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2010

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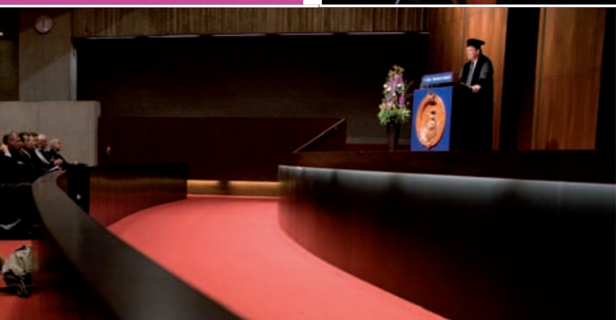

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Payments for Ecosystem Services:
Making Money Talk

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Rede uitgesproken bij de aanvaarding van het ambt van hoogleraar Economic Valuation of the Environment, in particular Water Economics, aan de faculteit der Aard- en Levenswetenschappen van de Vrije Universiteit op 15 april 2010.

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1. Introduction

Since the 1960s an important paradigm shift in science can be observed regarding our understanding of ecology, how it works, its complexity and dynamic internal and external relationships. These external relationships refer to what our natural environment and the ecosystems surrounding us mean for society. Parallel to the development of ecological system dynamics, increasing attention was also being paid to the co-evolution of natural and human systems in environmental and ecological economics. Take for example the work of Buzz Holling, an important natural scientist from the University of British Columbia in Canada who introduced the concept of functional response in population ecology and other intriguing ideas such as ecosystem resilience and adaptive capacity.

The way this was conceptualized and operationalized was through the functional performance of ecosystems. Humanity needs ecological systems, ecosystems in short, to survive. The organic and anorganic components and processes within ecosystems provide functions, which benefit human beings through the social and economically valuable 'goods and services' these biophysical functions provide. Examples include clean drinking water and food needed for humanity to survive, but also less obvious goods and services perhaps such as human health protection as a result of clean air, climate regulation and UV filtering by our ozone layer.

In the Netherlands Roefie Hueting was in the 1970s one of the first Dutch economists to point out the economic value of such functions based on what he referred to as functional scarcity. I'm grateful for having had the opportunity to work with him just before he retired and I just started as a freshman in this specific academic domain fifteen years ago. He wrote one of the first papers in the Netherlands on the topic with the title ‘Functions of Nature’ in 1969. He further elaborated the idea in his PhD published in English in 1980 under the title ‘New Scarcity and Economic Growth, More Welfare through Less Production’ [1]. One of his students was Dolf de Groot, now at Wageningen University, who wrote his PhD 12 years later on the same topic from a more ecological perspective also titled ‘Functions of Nature’ [2].

The basic idea behind ecosystem functions providing ecosystem services is simple and appealing from an economic point of view as it reduces the issue to a question of supply and demand. The environment supplies through its functionality goods and services which are in demand by human beings. We need clean water, air and food, all provided ‘free of charge’ by our planet, but also value nature as a place to enjoy recreational activities, find inspiration, and feel good, knowing that there is a place for wildlife with which we co-exist on this planet.

If demand for these goods and services is higher than supply and becomes competitive for the limited available goods and services, there is scarcity. This competing demand strips down the issue of sustainable environmental management, meeting demand now and in the future and carefully balancing social, economic and ecologic interests at the same time, to a fundamental economic question. This idea goes back to theories of classical economists like Thomas Malthus and David Ricardo in the 18th century who expressed their concerns about the limited availability of one of the first natural resources considered in economic thinking, namely land for agricultural production. Malthus was primarily concerned about absolute scarcity of land, so the available amount of land, while Ricardo thought it would be the quality of the natural resource land that imposes limits to economic growth.

Even though there exists no market for many ecosystem services where they are traded as a result of supply and demand, they do have value and are not really 'free of charge' or 'cost-free' so to say, as there are almost always opportunity costs involved. For instance, their current use is at the expense of their future use as is the case for non-renewable energy resources such as oil and gas, or their use by one particular group is at the expense of another group of people as is the case when we have overcrowded common pool resources like natural parks or beaches, or discharge wastewater in a river impairing recreational fishing opportunities.

The trade-off between economic development on the one hand and environmental conservation on the other hand based on an assessment of the supply and demand of ecosystem goods and services is nowadays generally accepted by both natural and social scientists. In the natural sciences important research questions relate to improving our understanding of the bio-geochemical structure and processes underlying ecosystem functions. For example, what is the role of biodiversity in ecosystem processes, how do nitrogen cycles induce environmental change and affect the provision and quality level of ecosystem services.

In the social science domain and then particularly in economics, the trade-offs mentioned here are also accepted in the two main strands of 'environmental economics' and 'ecological economics'. In the social science domain the question is more one of ideology, i.e. whether one believes that the economy is a subset of the ecosystem as is the case in ecological economics, or the other way around that natural capital is a subset of the human economic system, which is usually the overall approach and assumption in environmental economics. In practice, however, both strands of thought apply similar economic valuation and evaluation methods focusing on the same ecosystem goods and services. In the ecological economics literature perhaps more attention is paid to the underlying biophysical characteristics of the ecosystem providing the goods and services than in the environmental economics literature, and more use is made of evaluation tools like multi-criteria analysis. Ecological economists tend to claim to be more aware of the limits to economic growth and the extent to which natural capital can be replaced by man-made capital. They prefer to work (more) with safe minimum standards and the precautionary principle.

Valuation of the environment, the name of the chair to which I have been appointed last year, plays an important role in both strands of academic thinking and research. Economic monetary valuation is not always considered a plausible or realistic option by ecological economists. Environmental valuation to inform policy and decision-making, especially with the help of so-called stated preference research methods, is often considered too restrictive, narrow, and simple by ecological and institutional economists to properly describe (1) the environmental values people hold, (2) the process of value construction or (3) the way individual values are aggregated into a social value. Stated preference methods simulate market behavior to capture the public value of ecosystem services and have their roots in neoclassical economic theory based on the assumption of individual utility maximization. However, here too the academic debate is sometimes somewhat blurred by subjective beliefs about how policy and decision-making *should* be organized and informed based on more participatory approaches, including for example citizens' juries. In some of my early work published in journals such as Environmental Values and Ecological Economics I argued for combinations of the two and showed that the two do not necessarily bite and can go hand in hand [3,4].

In the context of ecosystem services valuation and payments for ecosystem services, one of the greatest challenges is to link the natural and social sciences. That is, the intermediary platform between the ecological and economic system, specified and measured in terms of changes in the provision of valuable environmental goods and services. It is relatively easy to qualify these goods and services and come up with generic lists like the Millenium Ecosystem Assessment or the one presented in Table 1 taken from the AQUAMONEY guidelines, a project I coordinated for the European Commission.

Table 1: Example of aquatic ecosystem functions and their goods and services

	Goods							Services																				
Function \ Outcomes	Potable water for household use	Water for landscape maintenance and peat soil	Water for crop irrigation	Water for livestock consumption	Water for food processing	Water for other manufacturing processes	Cooling water for power plants	Water transport	Prevention of saline intrusion	Water/soil support for prevention of land subsidence	Natural erosion, flood and storm protection	Shoreline stabilization	Sediment removal	Transport, treatment, medium for wastes and other by-products of human activities	Improved air quality through the support of living organisms	Biological diversity provision	Recreational swimming, boating, fishing hunting, trapping and plant gathering	Commercial fishing, hunting, trapping and plant gathering	Energy production	On and off site observation and study for leisure, education and scientific purposes.	Micro-climate regulation	Macro-climate regulation	Toxicant removal	Toxicant export	Cultural value provision	Historical value provision	Aesthetic value provision	Wilderness value provision
Water discharge	■	■	■	■	■	■	■	■						■	■	■	■	■		■	■			■				
Water recharge	■	■	■	■	■	■	■		■	■					■	■	■			■	■							
Flood mitigation										■	■					■				■								
Sediment retention	■							■		■	■	■	■		■	■	■	■		■			■					
Nutrient retention														■	■	■	■	■		■							■	
Nutrient export														■	■	■	■			■								
Trace element storage																■	■	■		■			■					
Trace element export																■				■				■				
Carbon sequestration																■	■		■	■		■						
Biodiversity maintenance															■	■	■	■	■	■	■						■	■
Culture/heritage																	■			■					■	■	■	

Source: Brouwer et al. (2010).

It is however difficult to specifically measure and quantify how changes in ecosystem performance affect the amount and quality of ecosystem good and service provision. These cause-effect relationships are often surrounded by tremendous uncertainty and therefore simplifying assumptions have to be made in order to be able to say anything at all about the expected impacts of, for example, reduced nutrient runoff from land upstream on water quality improvements dow*n*stream. It is also difficult because these impacts are measured by scientists in units like milligrams of nitrogen per liter, which then still have to be linked somehow to a variety of goods and services. Examples include the change in fish stock and recreational fishing opportunities, and the corresponding increase in the number of days a fisherman can go out fishing or the likelihood a fisherman will catch a specific fish species (catch rate). It is this latter kind of information that is used by environmental economists to value the recreational impacts of water quality improvements and it is this kind of information that is often not available.

In the European research project AQUAMONEY an attempt was made to create such a link for a number of water quality problems through the creation of so-called water quality ladders. These ladders display improved water quality levels on the one hand and their implications for a variety of water goods and services on the other hand. Spatial maps were used in case studies to test these quality ladders, where different colors of rivers indicated different water quality levels and corresponding water services. Although these maps and water quality ladders came about in close collaboration with ecologists, they were still based on assumptions about what would likely happen if water quality would improve to a certain level.

In order to be able to analyze the socio-economic value of for instance water quality improvements, one needs to know how these quality improvements affect ecosystem goods and services. And this remains a big puzzle despite decades of research and the recent popularity of payments for ecosystem services. In some cases we merely have some rough indication of what we think we are valuing in terms of ecosystem goods and services. This includes the impact of setting land aside upstream to reduce water quantity and quality problems downstream in many Payments for Ecosystem Services schemes.

2. Ecosystem services valuation

Economic valuation of the environment traditionally serves a number of purposes, including raising awareness. The fact that ecosystems provide goods and services, which have economic value, was given an enormous impetus in 1997 with the publication of the (in)famous paper in Nature by Robert Costanza and colleagues, some of whom economist, but most with a background in ecology and environmental sciences [5]. They suggested based on their calculations that the world's ecosystem services and natural capital, most of which are not traded in economic market systems, are worth almost twice the value of the world's gross national product. Although these estimates were surrounded by enormous uncertainty, the authors were nevertheless able to pin down this uncertainty to a value range between 16 and 54 trillion dollars per year, so say 50% more or less the estimated average of 33 trillion dollars. Ecosystem services provided by water and wetlands were among the most valuable assets on this planet according to this study, followed by tropical forests.

The paper evoked a lot of criticism, for instance in a special issue one year later in the journal Ecological Economics to which I contributed in a modest way with my colleagues then from CSERGE in the UK Kerry Turner and Neil Adger [6]. Although the paper was said by some of the authors to primarily deliver a policy message, many environmental economists felt that the simplicity of the exercise did no justice to the hard work in the past decades by scholars to test the validity and reliability of nonmarket valuation and benefits transfer specifically. Critical ecological economists considered it to be a grotesk example of benefits transfer by assuming that average value estimates from around 100 studies could simply be added and transferred through time and space, across different biomes and land uses, and groups of people and cultures.

In 1997 we already knew, for example based on meta-analysis, that value estimates are not only a function of the good and service characteristics, but also the beneficiaries of the goods and services and differences in applied valuation methods. Meta-analysis is a statistical method to detect systematic variation in observed outcomes for example of wetland valuation studies, which produced the highest economic value flows according to Costanza and colleagues.

In Table 2 I present the results from three different meta-analyses conducted by three different groups of researchers, myself and others published in 1999 [7], Richard Woodward and Yong-Suhk Wui from Texas A&M University published in 2001 [8], and my IVM colleagues Luke Brander and Jan Vermaat published some years later in 2006 [9].

These meta-functions show the impacts of a variety of explanatory factors on the variation found in different wetland valuation studies. So, the dependent variable here is the wetland value. Based on improved understanding of what drives valuation results, the explanatory power of the models increases somewhat in time. However, especially the high R-square of Woodward's model is striking based on the lowest number of observations. Although the numbers are not directly comparable, because they were measured in different units, there are four main points I want to make showing this table, which directly influence and at the same time undermine the validity and reliability of an exercise like the one published in Nature.

Table 2: Overview existing meta-analysis results for wetland ecosystem functions

Study	Brouwer et al. (1999)		Woodward and Wui (2001)		Brander et al. (2006)	
Dependent variable	1990 SDR/household/year		1990 US$/acre/year		1995 US$/hectare/year	
	Explanatory variable	*Coeff. estimate*	*Explanatory variable*	*Coeff. estimate*	*Explanatory variable*	*Coeff. estimate*
Constant		3.356***		7.872**		-6.98
Socio-economic					GDP per capita (ln)	1.16*
					Population density	0.47***
Wetland type			Coastal	-0.117	Salt marsh	-0.31
					Freshwater marsh	0.22
					Unvegetated sediment	-0.56
					Mangrove	-1.46**
					Wooded wetland	0.86**
Wetland function	Flood control	1.477***	Flood control	0.678	Flood protection	0.14
	Water supply	0.691**	Water supply	0.737	Water supply	-0.95
	Water quality	0.545*	Water quality	-0.452	Water quality	0.63
			Recreational fishing	0.582	Recreational fishing	0.06
			Bird hunting	-1.055**	Recreational hunting	-1.10**
			Amenity	-4.303**	Amenity	0.06
			Habitat and nursery	0.427	Habitat and nursery	-0.03
			Storm protection	0.173	Materials	-0.83**
			Bird watching	1.804**	Fuel wood	-1.24***
			Commercial fishing	1.360	Biodiversity	0.06
Wetland size			Acres (ln)	-0.286**	Hectares (ln)	-0.11**
Continent	North America	1.861***			South America	0.23
					Europe	0.84
					Asia	2.01
					Africa	3.51**
					Australasia	1.75*
Other geographic variables					Latitude	0.03
					Latitude squared	-0.00
					Urban	1.11**
Valuation method	CV open ended	-0.411***	Hedonic pricing	5.043**	CV	1.49**
			Travel costs	0.273	Hedonic pricing	-0.71
			Net factor income	2.232**	Travel costs	0.01
			Replacement costs	-0.341	Net factor income	0.19
					Replacement costs	0.63
					Production function	-1.00
					Opportunity costs	-0.03
					Gross revenues	-0.04
Payment vehicle	Income tax	1.880***				
Welfare measure			Producer surplus	-3.140**		
Study quality	Response rate 40-50%	-2.253***	Published	-0.154		
	Response rate > 50%	-1.904***	Data	0.000		
			Theory	-1.045		
			Econometrics	-3.186**		
Study year			0.016			
Other variables					Ramsar proportion	-1.32*
					Marginal value	0.95*
Pseudo R^2		0.365	R^2	0.582	Adjusted R^2	0.450
N		92		65		202

Source: Brander and Florax (2006).

First of all, the empirical evidence regarding the values of ecosystem services is mixed and this is probably due to the point I made earlier that our understanding of the link between ecosystem functioning and the provision of specific quantified ecosystem goods and services is still very limited. For instance, I found a significant impact of flood control, but this effect was not found in the other two studies which included a much more detailed list of ecosystem services. A lot of these more detailed services are hidden in the four categories I used. The fourth category in my study (not shown here) was the baseline category biodiversity. Brander and his colleagues show that ecosystem services with mainly direct use value are valued significantly less. In my own analysis the positive outcomes of the parameter estimates for flood control, water supply and water quality which have important direct use value to human beings too, indicate that these services are valued significantly higher than the biodiversity baseline category, which mainly consists of nonuse values. Nonuse values are values which are not directly related to any specific environmental use, but reflect the value people attach to the preservation of wildlife or conserve the environment for future generations. For instance, measured through donations we make to organizations like WWF and Greenpeace. So, more attention needs to be paid to linking ecosystem functions to ecosystem services and their specification to solve some of these apparently inconsistent findings in the literature.

Second, there exists no such thing as a constant ecosystem service value per hectare or acre that can be transferred and aggregated unconditionally across the world. The estimated values are sensitive to the size of the wetland, that is, sensitive to scope, and display in the two studies where this was tested decreasing returns to scale. This is a basic economic law: economic values are expected to decrease as scarcity increases and values diminish if there is more available of a good or service.

The third point I want to make is that not only size matters, also the number of people benefitting from the natural resource influence values. This is shown in the study by Brander et al. through the significant positive coefficient for population density. In my own study I measured wetland values per household per year to emphasize the fact that these values have to be aggregated across populations of beneficiaries, not merely biomes. Moreover, it is in most studies unclear if these values even have an underlying spatial dimension, which would allow us to express the values in a per hectare value. For example because of nonuse values.

My final point is that it matters where these wetlands are found and related to that the overall welfare level of the people living there as shown by the impact of GDP per capita. What people pay or are willing to pay depends also on what they can afford to pay.

So in conclusion, meta-analysis is a useful tool to summarize results from existing valuation studies and their application has increased substantially in the environmental valuation domain since the start of the 1990s. They were available for the global valuation exercise by Costanza and his colleagues. Far from perfect still, but nevertheless making those using valuation results in actual policy and decision-making aware of the pitfalls of simply transferring average point estimates, which is still common practice unfortunately in many if not most studies informing environmental policy based on environmental valuation of benefits.

3. Environmental damage assessment and liability

I now turn to another important use of valuation results. In the US, contrary to Europe, nonmarket values are used in court, the most famous example being the court settlement with Exxon in the 1990s after the Exxon Valdez oil tanker disaster in Prince William Sound, Alaska, the largest oil spill in US history. Exxon was ordered to pay the State of Alaska 287 million dollars for actual damages and 2.5 billion dollars for punitive damages on top of the 2 billion already paid to clean up the spill. This was based on the 2.8 billion dollar estimate from Richard Carson and his colleagues [10], a well-known scholar in the field of environmental valuation at the University of California, for the loss of nonuse values as a result of the spill. This study evoked a lot of discussion and resulted in new guidelines, the so-called NOAA-guidelines [11], for the valuation of environmental damage, which were thoroughly tested and refined in subsequent years.

In Europe, no cases are available yet where nonmarket valuation results are actually used in court. However, this may change as a result of the Environmental Liability Directive, which came into force in 2004 with the aim to prevent and remedy environmental damage, especially to habitats and species protected under European law. The Directive enforces the Polluter Pays Principle, making those responsible for environmental damage financially liable for environmental risks and actual environmental damage.

The Environmental Liability Directive prescribes the use of resource-service equivalence approaches. However, '*if it is not possible, alternative valuation techniques shall be used* [...] *for example monetary valuation*' (Annex II, paragraph 1.2.3). In addition, natural resource services are defined in the Directive as '*the functions performed by a natural resource for the benefit of another natural resource or the public*' (Article 2(13) and paragraph 1(d) of Annex II). Hence, when developing remediation options, the loss of the services contributing to public well-being should also be taken into account. This implies estimating the socio-economic value attached to the ecosystem services involved. For this, economic valuation methods may have to be used in view of the fact that existing habitat and resource equivalency methods do not include human welfare considerations.

In 2008 I participated in a European research project aiming to illustrate the potential use and usefulness of such nonmarket values. Paid by the European Commission I revisited together with my colleagues Harry Aiking, an eco-toxicologist, and Julia Martin Ortega, an environmental scientist, one of the largest toxic spills in European history, namely the one near the Doñana National Park in South Spain in 1998. We assessed the welfare implications of the environmental damage there using nonmarket valuation methods.

In April 1998, a dam of the pyrite mine owned by the Swedish-American company Boliden Apirsa in Aznalcóllar, 50 km north of the Doñana National Park, broke down and resulted in the release of 6 million cubic metres of toxic water and sludge into the Guadiamar river, a tributary of the Guadalquivir river. This contaminated material washed 40 km down the river, killing thousands of fish and birds, causing serious ecological damage. Most of the spill could be diverted away from the National Park by a series of quickly constructed dams outside the Park, home to many protected species including the Iberian lynx. Of the total area affected by the toxic spill (4200 hectares) only 100 hectares were in the National Park. However, also around the park, the water provides an important breeding and feeding place for migratory birds.

The company was fined almost 45 million dollars in 2002 by the Spanish Government, the largest fine ever issued by the Spanish government, including compensation for the ecological restoration costs. Just like Carson and his colleagues, we tried to assess the welfare loss of the environmental damage caused, only this time 10 years after the spill and at considerably lower cost than the million dollar budget they had. We examined taxpayers' willingness to pay for additional nature protection measures to avoid the risk of a disaster like the 1998 spill from ever happening again. Our main objective was to test the suitability and reliability of available nonmarket valuation techniques for environmental damage assessment in the specific context of the Environmental Liability Directive. In doing so, we tried to follow as much as possible the NOAA guidelines.

A bidding game was developed where people living at different distances from the toxic spill were asked a series of questions to elicit their willingness to pay. They were first asked whether they would be willing to pay a particular start bid, say 10 Euro and then depending on their reply to this start bid either a lower or higher follow-up bid. For example, if they said no to the 10 Euro, they were then asked if they would be willing to pay 5 Euro. If they said yes to the 10 Euro, they were asked if they were also willing to pay 15 Euro. In this way we are able to establish the interval in which public willingness to pay falls. One of the pitfalls of the approach is that the follow-up bids may depend on the start bid and this has to be accounted for in the estimated choice model.

Estimating the model, we found an average willingness to pay of 3 and a half Euro per household per year over and above their current income tax, which is very small for an average disposable household income of around 1600 Euro per month. Aggregated across the population of direct beneficiaries in Andalucia, this comes to a total economic value per year of 17 million euro or 27 million dollar in 2008 prices. This is less than the fine imposed by the Spanish government. But, note that we estimated an annual flow value, whereas the fine is a one-time off payment. Moreover, the value we estimated is only aggregated across the population of Andalucia where we conducted the study. The Doñana National Park is the biggest RAMSAR wetland in Europe, and hence one could argue with much wider than just local or regional benefits.

In our report we concluded that the use of nonmarket valuation approaches in the context of environmental liability and damage assessment is not problem-free. One of the most important methodological issues is related to the polluter pays principle and public perception of property rights and legal entitlements to environmental goods and services. Like Carson in the US, we too found that a considerable share of the population, some 18 percent, refused to contribute financially to the prevention of environmental damage because the polluter should pay. Hence the reason why we estimated a selection model to account for protest response. Carson and his colleagues took a year pretesting their valuation design, we had to do it in 3 months given our financial resource constraints. In both cases it appeared impossible to get rid of persistent protest.

In those cases where the polluter is publicly known and the pollution is big news, like Exxon in the 1989 oil spill and Boliden Apirsa in the case of the Doñana, is it likely that fingers will be pointed first of all at the polluters. This limits the usefulness of stated preference methods for nonmarket valuation ex post, that is after the event. In those cases, one could try to reach an agreement beforehand on the height of the fines discouraging particular activities and risks near important natural resources. This could still be based on nonmarket valuation, besides traditional command and control instruments and the application of so-called habitat and resource equivalency methods to restore and compensate environmental damage and corresponding costs.

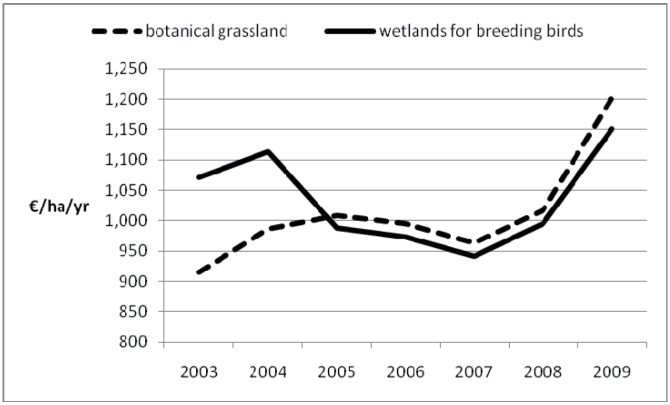
4. Payments for water services

Despite the absence of markets and prices for many ecosystem services, it is in some cases an observable fact that they have value and that they are assigned financial and economic values in policy and decision-making processes. Farmers get, for example, compensated for agricultural wildlife management. These compensations were introduced in the Netherlands after adoption of the so-called Relatienota in 1975, to account for the negative externalities of modern-day agriculture and encourage the production of positive externalities such as biodiversity and landscape amenities. Under the umbrella of the European Common Agricultural Policy, other European Member States such as the UK, Belgium, France, Italy, Ireland and Finland have introduced a wide variety of agri-environmental schemes to comply with European legislation such as Natura 2000 and the Rural Development Regulation.

The development of the area of land falling under agri-environmental schemes where farmers are compensated for wildlife and landscape management has increased in the Netherlands from 16 thousand hectares in 1990 to almost 78 thousand hectares in 2007. The objective is to have 118 thousand hectares of land designated as agricultural nature subject to agri-environmental schemes by 2018 as part of the so-called Dutch Ecological Structure (Ecologische

Hoofdstructuur). This sounds like a lot, but is in fact only between 5 to 10 percent of all agricultural land in the Netherlands. Currently around 65 percent of these 118 thousand hectares have been realized and it is unlikely that the objective will be met in 2018. Farmers are compensated for their wildlife and landscape management under the SAN subsidy: Subsidieregeling Agrarisch Natuurbeheer. Different prices are paid for different activities performed by farmers on their land. When examining the development of these prices in time (Figure 1), the unit prices for the provision of certain ecosystem services have risen considerably since 2007, possibly as a result of the delay in achieving the 2018 objectives and encourage farmer participation.

Figure 1: Development of prices for ecosystem services provision on agricultural land



Source: own adaptation based on LNV (2009).

The compensation price levels are based on actual labor costs and production losses and, depending on the exact activity, a mark-up between zero and twenty percent to encourage farmer participation. Some programs are well-known with the wider public, for instance to protect rare meadow birds like the black-tailed godwit. In some provinces farmers get paid for every nest found on their land.

Agri-environmental agreements may also play an important role in water policy. The agricultural sector is the largest water consumer in the world and also one of the most important emitters of nutrients in surface and ground water. Existing agri-environmental schemes can be used at low transaction costs given their long implementation history and institutional embedding across European Member States to stimulate the reduction of pesticide and nutrient emissions to water. This would provide an important step forwards to reach the environmental objectives laid down in the European Water Framework Directive adopted in 2000. The Water Framework Directive is a huge piece of new water legislation prescribing that all surface and groundwater in European Member States should be in a good state by 2015.

The Water Framework Directive provides an important stepping stone towards payments for water services on the basis of which farmers can, for example, be compensated for their water quality improvement activities on their land. The Directive essentially embraces the co-evolutionary ecosystems approach advocated since the 1960s in academia by scholars like Buzz Holling. A first key characteristic is that the Directive is implemented at river basin level, that is, the hydro-geographical boundaries of the water system, not existing administrative boundaries which usually stop at the border of a country. This poses a real challenge as it increases both the complexity and uncertainty of European water management. Related to this is the emphasis on both water chemistry and ecology, and here too important challenges have surfaced in past years, trying to establish reliable cause-effect relationships in the aquatic ecosystem.

A second key characteristic is that the Water Framework Directive is the first water directive in Europe in which economics plays such an important role and is an integral part of the river basin analysis. A means to an end to reach the environmental objectives of the Directive through the adoption of economic principles such as the polluter pays principle, economic methods like cost-effectiveness analysis (meaning that the objectives have to be reached at lowest cost possible) and economic instruments such as water pricing.

Water pricing is one of the Directive's most important innovations. The WFD refers to pricing in relation to water services in article 9. Adequate water pricing acts as an incentive for the sustainable use of water resources. However, if one looks at the definition of water services in the WFD, the interpretation is very narrow, and mainly refers to drinking water, wastewater collection and treatment. As far as I understand (I could not find any details anywhere) the European Commission has been taken to court over this definition issue by WWF, who also disagrees with the narrow interpretation of water services by most EU Member States in their draft river basin management plans submitted to the European Commission in 2009.

The implication of labeling a specific activity as a water service is that a pricing mechanism has to be in place for the recovery of the costs of the water service provided. In most EU Member States people already pay for drinking water provided by drinking water companies and often also for wastewater collection and treatment performed by public utilities. Maybe not the full costs of the service, including the environmental and resource costs involved, but usually a considerable share of it. Hence, defining water services in a narrow sense as drinking water provision and wastewater treatment is a safe bet of not having to introduce new and perhaps undesirable pricing mechanisms. It is understandable that there is resistance to pricing access to water services which have historically and traditionally always been 'free of charge'. Take water for irrigated agriculture, but also water use in industry and perhaps even navigation and hydropower. However, here tremendous opportunities are found to introduce more market-based incentives to change the way we use our water resources and engage with our water environment.

I am convinced that interpreting water services in a broader sense than currently is the case will be beneficial in the long term to achieve more sustainable levels of water use. Expressing nonmarket ecosystem goods and services in monetary terms – as long as we do this in a sound and robust way – provides an important signal to policymakers that our natural environment has a value and that using it in an unsustainable way comes at a cost. Money speaks louder than words, especially when dealing with companies whose business it is to make money. Introducing financial incentives to change the way they operate and use the environment as if the ecosystem services it provides are free of charge can be an important complementary policy to traditional command and control to change people's behavior. For example, making companies pay for permits issued by the government to pollute and allow some degree of market functioning where these permits can be traded to stimulate more efficient water use. In the Netherlands the government issues water pollution permits since the 1970s for which companies have to pay.

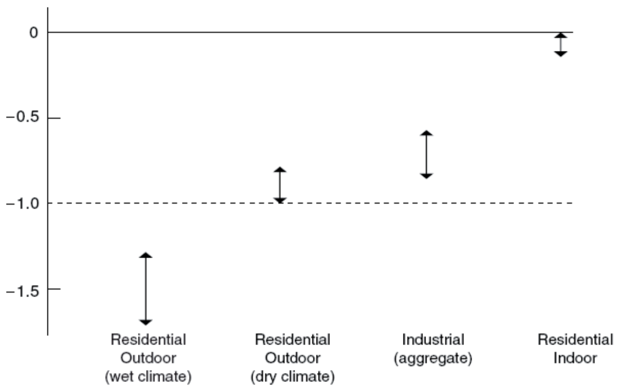
This happens under the Surface Water Pollution Act, now part of the new Dutch Water Law. However, there exists no possibility to trade these permits within the limits imposed by the government. This is what I mean in the title of my presentation with making money talk. Let money do the talking within the environmental boundaries imposed based on safe minimum standards or precautionary principles. With the help of a general equilibrium model, I have been able to show together with other colleagues at IVM that such a market system may have a moderating effect on the costs of implementation of the Water Framework Directive [12].

Water pricing too often is dismissed in policy and decision-making on incorrect grounds in my view, namely that water use is non-responsive to water pricing, an important point made also by John Briscoe in his contribution to the book ‘Cost-Benefit Analysis and Water Resources Management’ which I edited together with the late David Pearce [13]. Water allocation and water pollution rights are very much dominated by traditional technical standards and engineering solutions. We have a much better understanding already of the values underlying water services. One has to be careful to use some of these ball-park estimates to fix price levels, but if you look closer, some remarkably consistent results are found with important policy implications.

First of all, the value of water for many low value crops is universally low. When used for high-value crops, the value of water can be much higher, sometimes in the order of magnitude to the value of water in domestic and industrial uses. The value of water for domestic use is, not surprisingly, always highest, whereas the value for environmental purposes such as environmental flows, maintenance of wetlands, wildlife habitat vary widely, but typically fall in between the agricultural and domestic values.

Similar remarkably consistent results are found when examining available data on price elasticities (Figure 2). Price elasticities tell us how water users react to a change in the price of water, measured as the percentage change in water use for each percentage increase in the price of water. If the elasticity is zero, demand is said to be perfectly inelastic. It does not respond to a change in the water price. If the elasticity is between zero and -1, demand is said to be inelastic. The percentage change in water use is less than the percentage change in the price. Finally, if the elasticity is lower than -1, demand is said to be elastic. The percentage change in water use is higher than the percentage change in the water price.

Figure 2: Range of price elasticities of water demand in the United States



Source: Briscoe (2005).

The use of the word inelastic has caused some confusion as it is interpreted by some that water use is not reduced when the price increases. Residential water use is usually least sensitive to price changes because it is a basic need, followed by industrial water use, where water often is a necessary non-substitutable input in a production process. Take the production of beer, paper or textile. This cannot happen without water. On the other hand, for recreational activities the response to changing park entry fees, prices of hunting or fishing permits are usually more elastic.

The most important conclusion here is first of all that the price elasticity is negative. This means that water users do react to water pricing by reducing their demand if the price increases. It is often a misunderstanding that an inelastic demand for water means that water use is completely unresponsive to changes in water prices. The second important point is that the elasticity is higher if the price is higher. If users are already charged a high price for their water use, they will be much more responsive to an increase in the water price than if they are charged a low price. There is one important sector missing in Figure 2. Agriculture is not included. Price levels in agriculture are typically low and hence studies investigating the elasticity of irrigation water demand therefore usually find, not surprisingly, low elasticities. So, the reason for this low elasticity is usually not that farmers do not respond to prices, but because the response depends on the baseline level of the water price.

5. Future research directions

At the end of my presentation I'd also like to say a few words about future research directions and the ambitions I have with this new chair in the department of environmental economics. I see a number of important new research directions. These are related to the further development of so-called choice experiments, and their role in (1) facilitating improved understanding of the specific goods and services provided by ecosystems, and (2) informing the design of economic instruments like payments for ecosystem services. The increased use of choice experiments will be accompanied by further in-depth testing of economic valuation processes underlying value outcomes. Choice experiments work on the interface between economics and socio-psychology. We need to get a better understanding of choice processes. This will help us to better assess the robustness and reliability of the value outcomes, and hence their acceptability in actual policy and decision-making.

So, a lot of emphasis on choice experiments. We made enormous advancements and gained a lot of experience in the department over the past 5 years with choice experiments. These experiments were introduced about 10 years ago at the end of the 1990s into the environmental valuation domain and have made a significant contribution to the improvement of the validity and reliability of economic valuation. Most importantly to get rid of and/or better account for some of the flaws that troubled the other stated preference method contingent valuation. This includes for example preference construction, scope sensitivity, embedding, and trade-offs between substitutes. In the department of environmental economics I aim to provide a significant contribution to the further development of these choice experiments and the published literature in this area. Choice experiments allow for much more detail in the description of the baseline and policy conditions in environmental valuation. And it is this quality that makes choice experiments especially suitable for the assessment of payments for ecosystem services.

Together with Marije Schaafsma, one of the PhD researchers in the institute over the past 4 years, and Alison Gilbert, one of the ecologists in IVM, I have been working towards an improved natural scientific representation and break-down of individual ecosystem services in choice experiments, focusing especially on the substitutability between individual services provided at specific water locations, such as bathing and nature conservation. We tested this for example in the Scheldt basin, at three specific locations in one and the same confined geographical area. Locals pay for some of the water services already through their water board tax. The environmental objectives of the Water Framework Directive relating to the area's hydro-morphology, chemical and ecological status were translated in specific water services.

The PhD work, which is close to finished, has provided us with useful new insights and building blocks for future research. I want to mention two important findings here. The first one is that water services and their socio-economic values are site-specific. The relevance of this is that this site-specific context has to be accounted for in the valuation research design and when aggregating values from individual water bodies into a total economic value of a river basin. A second important finding is that despite the specific context, adding to the uniqueness of some of these sites and services, both sites and services are substitutable. The choice experiment allowed us to explicitly test the degree of substitution. This is a function of the specific ecosystem services characteristics, but also the spatial distribution of the population of beneficiaries of these services. We show that if the water management authority in the study area, Rijkswaterstaat Zeeland, improves water quality at one site, this has consequences for the economic values of water services provided at other sites. Knowing these substitution patterns, we get a much better understanding of the interrelationships between individual water bodies in a river basin. This in turn allows us to better advise policymakers on how to prioritize public investments in water quality improvements.

Another line of research is closely related to this, namely building spatial value maps, accounting for the factors that influence nonmarket values for ecosystem services. This work was initiated in the department of Environmental Economics by Luke Brander about 2 years ago, first for the European Environment Agency, then AQUAMONEY and now for example also for TEEB, The Economics of Ecosystems and Biodiversity. TEEB is a major international initiative to draw attention to the global economic benefits of biodiversity, funded by the European Commission, UNEP and so on. The GIS work in this research area is carried out by our GIS expert Alfred Wagtendonk. In the example shown in Figure 3 we first translated available European-wide water monitoring data to water quality levels and the different ecosystem services provided at these different levels using meta-analysis. Changing water quality levels affects the values of these ecosystem services. We relate these changes to population characteristics, which we know influence the values attached to water quality changes like income levels, but also relative ecosystem service scarcity and the distance people live away from water. This latter effect is called distance-decay, and is closely related to substitution effects. For specific water uses like recreational fishing or bathing, the further away someone lives from a water site, the less that person might worry about and hence value changes in water quality levels, but also the more substitution possibilities become available. And this then translates in the end into value maps where changes in water quality levels are measured in economic terms. Because this is all done in GIS, one can zoom into a specific area and look for regional or even local value changes.

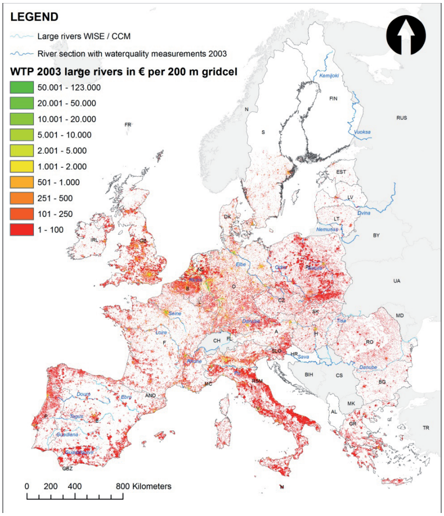


Figure 3: Example of a GIS based value map for water quality changes

Source: Wagtendonk, Brander and Brouwer (2009).

The use of choice experiments to inform the institutional-economic design of Payments for Ecosystem Services schemes is a research direction I developed recently in the research program Poverty Reduction and Environmental Management (PREM) funded by the Dutch Ministry of Foreign Affairs and coordinated by my colleague Pieter van Beukering. In PREM I explored for example the design of micro crop insurance to protect farmers in Bangladesh against the impacts of climate change and increasing flood risks [14]. In a new project in Ethiopia and Sudan I test together with Abonesh Tesfaye and Pieter van der Zaag from UNESCO-IHE how we can add in private interests in public-private partnerships using financial and economic incentives to change farmer behavior to use their land in a more sustainable way. Every year tons of soil erode and wash into the Blue Nile causing enormous constipation downstream in Sudan. The governments in Ethiopia and Sudan have public interests to safeguard the water services provided by the Blue Nile. At the same time farmers have a private interest in maintaining soil quality. Instead of doing what almost every Payment for Ecosystem Service scheme in the world does, that is compensating farmers for less intense land use and set-aside, we aim to introduce private incentives into contractual agreements between government and farmers. Farmers are not compensated financially, but given for example certain land use guarantees, extension services and credit facilities to take soil conservation measures on their land, for which they pay themselves. First field trials using choice experiments to test participation and incentive-compatibility constraints indicate that this seems to work.

Finally, it is an illusion to think that economic instruments like Payments for Ecosystem Services are the holy grail to solve the serious environmental problems we face worldwide. They will have to be part of a wider policy package and embedded in an institutional-economic context. The hypothesis is that economic instruments will be more effective and can be introduced at lower transaction costs when part of a broader package of policy instruments. Together with my colleagues at IVM I will take this hypothesis to the test in the coming years, focusing on payments for ecosystem services in a wider ecosystem governance context.

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