be conceptualised graphically when the variable 'nutrient concentration' is in the abscissa axis and 'algal biomass' in the ordinate axis in Fig. 1 (case b). Thus, this is clearly a case where ecological thresholds and discontinuities are present. Allelopathic substances and habitat conditions encouraging predator development or avoiding light penetration are some of the proposed synergetic mechanisms to explain the stability of these alternative states. Due to the complexity of the biological interactions underlying this pattern, and because the switch from the clear to the turbid state can be mediated by stochastic events like hurricane winds (Bachmann et al., 1999), the precise value of the nutrient concentration threshold can be hardly predicted (Donabaum et al., 1999). Thus, in this relatively well-studied case of alternative equilibria, the thresholds values are rather uncertain. In this example, ecological discontinuities are associated with important loss of environmental services, mainly fishery and recreation.

4.3. Habitat fragmentation

Habitat fragmentation, the process of subdividing a continuous habitat into smaller pieces, is the major cause of species extinction (Tilman et al., 1994a,b). Currently, it is probably the most important way of human intervention on natural ecosystems. The existence of thresholds associated to the habitat size is well recognised in the ecological literature. The theory of island biogeography (MacArthur and Wilson, 1967), based on certain assumed behaviour of immigration and extinction rates, predicts a linear and continuous relationship between island size and species richness. However, Ward and Thornton (1998) suggest that there may be alternative equilibria at intermediate island sizes. This means that islands of the same (medium) size may have alternative equilibrium species numbers. This model predicts a relationship between island species richness and island size similar to Fig. 1, when the variable 'equilibrium species number' is in the ordinate axis and 'island size' in the abscissa axis (case c). This may comprise a sudden 'switch' between species-rich to species-poor habitats when island (patch) size is reduced below a certain threshold. Patches in a fragmented habitat can be considered ecologically equivalent to islands in a sea. Thus, these predictions also could be applied to the general relationship between species richness and habitat size. Actually, Metzger and Décamps (1997) propose a similar 'non-linear' and discontinuous relationship between habitat fragmentation and biodiversity. This relationship can be summarised in Fig. 3, when the variable 'proportion of destroyed habitat' is in the abscissa axis and 'biodiversity' in the ordinate axis (case a). This model is based on two basic theoretical developments about the ecological consequences of habitat fragmentation.

1. One is related to habitat's geometrical qualitative and quantitative changes along a gradient of fragmentation. Spatially explicit models predict the existence of thresholds determining discontinuities in the relationship between the proportion of the original habitat destroyed and landscape connectivity, the spatial contagion of habitats (With and Crist, 1995). These discontinuities arise because quantitative changes in the original habitat size lead to qualitative shifts in the properties of the patches. Bascompte and Solé (1996) represent these non-linear interactions in a graph similar to Fig. 3, where the variable 'largest patch size' is in the ordinate axis and 'proportion of destroyed habitat' is in the abscissa axis (case b). Hence, after a certain fragmentation threshold, the effect of destroying an additional portion of the habitat is no longer quantitative (the reduction of the largest-patch's size), but qualitative, the original habitat starts to be broken into smaller patches. After this threshold, the 'border effect' and isolation might strengthen the effect of habitat loss and the decline in population size will be faster than predicted by a simple diminution in the 'sample' area (Andrén, 1994). Population responses to changes in habitat size can vary greatly from one species to another (Diamond, 1982; Soulé et al., 1992; Kruess and Tscharntke, 1994; Tilman et al., 1994a,b; Lawton et al., 1998a,b). Because they are always referred to a species and to a certain spatial and temporal scale, to make these thresholds operational (predictable and measurable) for environmental policy embraces serious difficulties.

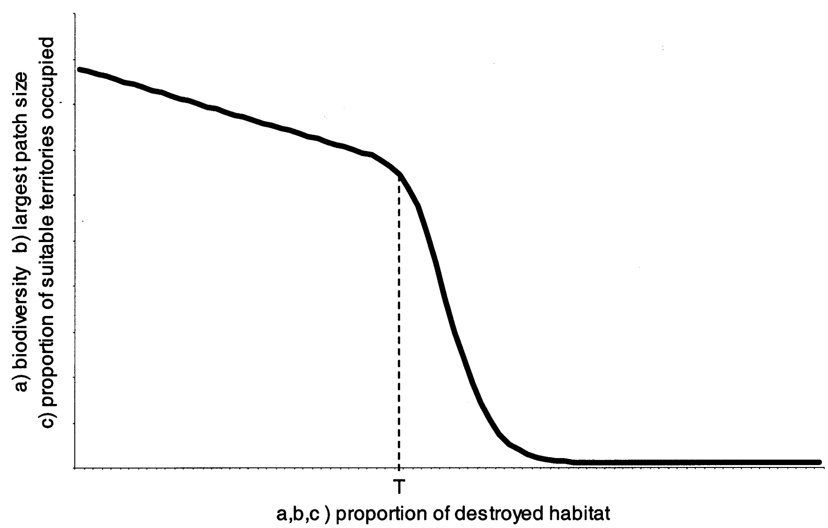

Fig. 3. Habitat fragmentation and ecological thresholds.

2. The other theoretical development predicting ecological thresholds linked to habitat fragmentation are metapopulation models, which propose multiple equilibria in the relationship between the probability of species occurrence and the area and isolation of the patches (Hanski and Gyllenberg, 1993). Graphically, this relationship, which has been supported empirically, can be visualised in Fig. 1 (case d), when the variable 'fraction of occupied habitat' (in a patch network) is in the ordinate axis and 'potential colonisation rate' (which depends on the patch size and the distance between patches in the network) in the abscissa axis (Hanski et al., 1995). At intermediate values of potential colonisation rate, small changes in patch size may cause the extinction of the metapopulation. This suggests the existence of an extinction threshold, which can be defined as the minimum proportion of suitable habitat that is necessary for population persistence (Lande, 1987). Extinction thresholds can be visualised in Fig. 3 (case c), when the variable 'proportion of suitable territories occupied' by a certain species is in the ordinate axis and the 'proportion of destroyed habitat' in the abscissa axis (Andrén, 1996). The results of some empirical studies are compatible with these propositions (Didham et al., 1998).

Extinction thresholds are produced by changes in the interaction between individuals and the surrounding landscape, but also by changes at the population or genetic level. At these levels there also can be discontinuities. Kuussaari et al. (1998) suggest that from certain population size down, the Alle effect (a diminution in population growth rate at low population densities) can play an important role in promoting extinction. Frankham (1995) reports empirical evidence showing that the most likely relationship between inbreeding (the probability to mate with relatives) and extinction (due to genetic reasons) is a threshold relationship. On the other hand, at higher organisational levels, system feedback and synergism may be involved in the appearance of discontinuities. As it was stated previously, species richness decline or keystone species loss produced by habitat fragmentation may lead to further species extinction due to ecosystem function or stability loss. This may generate a cascade effect and a switch in ecosystem properties.

4.4. Ecosystem management

Human intervention on natural ecosystems through manipulation of key internal or external variables can be also a source of ecological discontinuities. Human management of ecosystems usually is referred to as control of ecological perturbations. Manipulation of natural perturbations in grasslands is one of the cases where the role of ecosystem management in generating discontinuities has been addressed. Temperate savannahs around the world have experienced very often rapid switches from grass-dominated to woody-plants-dominated communities (Fuhlendorf et al., 1996). Overgrazing, fire suppression and CO_2 concentration rise have been cited as the most likely proximate causes of woody plant invasion into grasslands (Archer, 1989). All of these causes are related to human activities.

The most used rangeland management approach assumes that there is a 'climax' state in grasslands. According to this approach, any transformation in the properties of the system due to human perturbation may be reversed simply by suspending the perturbation (Laycock, 1991). Nevertheless, observations reveal that once the system has switched from grassland to woodland due to human management, stopping the perturbation during many years usually is not enough to recover the grass-dominated state (Walker et al., 1981). Once woody plants have substituted grass cover, positive feedback effects make the woodland system highly robust (Perrings and Walker, 1997). This led some authors to propose an alternative grassland dynamics model, which assumes the existence of alternative stable states and a non-linear relationship between perturbation intensity or frequency and grassland condition. This model makes explicit the existence of ecological discontinuities and thresholds (Walker et al., 1981; Archer, 1989; Fuhlendorf et al., 1996). Above a certain critical value of grazing pressure or below a certain fire frequency the system flips from the grassland to the woodland state. In order to recover the initial state, it is not enough to reduce grazing intensity or to increase fire frequency to the original threshold level. Reversion may require strong intervention by the manager, such as the application of herbicides or heavy machinery (Friedel, 1992). This dynamic can be visualised in a graph similar to Fig. 1 (cases e and f).

The case of fire reveals how management strategies intending to diminish variability may actually generate greater instability in the system. Suppression of forest fire in the national parks of the US has successfully reduced fire probability. Nonetheless, simultaneously this management has also produced fuel accumulation in the form of biomass, making the system prone to suffer fires of a magnitude never experienced before (Holling, 1986). After this kind of perturbation, the forest may switch to a different state, becoming a grassland for example. Another example, where human management intending to reduce variability in ecological systems may have produced the opposite results, is the control of the spruce budworm in boreal forests. This system experiences a natural cyclic behaviour. Periods of forest biomass accumulation and budworm low abundance are interrupted by periodic (20–40 years interval) and sudden outbreaks of the budworm density, which produces drastic diminution in foliage cover (Ludwig et al., 1978; Holling, 1992). This is a case of ecological thresholds in natural conditions. In order to preserve stable forest conditions, the pulp and paper industry has used insecticides to maintain low budworm density. This strategy has made the system prone to suffer disruptions due to pest outbreak in a magnitude never experienced before. As in the case of fire in forests, this extent of perturbation may switch the system to an alternative state difficult to reverse (for instance, from a conifer-dominated to an aspen-dominated forests).

Another case of natural cyclic pest outbreak occurs in larch ecosystems in central Europe. In this system, the larch bud moth rapidly increase to epidemic levels at intervals of 9–10 years and cause extensive diminution in larch foliage cover (Woodward, 1993). According to Holling (1986), management strategies that do not take into account the role of natural fluctuations in the maintenance of ecosystem properties may generate ecosystem resilience loss in the longterm. However, it is not clear the extent to which this kind of cyclic population behaviour is present in natural ecological systems. According to Holling (1992) cyclic population dynamics are common in nature. However, for Pimm (1984) 'such dynamics are theoretically fascinating but seem to characterise only a few populations'.

4.5. Biological invasions

Human activity has accelerated the rate of species invasion to new habitats by many orders of magnitude. Only in Canada, around 20% of current plant species were introduced by humans from elsewhere (Vitousek et al., 1997). Biological invasions are probably the second most important way of human-driven species extinction, after land use change (D'Antonio and Vitousek, 1993). Vitousek (1990) points out that biological invasions can alter ecosystem function by modifying three basic properties at the community level (a) resource or water flows; (b) the trophic structure and (c) the disturbance regime. The ecosystem effects of species invasion may be ecologically equivalent to the role of keystone species, exemplifying that the properties of individual species matter at the level of the whole ecosystem (Vitousek and Walker, 1989). Equally, they may comprise feedback effects and sudden shifts in ecosystem properties, although the notion of ecological thresholds has been less studied in the case of biological invasions. An example of the effects on nutrient cycling of invading species is the nitrogen-fixing tree *Myrica faya* in Hawaii. This plant, originally brought from the Azores and Canary Islands to Hawaii in the 19th century, possesses a nitrogen fixing actinorrizal symbiont which is absent in the native vegetation in Hawaii. Vitousek et al. (1987) determined that the presence of Myrica quadrupled the amount of nitrogen entering certain sites in Hawaii and increases the overall biological availability of nitrogen. This clearly alters ecosystem level characteristics, although the longerterm consequences of this alteration remains to be determined. In general, nitrogen addition in plant communities has produced a diminution of species richness (Vitousek, 1994; Tilman, 1997). Species richness changes, as it was stated above, can produce a cascade effect and discontinuities at the ecosystem-properties level.

The construction of the Erie and Welland canals in the 19th century induced an unexpected invasion of sea lamprey in the Great Lakes, which after 40 years produced a drastic abundance diminution in most of the economically important fish species. Lamprey invasion comprised substantial economic costs due to fishery and recreational activity decline. Predation was the main mechanism by which lampreys affected the structure of the Great Lakes fish community (Aron and Smith, 1971). In this sense, this example is equivalent to the previously mentioned sea otter case in Alaska. This constitutes an example of how invasive species may cause major and sudden disruptions in environmental services by altering the community trophic structure.

The effects of grass invasions on the fire regime are a worldwide well-documented event. The increase of fire frequency as the consequence of alien grass invasion has been reported in Latin America, North America, Australia and Hawaii (D'Antonio and Vitousek, 1992). Many grass invasive species re-grow more quickly than native plants after fire. Moreover, they are more flammable and have fire-resistant seeds. Due to these properties, grass invasions may initiate a grass/fire cycle, where invading grasses promote fire, which in turn favours foreign grasses over native species (Hughes et al., 1991). This cycle may have important consequences at the ecosystem level and it can produce higher incidence of fire and a relatively rapid switch from woodland to grassland state. Actually, it has occurred in many parts of the world (D'Antonio and Vitousek, 1992). Undoubtedly, this could imply dramatic shifts in local ecological services and considerable economic costs.

5. Concluding remarks

There are multiple theoretical developments in ecology predicting the existence of ecological discontinuities triggered by threshold values of internal or external variables to the ecological systems. Empirical studies reveal that ecological discontinuities as the consequence of human impacts are not uncommon in nature. Nevertheless, these studies face some difficulties, mainly because the definition of alternative stable states is highly dependent on the chosen temporal and spatial scales, as well as on the adopted notion of attractor. These difficulties also impose problems to the two most common notions of resilience, a concept that has not received a very rigorous treatment in the ecological economics literature. For many systems, the causes triggering ecological discontinuities are relatively well known. However, at the current state of knowledge, the ecological science is more able to predict the magnitude of change (the possible alternative state) than the threshold values. As prices are equally not able to assess the proximity of a system to a discontinuity, it seems that neither specialists nor consumers are capable of predicting exact ecological thresholds. Further ecological research probably will improve this predictive capacity, but given the complexity of the phenomenon and the stochastic variables involved, large uncertainties are likely to remain. To develop methods to deal with these uncertainties is a compelling task for policy-makers and a big challenge for participatory decision-making processes.

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SOME FINAL COMMENTS

The papers here compiled deal with different subjects, from diverse viewpoints and methodologies. Each paper has its own discussion section. Therefore, a general discussion covering all the themes here treated would be too broad and repetitive. Instead, I prefer to summarise the main subjects we have addressed and to foresee possible future areas of research.

One of the main points made by the previous articles is the idea that new insights may arise when the relationship between trade and the environment is tackled from a “physical” perspective. As shown before, different conclusions can be achieved if either a monetary or material-flows approach is adopted. Taking into account these considerations is one of the contributions of ecological economics *vis a vis* environmental economics to trade-issues, and to the general discussion about *weak* vs *strong* indicators of sustainability. Another focus of the current work is the “structural” aspects framing economic activities. The viewpoint here presented consider as key for understanding trade-environment relations the conditions of the world trade system determining the international division of production, as well as the long-term environmental and development dynamic effects of economic specialisation. The institutional aspects of North-South environmental and economic relations were also stressed in order to envision the possible outputs of increasing economic globalisation. In this sense, the old structuralist paradigm of development was revisited, from an environmental standpoint and taking into consideration current international economic and political circumstances. Besides the international displacement of environmental loads, we argued that ecological discontinuities and irreversibilities have to be kept in mind when addressing the relationship between income growth and environmental conditions. Finally, we pointed out that power relations and the social context (confidence, risk perception, ethical principles, etc.) are crucial matters to consider for achieving a legitimate decision-making procedure, either at an international or at very local level.

The approach here presented also opens some future themes for research. An interesting (and not enough developed) research strategy for studying empirically North-South environmental and economic relations is to analyse with detail the different stages of South-North commodity chains, in terms of both value added and environmental inputs or burdens associated with every step. This would allow comparing alternative production-consumption chains ("normal" vs "fair" trade networks, oligopolistic vs competitive markets, etc.), in relation to efficiency, equity and the international distribution of environmental impacts or hazards. This would contribute to inform Northern consumers about the distant environmental consequences of their consumption patterns.

Another issue not enough dealt with in the literature is the role of transnational corporations in the international distribution of economic benefits and allocation of environmental burdens. The mining sector is especially interesting because it seems to be witnessing a global privatisation wave and increasing North-to-South displacement of production. Paradoxically, despite the fact that the influence of transnational corporations in the world trade system has increased considerably in the last decades, the academic literature and political attention to this issue have diminished. The after-Seattle globalisation debate provides a stimulating atmosphere for retaking and rethinking, with a renewed vision, these old themes.