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Making Sure the Rural Poor Benefit

Agricultural Landscapes and Domestic Water Supply:

The Scope for Payments for Ecosystem Services in sub-Saharan Africa

By
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Agricultural Landscapes and Domestic Water Supply: the Scope for Payments for Watershed Services with special reference to sub-Saharan Africa

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Acronyms & Abbreviations

BMP: Best management practices (in agriculture)

CEA: Cost-effectiveness analysis

CBA: Cost-benefit analysis

DEP (or NYC DEP): New York City Department of Environmental Protection

EPA: United States Environmental Protection Agency

IIED: International Institute for Environment and Development

FAD: Filtration Avoidance Determination

M&I: Municipal and industrial (use of water)

NRC: National Research Council

PES: Payments for environmental services

PWS: Payments for watershed services

USGS: U.S. Geological Survey

WS&S: Water supply and sanitation

WHO: World Health Organization

1. Introduction

Natural resource and environmental managers are increasingly recognizing the importance of providing payments that promote the adoption of land management practices, which generate global, regional, or local ecosystem goods and services (henceforth referred to simply as ecosystem services). Pristine natural areas are often perceived to be the primary source of ecosystem services, however the agricultural, forestry, and water use decisions made by commercial and subsistence land managers are of critical importance in determining the overall level of ecosystem services in a given locale. Agricultural producers in rural areas of developing countries, including the poor, are therefore important potential suppliers of these ecosystem services. Channelling payments or other forms of support to these communities in return for the provision of ecosystem services could generate not only benefits in terms of improved natural resource and environmental management but also indirect benefits in the form of increased food security and enhanced well-being.

Watershed services derived from managing hydrological function – the land and water interactions – are an integral component of ecosystem services. These services are provided when upstream land managers steward their land and other resources in such a manner as to provide water quantity or water quality services to downstream water resource users. Payment for watershed services (PWS) schemes have evolved in developed and developing countries as a means to provide incentives to upstream managers to take account of the downstream consequences of their actions. Initial PWS schemes were incorporated in national level land programs for land fallowing, forest conservation and reforestation.

However, the cause and effect of watershed services occurs within the confines of watersheds and at scales that range from a micro-catchment to a large river basin. Gradual recognition that the hydrological function that underpins watershed services are highly variable and specific to the scale and conditions in the watershed – and are therefore not generalizable across watersheds has led to increased effort to develop PWS schemes that respond to watershed conditions. These rely on identification of demand and supply for watershed services. Demand for watershed services mostly originates from downstream water users, and given the local nature of demand and the presence of a limited number of well-organised beneficiaries (e.g., water or hydroelectric utilities, irrigation commissions) it may be possible to mobilize downstream beneficiaries and involve them in PES schemes (low transaction costs). Also, there is an increased willingness on the part of beneficiaries to pay for services, as awareness grows as to the importance of conservation in upper watersheds for the maintenance of downstream watershed services.

PWS schemes largely evolved out of efforts to find sustainable financing for forests and protected areas. Until now, much work has been done on the potential supply of watershed services from these ecosystems, but much less is known about the potential supply of watershed services from agricultural landscapes. While generally little is known about the current and potential capacity and willingness of beneficiaries to pay for watershed services, this is particularly true at the local level where upstream farmers may be from the same community or neighbouring communities to those that use downstream water resources. To date most ongoing PWS schemes are found in North America, Europe and Latin America. Questions therefore arise as to how applicable these schemes may be to rural areas in lower income developing countries, such as in sub-Saharan Africa.

Nevertheless, one potentially important source of demand for watershed services from agricultural landscapes is their capacity to reduce the costs and improve the effectiveness of efforts to meet economic development and social objectives. This includes improving the quality of water for household/domestic use and the health of low income populations as well as the income of poor farmers. With ambitious

Millennium Development Goals in place it is relevant to ask whether PWS schemes have a role to play in the context of sub-Saharan Africa.

The purpose of this paper is to assess the state of knowledge on these issues and to provide an initial assessment of the potential for PES in agricultural landscapes to be an important means of addressing water quality issues in developing countries–and most particularly in sub-Saharan Africa. The paper is organized in two Parts as described below.

Part 1: Employing the wider literature, but drawing special attention to studies and evidence from developing countries the paper (in Sections 1 through 5):

1. Summarizes the state of knowledge on the relationship between land use, land management and environmental management in agricultural landscapes, and the quality of water for household/domestic use (from surface and ground water sources).

2. Summarizes studies on the social and economic costs associated with poor quality of water for household/domestic use (e.g. health impacts).

3. Summarizes available assessments of the type, magnitude and cost of the changes in environmental management in agricultural landscapes needed to improve the quality of water for household/domestic use towards established standards.

4. Summarizes available assessments of the types of technology that are employed to treat water for household/domestic use and identify the per unit costs of such technologies.

5. Summarizes experience and cases where PES have been employed to remedy quality problems related to water for household/domestic use, particularly with respect to water quality impairment from agricultural landscapes.

Part 2: With respect specifically to sub-Saharan Africa the paper (in Section 6):

1. Identifies current strategies to improve quality of water for household/domestic use and estimate spending to meet water quality objectives, including both national public sector spending as well as overseas development assistance.

2. Identifies the potential role in improving quality of water for household/domestic use for improved ecosystem management in agricultural landscapes and assess the potential of environmental services to reduce current costs society is bearing regarding the provision of clean water for household/domestic use, including assessment of potential willingness to pay for these services and the costs of alternatives

3. Identifies the potential for developing innovative institutional arrangements and financial mechanisms for developing payment programs for water services from agricultural landscapes.

4. Assesses potential legal and regulatory barriers to the establishment of payment programs for water services from agricultural landscapes and identify likely solutions.

2. Agricultural Landscapes, Water Quality and Domestic Water Supply

Water quality is defined as the physical, chemical, biological characteristics of a body of water, typically with respect to its suitability for some purpose (USGS 2001). When water quality is degraded this is referred to generally as water pollution. This section examines the biogeochemical aspect of water pollution by summarizing the state of knowledge on the relationship between land, resource and environmental management in agricultural landscapes, and the quality of downstream water for household and domestic use. This analysis is undertaken in recognition of a number of factors including:

- that water for domestic use may come from surface or ground water sources.
- that the agricultural landscape includes land areas that may not be in an agricultural land use
- that the agricultural landscape may include communities and industrial activities that themselves are sources of water pollution
- that water pollution can generally be divided into point and non-point sources.

2.1 Water Quality and Pollutants in Agricultural Landscapes

All water bodies have a naturally occurring level of water quality. This water quality will vary with the season and with flow levels. As water quality is often defined in terms of the concentration of different dissolved substances water quantity will affect water quality. Naturally-occurring dissolved solids include common elements such as calcium and sodium, plant nutrients such as nitrogen and phosphorus, and trace elements that include selenium and arsenic. A number of hydrological, physical, chemical and biological processes will affect the nature and level of chemical elements and compounds in water bodies (Meybeck et al. 1996). These are summarized in Table 1.

Table 1. Processes Affecting Water Quality

Category	Process	Occurrence
Hydrological	Dilution	All water bodies
	Evaporation	Surface Waters
	Percolation and leaching	Groundwater
	Suspension and settling	Surface Waters
Physical	Gas exchange with atmosphere	Mostly rivers and lakes
	Volatilisation	Mostly rivers and lakes
	Heating and cooling	All water bodies
	Diffusion	
Chemical	Photodegradation	
	Acid base reactions	All water bodies
	Redox reactions	All water bodies
	Dissolution of particles	All water bodies
	Precipitation of minerals	All water bodies
	Ionic Exchange	Groundwater
Biological	Primary Production	Surface waters
	Microbial die-off and growth	All water bodies
	Decomposition of organic matter	Mostly rivers and lakes
	Bioaccumulation	Mostly rivers and lakes
	Biomagnification	Mostly rivers and lakes

Source: Meybeck et al. 1996

Anthropogenic impacts on water quality may improve or degrade this quality through altering these processes, thought typically the impact of land use change and development is to degrade this quality relative to human and ecosystem uses of water. Pollutants can be classified according to their impact as physical, organic or toxic pollutants. Physical pollutants include those that alter the physical characteristics of a water body such as discharge of heated water, increases in dissolved solids, and soil erosion and mass wasting that lead to sediment deposition. Physical pollution is typically measured as increase in temperature, turbidity and sedimentation. Organic pollution affects aquatic ecosystems and consists of the discharge or leaching of a large number of organic wastes and compounds. Typically water bodies have a certain natural resilience to these pollutants, but as pollutant concentrations increase water quality and ecological function will be affected. Typical indicators of organic pollution include levels of oxygenation, eutrophication and acidity. Toxic pollutants are more directly of concern as they directly affect, or poison, humans, animals and other species. Toxics typically include chemicals and radiological materials.

Note that the above definition of pollutants does not consider the insertion of large infrastructure, such as dams and levees, the channelisation of waterways or the abstraction of water as pollutants. This said, these human activities can have significant impacts on water quality through altering hydrological processes that govern natural water quality. For example, the abstraction of water from a water body for human use leads to a reduction in water quantity and will make the water body more susceptible to increases in temperature and, other things equal, will raise the concentration of other pollutants. In a similar fashion other efforts to use and store water quantity for hydropower, irrigation, municipal use, flood control and transport will have an impact on water quality. It is important to be cognizant of the full range of sources of poor water quality, so that the design of a particular solution can accurately predict the resulting change in water quality.

As described above there are many types of water pollutants and these pollutants may originate from a number of human activities (Fulton Forthcoming). Table 2 provides a brief summary of the types of pollutants, examples of their constituents and their origins. Typically pollutants are described as being point or non-point sources, referring to whether or not they have a specific spatial origin in terms of their discharge or whether they come, in effect, from across the landscape and therefore have many and diffuse source 'points.' Point sources typically include effluent from industrial and municipal plants, whereas point sources typically include the run-off and leaching of by-products from different land uses.

Table 2. Water Pollutants

Type of pollutants	Pollutants	Origins
Physical	Sediment, Temperature, Turbidity	land surface erosion litter and mismanaged solid waste organic matter runoff from buildings or construction sites diversion of flow from rivers storage of water behind dams
Organic	Nitrogen, Phosphorus, Microbes, Bacteria	organic matter fertilizers sewer overflow detergents animal and human wastes
Toxic	Chlorinated compounds Solvents, Acids, Alkalis, Heavy Metals, Pesticides and Oil	pesticides herbicides runoff from buildings and roads (oil) detergents

Source: Fulton (Forthcoming)

2.2 Water Quality Risks to Domestic Water Supply in Agricultural Landscapes

This study focuses on rural sources of pollution in agricultural landscapes including non-point sources such as sediment, nutrient runoff and leaching, pesticides, and livestock wastes. However, as mentioned earlier industrial and municipal point sources may also inhabit the agricultural landscape. For example, agricultural processing or mineral processing may take place in agricultural landscapes and therefore be an important component of downstream water quality problems. Similarly, household waste from one agricultural community may affect another community just downstream. Therefore any analysis of watershed services in agricultural landscapes must be cognizant of both point and non-point sources. Market-based mechanisms such as PWS, may target land management and non-point sources, but may serve equally well in dealing with point sources. However, it can be argued that it may be more feasible and less costly to tackle significant point source polluters prior to taking on the non-point challenge. This is discussed further below.

Erosion, Suspended Sediment and Sedimentation. The concentration of particulate matter in surface waters due to agricultural activities is largely due to land clearing and tillage which in turn raises erosion rates above background levels (see Table 6). Erosion comes from not just farming practices per se but also from the establishment of associated infrastructure and, in particular, from road-building. Gully, rill or sheet erosion leads to the deposition of sediment material into waterways. Depending on the physical and chemical properties of the materials involved and the type of waterway this material will persist as suspended sediment for some distance downstream. Turbidity is a common measure of the level of suspended sediment. Once the sediment settles it is deposited on the bottom of the waterway as sedimentation. Suspended sediment may negatively affect the appearance and taste of water and can harm industrial processes and equipment. Sedimentation may impose additional costs on domestic water providers due to the impairment of storage facilities, offtake points and distribution systems.

Nutrients. The concentration of nutrients due to the application of fertilizers and manure is one of the major water quality problems associated with agriculture. The primary inorganic nitrates are potassium and ammonium nitrates used in fertilizers. Organic nitrogen in the form of nitrogen and ammonia coming from human sewage and manure is converted to nitrate once it enters waterways. In developed countries with high fertilizer application rates, agricultural areas may have high levels of nitrate concentrations in streams and shallow groundwater. This is particularly true where soils are artificially drained in order to improve crop production (USGS 2007).

Nitrates are relatively non-toxic but are converted to nitrites by bacteria in the environment and human bodies. High nitrate concentrations lead to eutrophication of surface waters, meaning excessive nutrient levels cause rapid plant growth, which in turn depletes oxygen levels in the water (BOD). Lower oxygen levels can lead to die offs of phytoplankton, the decomposition of which further consumes available oxygen. Changes in BOD lead to illness, disease and death of aquatic species; and consequent impacts on drinking water and water supply infrastructure. Nitrate in human drinking water can lead to serious illness and even death. Nitrate is converted to nitrite in the human body, and nitrite can interfere with the absorption of oxygen in the blood. Children are particularly susceptible to this condition, called methaemoglobinemia or 'Blue Baby Syndrome' after the shortness of breath and blue colour of the skin when affected in this manner. Long-term exposure to excessive nitrate levels can cause diuresis, increased starchy deposits and haemorrhaging of the spleen.

Nitrate pollution is commonly thought of as a pressing water quality concern in Europe and North America and the United Nations predicts that it will become a serious problem in other countries, such as India and Brazil, if trends continue (UNDSD 1999). Nevertheless, a review of the evidence undertaken for the 2nd World Water Development Report sounds a note of caution with respect to the health impacts of nitrate pollution (Fewtrell 2004). The Report subsequently recommended against linking the incidence of Blue Baby Syndrome to drinking water nitrate levels stating that the low incidence of the disease and multiple causal agents make this inappropriate at this time (WHO and UNICEF 2006).

Pesticides and Medicines. Pesticides and other chemicals are used on farms to retard or kill plant and animal species that compete with or would damage crops and livestock. These compounds find their way into soils, streams and groundwater. As demonstrated by the case of DDT, a danger with these compounds is that they persist in the environment and bio-accumulate as they move up the food chain with negative consequences for fish, wildlife and humans. An example of how such chemical compounds find their way into the hydrological system and their impact on human health is illustrated by the example of atrazine presented in Box 1. Beyond pesticides a number of potentially harmful chemicals including antibiotics and hormones that enter the hydrologic system come from veterinary medicine used as part of livestock raising. A number of environmental factors determine the persistence and concentration of these compounds including temperature, PH and soil type. As a result the degree to which these compounds degrade quickly or persist through to downstream waters varies with local conditions. Although these compounds are currently being detected in developed country waterways at levels an order of magnitude lower than acute toxicity levels these emerging contaminants remain re a cause of concern as the toxic effects of these chemicals are not yet well understood. While these are being detected in developed countries, this is due to higher effort expended on monitoring and not that these may not be increasingly a problem in developing countries.

Box 1. Atrazine and Human Health

Atrazine is used by farmers in the US and other countries to eliminate broad-leaf weeds that compete with a number of crops. In the United States annual application of atrazine is equivalent to 76 million pounds. Atrazine is the most commonly detected pesticide in US streams and groundwater. The presence of atrazine in shallow groundwater is determined by the rates at which it is applied and the proportion of land in agriculture as well as a number of physical parameters including soil infiltration rates, presence and rate of human-caused drainage, water holding capacity of soils, soil permeability and groundwater irrigation. Atrazine persists in soils and is moderately soluble in water and, therefore, easily finds its way into shallow groundwater and stream flow. Atrazine adversely affects the reproductive system in humans and other animals through its impact on the endocrine system.

Source: USGS (2007)

Pathogens. Bacteria, viruses and protozoa are three groups of pathogens or microorganisms that cause waterborne disease. Bacteria are widely distributed and commonly known diseases caused by bacterial pathogens include Salmonella, Shigella, Typhoid fever, and Cholera. Fecal coliform bacteria (including *E. Coli*) enter streams and groundwater from human, pet, wildlife and livestock sources. In agricultural landscapes, point source discharge from feedlots, and hog and poultry farms can be of particular concern. Viruses are protected in water by protein coats and are active only upon finding a living host cell. Principal viral diseases transmitted through water include hepatitis A and Norwalk virus. Protozoa are far larger than bacteria and viral pathogens and often form protective cysts while awaiting a host. Key waterborne diseases caused by protozoa include Giardia, Amebiasis and Cryptosporidosis. Typical symptoms from these diseases include fever, abdominal pain, diarrhoea and vomiting. If not treated these symptoms can lead to long-term incapacitation and even death

2.3 Water Needs and Quality Standards

Domestic water requirements can be defined as those required for consumption and hygiene. Additional domestic uses – such as for lawn irrigation – are not ‘requirements’ and thus are omitted here. WHO suggests that as consumption and hygiene requirements are met at levels from 5 liters/capita/day through to 100 l/c/d that the level of health concern goes from very high to very low (Howard and Bartram 2003). In order to achieve a very low level of health concern then the per capita water requirement would be 36.5 m^3. Losses resulting from leaks in the distribution system would mean a proportionately higher total amount of water would be required at the source.

Table 3. Per capita requirements for water service level to promote health

Service level	Access measure	*Needs met*	Level of health concern
No access (quantity collected often below 5 l/c/d)	More than 1000m or 30 minutes total collection time	Consumption – cannot be assured Hygiene – not possible (unless practised at source)	Very high
Basic access (average quantity unlikely to exceed 20 l/c/d)	Between 100 and 1000m or 5 to 30 minutes total collection time	Consumption – should be assured Hygiene – handwashing and basic food hygiene possible; laundry/ bathing difficult to assure unless carried out at source	High
Intermediate access (average quantity about 50 l/c/d)	Water delivered through one tap on- plot (or within 100m or 5 minutes total collection time	Consumption – assured Hygiene – all basic personal and food hygiene assured; laundry and bathing should also be assured	Low
Optimal access (average quantity 100 l/c/d and above)	Water supplied through multiple taps continuously	Consumption – all needs met Hygiene – all needs should be met	Very low

Source: Howard and Bartram (2003)

The usefulness of domestic water supply is not just a matter of quantity but also of quality. Water quality is generally measured in terms of levels or concentrations of different physical, chemical or biological constituents of water (Bartram and Ballance 1996). In managing water quality it is typical to define a water quality standard. Standards may be set for water bodies or for effluent discharged into such bodies. Standards will typically set the criteria (numeric pollutant concentration) for a water body with regard to the different uses for water. The level of water quality necessary to ensure safety for irrigation water would differ from that for swimming as versus drinking, for example. It is not so much that the prescribed standard is set at a threshold between a level that is unsafe and one that is safe, but rather the standards are typically set at a level which – based on available science – seems to guarantee a very high level of safety.

In the case of the US EPA and the Clean Drinking Water Act, for example the US EPA is charged to choose a Maximum Contaminant Level Goal, which is the ‘maximum level at which no known or anticipated adverse effect on the health of persons would occur and that allows an adequate margin of safety’ (US EPA 2007). Examples of these maximum levels for a number of the contaminants discussed in this paper as prescribed by the US EPA, the European Union and WHO are provided in Table 4. Note that there is some variation between the agencies, particularly as to whether specific numeric targets are set. Reflecting the varying capabilities of countries from around the world, WHO tends to discuss methods for setting targets rather than specifying hard and fast targets per se.

Table 4. Drinking Water Quality Standards for Agricultural Contaminants

Contaminant	Health Effects	USEPA – MCL or TT*	US EPA - Public Health Goal	WHO	EU
Atrazine	Cardiovascular system or reproductive problems	0.003 mg//	0.003	0.002	(.0001 for pesticides)
Cryptosporidium	Gastrointestinal illness	TT: 99% removal	zero	n/a	n/a
Giardia lambia	Gastrointestinal illness	TT: 99.9% removal / inactivation	zero	n/a	n/a
Heterotrophic Plate Count	Indicative of presence of a variety of bacteria	TT: 500 bacterial colonies/ml	n/a	n/a	100 colonies/ml
Nitrate (measured as nitrogen)	Illness and death for babies less than 6 months	10 mg/l (as nitrogen)	10	n/a	50 mg/l (as NO3)
Total Coliforms (including *E. coli*)	Indicates if other health threats present (WHO-not the best indicator)	TT: no more than 5.0% samples in a month	zero	Not detectable in 100 ml	Not detectable in 100 ml (E. coli in 250 ml)
Turbidity	Indicative of presence of helath threats including viruses, parasites and some bacteria	TT: 5 NTUs	n/a	n/a	Acceptable to consumers and no abnormal change
Viruses (enteric)	Gastrointestinal illness	TT: 99.9% removal	n/a	n/a	Not detectable in 250 ml

Notes: *TT are treatment techniques that agencies are required to use to reduce the level of contaminants in drinking water. Atrazine is an example of an organic chemical generated through agricultural applications, EPA lists another 15 or so herbicides, pesticides and fumigants. WHO does not specify limits for chemical contaminants (Lenntech 2007).

Sources: US EPA(2007), WHO (2006), Lenntech (2007)

Related to water quality standard is the level of contaminant that a water body may receive before it threatens to exceeds the criteria for a given contaminant. In the US the term total maximum daily load (TMDL) is used to reflect this calculation. Water quality planning then typically occurs locally and will be focussed on defining the TMDL that a water body can sustain, given the absorptive and processing capacity of the water body and the uses to which it is put. One the TMDL is specified it can be compared to existing levels of pollution and appropriate course of action

2.4 Water Quality and Human Health

Global data on disease and other health risks does not necessarily disaggregate that portion of health impacts originating from poor water quality. The 2nd World Water Development Report lists a number of diseases related to water and sanitation including diarrhoeal disease, malaria, schistosomiasis, lymphatic filiarieis, onchocerciasis, dengue, and japanese encephalities (WHO and UNICEF 2006). These diseases include vector-borne diseases where water provides habitat for, or the place of contact with, pathogens. Poor water quality in the environment may be a factor in the incidence of these diseases but the disease itself is not communicated through domestic water supply systems. Water-borne and water-washed diseases, on the other hand, are diseases transmitted through direct human contact with the disease agent where water and water quality is either an infection route or a potential solution to a hygiene or sanitation problem. Diarrhoeal disease, in particular, stands out as a disease that can be caused by poor quality of domestic water supply. WHO and UNICEF (2006: 210) classify diarrhoeal disease as 'related to lack of access to safe drinking water, poor sanitation and insufficient hygiene.' Poor domestic water quality is, therefore, just one of a number of contributing factors to these water-borne diseases.

The impacts on health of disease are often expressed in terms of mortality and morbidity, the latter reflecting the disease burden on surviving members of a population. Table 5 presents annual mortality and morbidity (expressed in DALYs) for diarrhoeal diseases. Almost 2 million people per year are estimated to die of diarrhoeal disease each year and 90% of these are less than 5 years old. The DALY (Disability-Adjusted Life Year) is a combined measure of mortality and morbidity. It discounts and sums the years of healthy life lost to premature mortality and the morbidity associated with the incidence of disease (in that year). The total for diarrhoeal disease is almost 62 million DALYs per year. This is a very significant number for one disease, representing 4% of the total global disease burden (Hutton and Haller 2004). In the case of diarrhoea the prevalence of the disease is primarily a developing country phenomenon with some 40% of DALYs and mortality occurring in sub-Saharan Africa and another one-third of the total in South-East Asia (including India).

Table 5. Global Health Impact of Diarrhoeal Disease

Health Impacts	Total Extent	By Age 0-4 yrs	Distribution by Region SS Africa	S-East Asia (incl. India)	W. Pacifc (incl. China)	E. Medit	Americas	Europe
	('000)	(%)	(%)	(%)	(%)	(%)	(%)	(%)
Mortality	1,798	90	39	34	9	14	3	1
DALYs	61,966	91	38	33	11	14	4	1

Source: WHO and UNICEF (2006)

While virus cause more cases of diarrhoea, dysentery caused by *Shigella* is the primary cause of mortality from diarrhoeal disease. Experts estimate upwards of 160 million cases of Shigellosis annually causing 1.1 million deaths (WHO and UNICEF 2006). Practically all of these occur in developing countries. The risk of Shigellosis can be greatly reduced by such simple hygiene measures as hand washing after defecation. This most severe of water-related health impacts is therefore only indirectly related to water quality management in agriculture insofar as the main cause of infection is hand to mouth and the disease is spread through poor sanitation and hygiene, as much as poor water quality per se.

Amoebiasis is the number two cause of death from waterborne illness. Caused by infection from a protozoan parasite (*Entamoeba histolytica*) amoebiasis leads to the destruction of the intestinal mucosa and potentially damage to other organs including the liver. For patients infected with HIV-AIDS and without access to retrovirals, amoebiasis represents a significant cause of morbidity and mortality. More recently *Crypotosporidium parvum* has been identified as source of water-borne diseases particularly in developed countries. These protozoans are spread through cysts that persist in the environment, including human bodies. Of particular concern to health authorities is that these cysts are resistant to chlorine as commonly employed as a disinfection agent as part of water treatment. Cryptosporidosis is a growing and major health threat. It is delivered through water supply systems and can therefore be linked to environmental sources, particularly wildlife and livestock, in watersheds and agricultural landscapes (National Research Council 2000).

Additional diarrhoeal disease that are a product of poor water quality and sanitation include in developing countries, cholera, and in both developing and developed countries *E. Coli*, Hepatitis E and *Legionalla pneumpophila*. Another widespread disease associated with water quality is Typhoid Fever with an incidence of 21.6 million globally and 216,000 deaths in 2000 (WHO and UNICEF 2006). Infectious skin and eye disease (e.g. Trachoma, and intestinal helminth infections (e.g. roundworm) are other water-related diseases, but ones that are largely related to access to water for hygiene and sanitation.

As with all of the diseases mentioned above proper hygiene and sanitation are important strategies for limiting reinfection and the cycle of the disease. However, proper treatment of community and municipal water systems to ensure the elimination of infectious agents is also a critical component for a number of these diseases. It is, however, difficult to assess to what degree management of agricultural landscapes contributes to these problems – given that in many cases the incidence and prevalance of disease is a result of poor hygiene and sanitation in the urban environment. As far as vector-borne diseases are concerned their mode of infection is not through domestic water systems. However, they are often linked to agricultural activity. For example, 800 million people are at risk of malaria due to their proximity to irrigation schemes (WHO and UNICEF 2006). Thus, as environmental management activities are considered with regard to public health the linkages to vector-borne disease may be worth inclusion – even if urban domestic water system users would not be the beneficiaries.

A final category of waterborne illness is long-term exposure to inorganic chemicals, particularly in groundwater. As the use of small hand pumps increases access to groundwater for communities the risk of exposure is increased. The case of high arsenic levels in groundwater in Bangladesh and the increased incidence of skin lesions and cancers is a particularly acute example. The difficulty is that the move to groundwater was in part motivated by an interest in reducing dependence on untreated surface water and resultant diarrhoeal disease. Excess consumption of fluoride is another example of how excessive inorganic chemical concentrations in groundwater can lead to illness and health problems. These examples underscore the need to more carefully inspect and monitor groundwater quality prior to bringing in tubewell technology (WHO and UNICEF 2006). This adds to the costs of such programs but ensures that a bad situation is not made worse.

Similarly increased use of organic pollutants such as nitrates and phosphorus may have negative impacts on water supply systems. This may occur either directly through high concentrations in source waters (surface or ground water) or indirectly through discharge to surface water bodies leading to nutrient loading and subsequent eutrophication. Eutrophic water, low levels of oxygen and subsequent accumulation and die-off of algae and other plant matter can clog water intakes and overwhelm water treatment systems.

For the purposes of this paper though the key issue is whether such poor water quality relates back to agricultural activity or whether these are naturally-occurring concentrations. Assessing changes in levels of organic and inorganic pollutants due to agricultural point and non-point sources is therefore critical. Also, as alluded to earlier, agricultural activity, can contribute to these problems by removing water from the system (through irrigation) that might otherwise have diluted contaminants to acceptable levels. This may even occur as a byproduct of constructive efforts by farmers to increase water use efficiency through on-farm mechanisms (such as use of drip or sprinkler irrigation) or through lining and piping canals the recharge of local groundwater reserves is affected. Where communities rely on this seepage as a source of groundwater for domestic uses these efficiency improvements may pose problems for small-scale community water supply in developing and developed countries alike.

It is also critical to identify that poor water quality is significant enough to actually lead to poor health and subsequent morbidity and mortality. For example, the 2nd World Water Report states that agricultural pesticides are generally either not found in drinking water or occur at concentrations well below those that may cause toxic effects. As noted earlier the Report also questions the linkage between nitrates and Blue Baby Syndrome. These statements differ somewhat from the emphasis placed on these compounds in water quality management on the part of major developed country agencies (for example USGS and USEPA in the United States).

3. Economics of Water Supply, Water Quality and Health

Having identified the linkages between agricultural landscapes and water quality problems the paper turns to the social and economic costs associated with poor quality of water for household/domestic use (e.g. health impacts). A typology of impacts is presented here and followed in the next sub-section by a brief review of social and economic measures of these impacts.

3.1 Typology of Impacts

A change in hydrological function occasioned by alteration of land use or land management practices will lead to changes in the downstream hydrological outputs associated with a given land unit (Bruijnzeel 2004; Aylward 2004). Table 6 summarizes the impact of different agricultural activities on surface and ground water quality.

Table 6. Water Quality Impacts of Agricultural Activity

Agricultural Activity	Surface Water	Groundwater
Land clearing	Erosion of land, leading to high levels of turbidity in rivers, siltation of bottom habitat, etc. Disruption and change of hydrologic regime, groundwater recharge and transpiration effects alongside reduced evaporation typically increase annual surface runoff; Effects on flow during dry periods depends on balance between ET and infiltration, if lowered then concentrates nutrients and contaminants in surface water . Erosion and flow changes can causes public health problems due to loss of potable water.	Decreased groundwater recharge; Typically less transpiration of soil moisture.
Tillage/ploughing	Sediment/turbidity: sediments carry phosphorus and pesticides adsorbed to sediment particles; siltation of river beds and loss of habitat, spawning ground, etc.	
Fertilizing	Runoff of nutrients, especially phosphorus, leading to eutrophication causing taste and odour in public water supply, excess algae growth leading to deoxygenation of water and fish kills.	Leaching of nitrate to groundwater; excessive levels are a threat to public health.
Weed and pest control	Runoff of pesticides leads to contamination of surface water and biota; dysfunction of ecological system in surface waters by loss of top predators due to growth inhibition and reproductive failure; public health impacts from eating contaminated fish. Pesticides are carried as dust by wind over very long distances and contaminate aquatic systems 1000s of miles away (e.g. tropical/subtropical pesticides found in Arctic mammals).	Some pesticides may leach into groundwater causing human health problems from contaminated wells.
Livestock feed and disease control	Spillage or excretion of animal feed or active, undigested veterinary medicines may enter runoff contributing to eutrophication and or toxicity of surface water	Nutrients and medicines may leach through to groundwater
Manure spreading	Carried out naturally or as a fertilizer activity; spreading on frozen ground results in high levels of contamination of receiving waters by pathogens, metals, phosphorus and nitrogen leading to eutrophication and potential contamination.	Contamination of ground-water, especially by nitrogen
Feedlots/animal corrals	Contamination of surface water with many pathogens (bacteria, viruses, etc.) leading to chronic public health problems. Also contamination by metals contained in urine and faeces.	Potential leaching of nitrogen, metals, etc. to groundwater.
Irrigation	Runoff of salts leading to salinisation of surface waters; runoff of fertilizers and pesticides to surface waters with ecological damage, bioaccumulation in edible fish species, etc. High levels of trace elements such as selenium can occur with serious ecological damage and potential human health impacts.	Enrichment of groundwater with salts, nutrients (especially nitrate). Prevention of seepage through piping/lining of canals can reduce community access to clean water
Aquaculture	Release of pesticides (e.g. TBT1) and high levels of nutrients to surface water and groundwater through feed and faeces, leading to serious eutrophication.	

Source: Based on Ongley (1996), USGS (2007) and Bruijnzeel (2004)

The water in rivers, streams and other water bodies has the following attributes or constituents:

- physical: the volume and rate of stream flow over a given time period and the level of associated sediment and temperature
- chemical: the nutrients, pesticides and medicines.
- biological: bacteria, viral and protozoan pathogens

The spatial and temporal point at which these outputs are evaluated will depend on the type and location of the affected economic activity.

In general, there are three ways that these hydrological outputs (taken separately, as a group or as a whole) enter into human welfare or economic utility - the economist's measure of well-being (Aylward 2004).

- Outputs may enter directly into individual utility, for example if the degree of suspended sediment in surface waters affects the aesthetic pleasure derived by a recreationalist from sightseeing or hiking.
- Outputs may be an input into the household production of utility-yielding goods and services, for example if poor quality of water withdrawn from a stream affects the health of people in the household.
- Outputs may serve as a factor input or impose additional costs on the production of a marketed good that in turn enters into the household production function or individual utility, for example if sediment clogs a water intake used by a municipal water supplier.

The two typologies presented above can be plotted against each other to identify how each water quality problem affects the economy. Again the focus here is solely on agricultural landscapes and domestic water supply. Domestic water supply can take place at different scales from a household taking water directly from a stream or well, through to a full distribution system operated for municipal and industrial (M&I) purposes. At small scales the impacts will be felt generally through the household or impacts on the community water system. At larger scales direct impacts will be on the production side of M&I water systems. Indirect impacts on individual utility will be largely confined to those that result from water quantity and quality at the point of final consumption. Following a discussion of the likely major categories of economic impact a summary table identifying the intersection points is provided (see Table 7).

The discussion begins with physical constituents of water quality. Although the focus of the paper is water quality, water quantity merits mention as the two are so interrelated. It is important to be cognizant of any changes agricultural land use will have on water quantity. Changes in water quantity may come from changing land use or from modifications to water resource management occurring as a result of agricultural production. Water storage and abstraction for irrigation are important factors that may change water availability downstream. In the extreme case irrigation reduces stream flow to the point where downstream communities are short of water during dry periods or, as population and demand, expands water is limited throughout the year. The economic impacts of water quantity being a limiting factor are water shortages, which can lead to a series of human health impacts as people turn to other, lower quality sources or simply do with less water; with consequent impacts for hygiene and health.

The impact of suspended sediment on household utility is likewise indirectly felt through municipal systems. Suspended sediment may affect colour and taste of water but is unlikely to have major implications on its own, rather it may indicate larger problems as removal of sediment is typically a first order treatment level for domestic water supply systems. As suspended sediment levels change so may the time and effort required in treating water, leading to increasing expenditures on maintaining water quality at the household or water supply system level. Also if suspended sediment is not eliminated additional equipment capital costs and operations and maintenance (O&M) costs may be incurred as the sediment moves through and settles in the system. Another physical impact would be the accumulation of sedimentation in water storage reservoirs, settling ponds or other intake facilities. Again, the economic implications are a loss of productive capacity or increased expenditure on O&M to avoid this loss of capacity.

Economic impacts from worsening water chemistry are numerous. Left unaddressed pollution can lead to illness and death. Morbidity and mortality lead to the loss of individual utility in terms of lost consumption and at the household level a loss of household income. Nutrient pollution can also lead indirectly to production impacts as, for example, nitrate-caused eutrophication of water bodies may reduce water quality at the intake for municipal water systems. Fish and algae kills resulting from low oxygenation may lead to increased risk from biological pathogens. Again there is a water treatment option in treating the poor quality water – and worsening pollution presumably leads to accelerating costs of treatment.

The economic impacts of biological pathogens are felt in one of two ways. Either the household or the municipal supplier will need to increase water treatment costs to eliminate the risk or these pathogens will be distributed through the system. Resulting illness and death will lead to economic loss for the individual, the household and society.

The summary presented in Table 7 suggests that the two most important agricultural water pollution impacts on economic utility associated with domestic water supply will be health impacts on the individual and household, or production impacts on water treatment (whether by households, community systems or municipal suppliers). The economic consequences of these impacts are explored further in the next sub-sections. Ancillary impacts include impacts on the satisfaction derived by consumers from drinking water and indirect ecosystem impacts, where the first round impacts from water pollution on ecosystems end up causing illness/death/water treatment issues. The impacts on water quantity may also be significant, however, it needs to be recognized that generally M&I water supply needs are quite small compared to irrigation needs (in arid areas). It is therefore likely that the solution here is to reallocate water from low value agricultural use to higher value M&I use (thereby resolving the problem) than to undertake improved environmental practices.

Table 7. Relationship between Types of Water Pollution and Economic Utility

	Individual Utility	Household Utility	Production
Physical			
	Suspended Sediment – impairs color, taste and odor of drinking water, leads to a reduction of satisfaction in consumption	Suspended Sediment – leads to increasing water treatment costs or cause reduction in satisfaction in consumption	Water quantity – shortfall leads to investment in new supply or conservation and/or a loss of individual utility, household income or economic production. Shortfall may also exacerbate water quality problems.
	Temperature – impairs taste		Temperature – may lower utility of industrial uses of treated water as heat sink.
			Suspended Sediment – water treatment costs or equipment costs
			Sedimentation – loss of water storage (reservoirs), clogging of intakes and pipes and equipment
			Indirect effects – transporting chemical pollutants or biological pathogens
Chemical			
	Nitrates and pesticides – cause illness and death, leading to a reduction or loss of consumption	Nitrates and pesticides – lead to increase in water treatment costs or cause illness and death, leading to loss of household income	Nitrates and pesticides – lead to increase in water treatment costs (or cause illness and death if untreated)
			Indirect effects – nutrient pollution, eutrophication and low BOD may lead to an increase in biological pathogens and/or resulting water treatment costs.
Biological			
	Pathogens - cause illness and death, leading to a reduction or loss of consumption	Pathogens - lead to increasing water treatment costs or cause illness and death, leading to loss of household income	Pathogens – lead to increase in water treatment costs (or cause illness and death if untreated)

It is important to reiterate that the intent is to isolate the economic consequences of poor water quality resulting from agricultural landscapes. This eliminates from consideration water quality problems from other landscapes and ecosystems. It also eliminates from consideration the impact of poor water quality from agricultural landscape on other economic activities and resources such as hydropower, irrigation, ecosystems and flood control. It also largely (see above) excludes consideration of water quantity impacts. However, this winnowing is necessary if intelligent choices are to be made with regard to whether and how domestic water supply might usefully contribute to better environmental management of these landscapes. Ideally, the full set of services provided by environmental improvements will be taken into account in designing economic incentives that lead landowners and others operating in the agricultural landscape to improve uses and practices. However, this begs the first question of whether this is an important question to consider and how to better tease out and identify the biogeochemical and economic relationships involved.

3.2 Economics of Agricultural Water Pollution and Health Impacts

Assessing the economics of agricultural practices, the resulting water quality and health impacts from waterborne disease provides important information in identifying and designing responses at the policy, incentives and project level. Neoclassical economics suggests that the social objective should be to minimize the sum of the costs of efforts to abate pollution *and* the damages caused by the remaining pollution (Mendelsohn 2002). The marginal condition for undertaking further pollution abatement is that the additional costs of abatement are less than the costs of the health damages that are eliminated by abatement. Otherwise, in eliminating the last dollar of health damages, many times that amount would be spent. Given that there are always alternative means of improving human welfare economist would suggest spending these dollars instead on activities in which the benefits would exceed the costs.

While elimination of pollution and provision of clean water and sanitation to all may be a moral responsibility for some and a policy objective for others, the application of economic reasoning suggests that there will be some optimal level of pollution that is greater than zero.

Below the frameworks and methodologies involved in the valuation and evaluation of these impacts are summarized before turning to a summary of the empirical literature.

3.2.1 Valuation Methods

Valuation studies are those that analyze specific impacts and translate quantitative indicators such as working days lost into economic values. Evaluations are studies that compare alternatives and in so doing provide information about choices that present themselves to individuals or society. Evaluations inevitably rely on valuation methods in order to value specific costs and benefits. While many costs and benefits can be easily acquired from market data on prices and quantities this is not typically the case with the two topics considered here: health impacts and environmental management. In valuing what are called ‘non-market’ benefits a number of techniques have been devised by economists (Freeman 1993; Braden and Kolstad 1991). An overview of valuation methods including methods for both market and non-market methods presented in Figure 1 (Aylward et al. 2001). Of these two are of particular importance for the topic at hand: productivity and avertive expenditure methods

Figure 1. Economic Valuation Methods

	Observed Behavior	Hypothetical
Direct	MARKET PRICES (Direct Observed) Competitive market prices Shadow prices	STATED PREFERENCES (Direct Hypothetical) Contingent valuation (dichotomous choice, willingness-to-pay, bidding games)
Indirect	REVEALED PREFERENCES (Indirect Observed) Productivity methods Avertive (defensive) expenditure Travel cost Hedonic pricing Substitute goods	CHOICE MODELLING (Indirect Hypothetical) Contingent ranking Contingent referendum Contingent activity Potential expenditure methods (Replacement cost, relocation cost, mitigation cost, shadow project)

Source: After Aylward et al. (2001), based on Freeman (1993)

Both productivity and avertive expenditure methods are based on actual human behaviour that reflects utility maximization (Freeman 1993). However the environmental and health services (or impacts) being valued are not traded in markets. Thus the value of these services needs to be inferred from modelling the relationship between market goods and the service. In the case of avertive expenditures the action of removing pollution is a substitute for water or air quality. In the case of productivity methods the relationship is typically one of the environment (or health) as an input in the production of a market good.

The avertive expenditure approach is used to assess the impact of pollution through documenting the expenditures made by households in order to avert (or avoid) the pollution. For example, the treatment of water in the home with disinfectant would be a response to polluted water. The costs of this treatment to the household must exceed the costs of not doing so – i.e. the resulting health impacts – otherwise the household would not have undertaken the treatment. Thus, the avertive expenditure approach provides a valid estimate of the benefit of pollution prevention.

Productivity methods for environmental services and their variant in the case of health – the dose/response method – rely on a two-step analysis. The first step is linking cause and effect to determine the biophysical linkage between human activity and economic production/productivity. The second step is then to value the change in productivity using market prices or, if those are not available, one of the other methods for eliciting the price of a non-market good or service. In the case of health impacts, the dose – in this case the concentration of the pollutant – is linked to the incidence of the disease, which in turn is linked to days, or years of lost production due to sickness or death. In the case of environmental change, the change in land use must be linked to changes in downstream hydrological function (e.g. sedimentation rates) which in turn are then linked to changes in production (e.g. lost hours of hydropower production) and valued at their market rate.

In the case of the health impacts of waterborne disease caused by pollution from agricultural landscapes all of these methods come into play. However, it is important to highlight a key distinction between these benefit valuation methods and the replacement cost method, which is often used in these cases. In order to be a measure of benefit these valuation methods must be based on observed choices or at the very minimum hypothetical elicited values. These observed choices need to be made by welfare-maximizing individuals or households, or by firms that are cost-minimizers. Otherwise, there can be no guarantee that the benefits will not be over-stated, perhaps greatly so. This becomes an important feature that differentiates between a study that relies on calculations of costs actually incurred as a result of ongoing pollution, as opposed to a study that merely estimates the future costs based on some rough estimate of the change in level of impact.

Studies that rely on assumed willingness to avoid pollution can be categorized as replacement cost approaches. Such studies make the assumption that the cost of reducing pollution or providing the environmental service is inherently worthwhile. These studies simply rely on engineering cost estimates to provide an estimate of the benefit of reducing pollution. In fact, they are simply calculating cost projections but are not providing any information about benefits (Freeman 1991). This is an important point in understanding the economics of ecosystem services and one that will re-appear in later sections in this paper.

Another way of presenting this problem comes from a recent World Bank review of the water supply and sanitation sector, which found limited rigorous, empirical impact evaluations (Poulos, Pattanayak, and Jones 2006). One of the difficulties encountered in evaluating the costs and benefits of successful water quality improvement programs is that when such programs prevent future health impacts and consequent economic damages, these rely on projections of the counterfactual – that is what would have happened had the program not been implemented. While there are ways to construct such counterfactuals these are not quick and inexpensive methods and, therefore, while potentially applicable in illustrative case studies they will not be applicable generally. This situation has also been noted in the case of environmental conservation projects (Ferraro and Pattanayak 2006).

The problem therefore emerges in how to use site-specific case studies in drawing conclusions about projects, programs and impacts that occur across large and differentiated regions. As available studies are summarized below it is important to recall that as the scope and scale of the exercise increases, the assumptions and extrapolations required often also grow, lowering the reliability that can be ascribed to the numerical results.

3.2.2 Evaluation methods.

Evaluation is the science and art of comparing one alternative with another. As stated above valuation is a component of any evaluation. Before turning to the applicable evaluation methods it is useful to clarify that the even valuation may involve comparisons of different states of nature and, that this needs to be understood in order that the values employed in an evaluation are the correct ones.

One of the more misundersto0d issues in valuation/evaluation of ecosystem services is that even valuation includes a 'hidden' evaluation. Assume an agency wants to examine the costs of high rates of erosion that lead to sedimentation of water treatment facilities. In other words the agency wants to see what the benefits would be of dealing with this problem. Assume the only impact is that the agency regularly has to dredge the settling ponds due to high sediment loads. How do they calculate the costs? If they take the full costs of the dredging as indicative of the sedimentation costs then they are implicitly comparing the current situation with one in which there is no sedimentation. This may not be realistic. But even if they choose a level of dredging costs incurred prior to the increase in erosion and sedimentation there is still an implicit comparison of one situation with another. In this latter case they are comparing a 'background' level of erosion and sedimentation with the current 'accelerated' levels. Similarly with health impacts there is likely to be some background level of morbidity and mortality that needs to be zeroed out as a starting point in valuing the benefits of improved water quality. The point here is simply that the valuation methods discussed above, particularly the productivity method, implicitly relies on a comparison of one state of nature with another. Therefore, any evaluation needs to ensure that the alternatives being compared are consistent with those employed in the valuation approaches themselves.

In the example outlined above if a valuation study examines the cost of dredging vs no dredging at all the cost may be quite large. This might suggest to the uninformed that the benefits of avoiding the sediment in the first place are large. This may or may not be so. If the baseline or natural rate of erosion is high all the calculated 'benefit' in this case is not really available. Maintaining a forest area or improving agricultural practices might not generate significant benefits. Thus, a failure by the analyst to make this 'hidden' evaluation component transparent or the failure of a decision-maker or the public to understand this distinction may provoke confusion and possibly lead to poor decisions or public misperceptions.

With the provisos mentioned above taken into account regarding the evaluation inherent in valuation and the issue of counterfactuals and replacement costs, it remains true generally-speaking that the health costs of waterborne disease are equivalent to the benefits of improvements in access to clean water and sanitation. Two economic evaluation methods are generally employed to assist in choosing from available alternatives to remediate the impacts of poor water quality: cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA). A third technique, multi-criteria analysis (MCA) may include economic information such as valuation results on specific costs or benefit but does not rely exclusively on quantifying impacts in economic terms. Below CBA and CEA are reviewed briefly as they apply to the topic at hand.

CBA responds to the broad societal choice raised at the outset with regard to the level of investment that should optimally be made to remove pollution. The costs of improvements are compared with the benefits of removing pollution in order to evaluate whether or not, and to what degree, different alternatives generate net benefits for society. Each set of actions analysed is then, loosely speaking, the marginal set of actions and when the benefits of taking action are outweighed by the costs it can be assumed that an optimal level of pollution has been reached.

CEA still serves to evaluate alternative courses of action but it does not answer the broader question of whether to act or not. In CEA a type and level of benefit is specified. Alternative ways of providing this benefit are defined and the costs calculated. In the case of health interventions the preferred unit is the change in Quality Adjusted Life Years (QALY) (Weinstein and Siegel 1996). The cost-effectiveness ratio is then the $ per unit figure (costs in the numerator and benefits in non-monetary units in the denominator). The costs of reaching some level of improvement in the output (e.g. QALYs) with the different alternatives can then be calculated and compared in order to find the least-cost alternative.

The advantage of CBA is that it provides information on not just which alternative to choose, but whether to choose the best alternative or do nothing (i.e. invest the funds in other welfare-enhancing opportunities). The disadvantage of CBA is that it requires an economic valuation of benefits. This is typically requires more research effort and the existence of relevant data in order to provide reliable results. The advantage of CEA is that it does not pretend to evaluate the usefulness of the objective merely on economic grounds. Typically such investment decisions are not purely economic in any event, involving as they do a host of sociopolitical factors. The technical advantage of CEA is that it avoids the issue of measuring benefits as each of the solutions has the same benefit level. Instead the analysis hinges on the estimates of the direct costs of implementing the alternative, in this case being the cost of different land use practices or water treatment solutions. Comparatively-speaking these are far easier to obtain then health benefits using valuation methods.

A hypothetical but generic example serves to summarize this discussion. Imagine a watershed where biological contamination from non-point agricultural sources are causing poor water quality which is leading to poor quality piped water to households and a high incidence of diarrhoeal disease. Three generic ways of resolving the problem exist: (a) promoting improved agricultural practices (b) investing in improved treatment at the intake from the river and (c) distribution of disinfectants for use in households. A CBA approach would involve developing information on the costs of these approaches, the resulting level of health improvement and the economic benefits of these improvements. In some cases even within an alternative there would be choices as to scale or exact approach, and therefore the level of costs and benefits. It might also be that these are not mutually exclusive alternatives but that there would be some combinations of approaches that have promise. The economic profitability of all the alternatives would then be examined and, subject to funding, the most profitable approach would be selected, subject to it earning an acceptable rate of return on the funds employed.

With CEA the decision would be reduced to setting different target levels for the disease or the contamination and then working backwards to finding the associated costs of each alternative. The selected alternative(s) would be the solution that maximizes the reduction in pollution and improvement in health for a given level of expenditure. This would be the cost-effective solution at that level of funding.

While CEA is a fairly straightforward concept as proposed in economics textbooks, there are a number of variants and issues that crop up in implementation that provoke debate as to the correct application of the approach. This is particularly true in the health field and considerable effort has been expended by various agencies and groups, including WHO, the British Medical Journal Economic Evaluation Working Party and United States Panel on Cost-Effectiveness in Health and Medicine USEPA, in trying to delineate clear guidelines for how CEA should be applied (Weinstein and Siegel 1996; Edejer et al. 2003; Drummond and Jerrerson 1996). A number of the key issues include (Drummond and Jerrerson 1996; Hutton 2000; Murray et al. 2000):

- The problem of joint products: health interventions, particularly environmental health interventions may produce additional benefits, including benefits to other than the health sector, which are typically ignored if the unit benefit is constrained tightly to the desired health outcome. This makes interventions subject to this problem look less attractive financially in CEA and may lead to choosing alternatives that do not really profit the optimimum social return on investment
- Ministries of Health are unlikely, in any event, to consider non-health outcomes in their evaluations of interventions, unless these outcomes provide a means of obtaining a cost-share contribution (financing) on the intervention.
- Methods for the quantification and economic valuation of benefits of health interventions, particularly in developing countries, are underdeveloped and using CEA (or CBA) to analyze context-specific decisions is likely to be too expensive in these countries.
- Emphasis on CEA of new interventions often fails to properly address existing misallocation of resources (i.e. cost-ineffective interventions already in use).
- There are significant disagreements about how to account for equity in CEA calculations including with regard to future generations (use of the discount rate) and social inequality (the use of distributional weights).
- Determining the impact of interventions remains difficult due to a number of methodological issues including in particular the dividing line between the direct costs of the intervention (the numerator) and the savings or benefits (the denominator)

As regards the last bullet point it is recommended that cost savings realized by the implementation of a health intervention are deducted from the direct costs of the intervention in arriving at the numerator (Edejer et al. 2003). Such costs include medical costs previously expended to cope with the disease. The difficulty with this, as explained above, is that costs previously incurred due to illness or disease that are subsequently saved due to the intervention are in fact benefits of the intervention not costs. While this approach may not produce incorrect results, it highlights some of the complications of actually carrying out such studies. Further, as argued above if such saved costs are merely assumed as opposed to being observed, such studies may vastly understate the cost-effectiveness of an intervention.

A final difficulty emerging from Hutton's (2000) review of available studies of the economics of health interventions is that there is a serious lack of such studies available to guide decision-making. CBA and CEA can be undertaken at different scales, from site-specific case studies to regional or global analyses. Hutton's comment refers to rigorous analyses of specific cases, which by their very nature are context-specific. In debating the future of CEA in health decision-making Murray et al (2000) raise the counterpoint that merely trying to establish 'generalized' CEA is sufficient. That is rather than studying specific interventions in specific contexts, which is very expensive, it would be far more useful to try and develop generalized cost-effectiveness indicators for commonly used interventions. Unfortunately, at present evaluations of interventions are commonly undertaken in comparison to the current situation and the set of interventions in place. This makes transferring CEA information from one place to another difficult as the existing interventions may vary from one locale to another even though they have comparable health systems (read costs) and epidemiological profiles (i.e. response to the intervention). Murray et al. (2000) suggest establishing generalized CEA, which would measure effectiveness against the no intervention case rather than the current interventions case. In this way generalized CEA estimates for standard interventions could be developed for regions that are relatively homogenous in terms of health systems and epidemiological profiles. However, they acknowledge that such information is largely lacking at present.

This perspective from the health field on CEA is valuable as later in the paper the same issue of context-specific CEA vs. generalized CEA will occur in the case of environmental management of agricultural landscapes.

With the understanding of valuation and evaluation approaches the paper now turns to what is currently known about the economics of agricultural landscapes, water quality, domestic water supply and human health. There is no single case study (or literature review) that examines the full set of linkages as described in Section 2 of this paper. There are studies that examine the costs and benefits of levels of water supply and sanitation, and in this context the economics of poor water quality and the economics of improvements that affect waterborne disease. These occur at different levels but a recent study by WHO at the global level is instructive. The study does not specifically consider the contribution of ecosystem management, particularly in agriculture, but it does provide a sense of the overall domestic water quality problem and the potential aggregate role of ecosystem management in this context. There are also a number of studies that have examined generally the economic costs and benefits of ecosystem management as it relates to water quality. These are not necessarily specific to domestic water supply and human health – but they serve to illustrate the general level of knowledge regarding the economic consequences of ecosystem management with regard to water quality impacts.

3.3 Global Estimates of Costs and Benefits of Health Impacts in Water Supply and Sanitation

A recent attempt by WHO to assess the costs and cost-effectiveness of different interventions in meeting the Millennium Development Goals (MDG) for water, as well as other objectives, provides an excellent source of information and perspective on what might be the potential value and market for reductions in water pollution from agricultural landscapes (Hutton and Haller 2004). The analysis examined the 5 interventions listed below (which are further defined by reference to the scenarios in Table 8:

1. Meeting MDG goals for water supply which require halving by 2015 the number of people without access to safe drinking water (Scenario VI to Vb, or Scenario Va to IV)

2. Meeting MDG goals for water supply and halving by 2015 the number of people without access to necessary sanitation (Scenario VI to IV, or Scenario Va or Vb to IV)

3. Increasing access to water supply and sanitation for all (Scenario VI, Va and Vb to IV)

4. Providing for 3 above as well as point of use disinfection (Scenarios VI, Va, Vb and IV go to Scenario III)

5. Providing regulated piped water supply in house and sewage connection with partial sewerage for all (Scenarios VI, Va, Vb, IV and III go to Scenario II)

Table 8: Selected Exposure Scenarios

Level	Description	Environmental faecal-oral pathogen load
VI	No improved water supply and no basic sanitation in a country which is not extensively covered by those services, and where water supply is not routinely controlled	Very high
Vb	Improved water supply and no basic sanitation in a country which is not extensively covered by those services, and where water supply is not routinely controlled	Very high
Va	Improved sanitation but no improved water supply in a country which is not extensively covered by those services, and where water supply is not routinely controlled	High
IV	Improved water supply and improved sanitation in a country which is not extensively covered by those services, and where water supply is not routinely controlled	High
III	Improved water supply and improved sanitation in a country which is not extensively covered by those services, and where water supply is not routinely controlled, plus household water treatment	High
II	Regulated water supply and full sanitation coverage, with partial treatment for sewage, corresponding to a situation typically occurring in developed countries	Medium to low
I	Ideal situation, corresponding to the absence of transmission of diarrhoeal disease through water, sanitation and hygiene	Low

Source: Hutton and Haller (2004)

Costs and benefits of the resulting decrease in diarrhoeal disease were calculated for each intervention. The analysis was undertaken for these interventions in comparison to actual country conditions in 2000. Results were calculated by country and regional results formed based on country averages weighted for population. Outcomes suggest that all the interventions are profitable investments for society. The largest contributor to the benefits was the time savings from better access to water supply and sanitation (WS&S). Below the elements of this analysis of relevance to this paper are identified and discussed. As one of the regions was sub-Saharan Africa, the study by Hutton and Haller (2004) provides important information on conditions underlying Section 6 of this paper and illustrative costs and benefits will be summarized for Africa.

3.3.1 Costs

The analysis of costs by Hutton and Haller (2004); hereafter the WHO study, included:

- the investment costs of each intervention including: planning and supervision, hardware, construction and house alteration, protection of water sources and education that accompanies an investment in hardware
- the running costs of each intervention including: operating materials to provide a service, maintenance of hardware and replacement of parts, emptying of septic tanks, and latrines, regulation and control of water supply, ongoing protection and monitoring of water sources, water treatment and distribution, and continuous education activities

In Africa the most expensive treatment for improved water supply is providing in regulated water supply (house connection) and sewerage with an investment cost of \$102 per person. Household connection are estimated to have a length of life of 40 years with operations and maintenance of 30% percent and water source protection as 10% of the annualized cost. In addition water treatment costs are estimated at \$0.30/m^3 in Africa. When computed using a loan value at 3% interest rate for the investment cost these suggest annualized costs of \$12.75 per person reached in Africa for regulated water supply. Table 9 provides information on annualized costs for all the interventions examined by the WHO study. Of particular interest here is the proportion of total cost of regulated water supply represented by water treatment. With water requirements of 60/liters/person/day the total annual per capita water requirement is 21.9 m^3. So the total water treatment costs in Africa would be \$6.60 per person/year or almost half of the total costs. This suggests that avoiding the need for water treatment costs or reducing existing levels of treatment cost may be a significant economic savings in water supply and sanitation. However, these treatment costs are likely an annualized figure that reflects both capital expenditure and O&M and further effort is required to understand how these expenditures would shift in response to changes in water quality. This topic is taken up further below in Section 4.3.

Table 9. Annual Costs for Improvements on a Per-Person-Reached Basis

Improvement	Annual cost per person reached (US\$ year 2000) Africa	Asia	LA&C
Improved water supply			
Standpost	2.40	4.95	3.17
Borehole	1.70	1.26	4.07
Dug well	1.55	1.63	3.55
Rain water	3.62	2.51	2.66
Disinfected	0.33	0.26	0.58
Regulated piped water in-house (hardware and software)	12.75	9.95	15.29
Regulated piped water in-house (software only)	8.34	5.97	9.06
Improved sanitation			
Septic tank	9.75	9.10	12.39
VIP	6.21	5.70	5.84
Small pit latrine	4.88	3.92	6.44
Household sewer connection plus partial treatment of sewage (hardware and software)	10.03	11.95	13.38
Household sewer connection plus partial	4.84	5.28	6.46

Source: Hutton and Haller (2004)

In aggregate the WHO study finds that the annual costs of the different interventions varies widely, from just under \$2 billion for #1 to \$25 billion for #4 and \$136 billion for #5. Leaving aside #5, these interventions would require the expenditure of from \$0.5 to \$4.3 billion in sub-Saharan Africa. Clearly, this is a sizeable annual investment in resolving a serious human health problem. Investigating whether better ecosystem management in agricultural landscapes could play a useful and cost-effective role in solving this problem therefore appears a worthwhile endeavour.

3.3.2 Benefits

The WHO study focuses on waterborne and water-washed diseases, as these are the ones most directly linked to household water use and sanitation. Diarrhoeal diseases of various origins are the primary culprits. It is important to reemphasize that these diseases have a number of transmission routes, only one of which is poor domestic water quality. Thus, the benefits described here could only proportionately be allocated to improvements in water quality.

Benefits are calculated based on reductions in incidence and mortality rates for infectious diarrhoea. In Africa the population starts from what comparatively is the worst situation in terms of exposure scenarios (as per Table 8). Approximately 35% have no improved water supply and no access to sanitation (Level VI). The remainder of the population land in exposure scenarios IV and V implying no 'regular control of water supply.' In other words the bulk of the population in sub-Saharan Africa is at a high or very high risk for diarrhoeal transmission.

Three categories of benefits were considered by the WHO study:

- Direct economic benefits of avoiding diarrhoeal disease
- indirect economic benefits related to health improvements
- non-health benefits related to water and sanitation improvements

Each type of benefit is discussed in term below and the results are summarized in Table 10.

Direct economic benefits considered by the study include the resource costs that the health sector, patients and employers would have incurred in the absences of the health intervention. In other words the benefits are framed as the averted expenditures associated with illness and death. The range of health costs avoided is calculated at between \$10 and \$23 per case of diarrhoea depending on the region and for patients the avoided expenditures (on travel and food) are estimated to be \$0.50 for outpatient care and \$2 per inpatient. As per the earlier discussion a critical assumption made is that the average case would visit a health facility once and that 8% of cases would be hospitalised. The avertive expenditure approach is only a valid measure of benefit if it is clear that the expenditure was incurred prior to the remediation of the cause of the problem. The extent to which families seek medical care for every incidence of diarrhoeal disease might be questioned. WHO data is used to arrive at the hospitalization rate but there appears to be no similar observed data set for the ambulatory cases. The WHO study may therefore overstate the direct benefits. This may be particularly true for Africa where those in rural areas may not have access to health care facilities. However, to compensate for this the WHO study does use 0.5 visits per case in the sensitivity analysis.

These direct benefits are valued annually at \$2.1 to \$52 billion globally for the range of interventions. For Africa they are \$0.6 to \$9.2 billion and make up between 15 and 20% of total calculated benefits for interventions #1 through #4 (excluding in house connections).

Indirect economic benefits of improving WS&S and lowering infectious disease rates include the increased human productivity from living longer and being sick for fewer days. The valuation problem in this case is how to value time. The study uses the minimum wage, but in the sensitivity analysis uses per capita Gross National Product as the low value scenario for adults. This, as the study, admits that in many developing countries it is not reasonable to assume that all increases in available working time would earn the minimum wage, given unemployment, underemployment and lower than minimum wage returns in rural sectors of the economy. Again, this criticism of the approach taken is particularly salient in the case of Africa where the formal employment sector is limited in scope.

With respect to incidence of disease for school age children the study also values time (and school absenteeism due to illness) at the minimum wage. For infants a value of 50% of the wage rate is used based on lost productivity of parents. These are merely assumptions rather than attempts to value benefits and their foundation is not clear. The study relied on a few specific studies to estimate the length of average episodes for the different age groups. Obviously this is an area for further research and the results, aggregated as they are across different regions are subject to question.

The benefit from reduced morbidity is valued annually at between $3.0 and $75 billion globally. For Africa the benefits are calculated to be between $1.5 and $18 billion, making up between 15 and 22% of total benefits.

A final indirect benefit is the loss of productivity associated with mortality. The study used general estimates of productive life for each age group and the minimum wage to reflect the opportunity cost. Resulting annual benefit streams were then discounted back to the present at 3%. These benefits were valued annually at between $1.0 and $23 billion globally and, in Africa, between $0.6 and $9.4 billion. These benefits were less significant as a portion of total benefits in Africa varying between 6 and 12% of total benefits.

While there may be many non-health benefits the study quantified only the benefits emerging from the time savings due to improved access to water supply, i.e. standpipes or inhouse connections. As these have nothing to do with water quality and health they are not discussed further here other than to point out that they were the largest single category of benefits calculated by the WHO study. Globally they varied between $12 and $400 billion annually and, in Africa, they contributed between 55 and 69% of total benefits calculated in the study.

It is quite likely that these benefits are substantially overestimated (apart from sensitivity analysis) and therefore the conclusion that the benefits outweigh the costs by such large multiples is suspect. While it is hard to know how far to take this criticism it is worth pointing out that with benefit-cost ratios for interventions #1 through #4 in Africa ranging between 10 and 14 it does appear that doing something about the WS&S problem in Africa is warranted. Benefit-cost ratios that exceed 1 are generally considered for investment as the ratio implies that the benefits of the activity exceed the cost. For example, even with the time savings excluded all the interventions remain beneficial in the African Context. Further substantiation comes from the WHO study's sensitivity analysis in which the high cost and low benefit scenarios were evaluated. For Africa the results produced benefit cost ratios of from 1.1 to 2.9. Even for intervention #5, in house connections demonstrated benefits that outweighed the costs.

Returning to the base case direct and indirect health benefits that are attributable to changes in the quality of water (and not just access to water), substantial benefits are suggested by the WHO study, ranging from $2.3 to $36 billion per annum for sub-Saharan Africa. Again, this suggests the utility of investigation the potential role of ecosystem management in the agricultural landscape.

Table 10. Summary of African and Global Costs and Benefits from Investments in WS&S

all figures in US$ millions	Interventions 1	2	3	4	5
A. Total annual cost of interventions					
Africa	490	2,021	4,043	4,360	24,729
World	1,784	11,305	22,609	24,649	136,515
B. Benefits					
B1. Annual health sector treatment costs saved					
Africa	564	1,695	2,410	6,742	8,625
World	2,020	6,975	11,624	38,337	50,022
B2. Annual patient treatment costs saved					
Africa	36	107	152	427	547
World	97	341	565	1,787	2,322
B3a. Value of productive days gained					
Africa	38	116	168	472	605
World	210	737	1,252	4,212	5,508
B3b. Value of school absenteeism and baby days lost					
Africa	1,093	3,293	4,727	13,287	17,011
World	2,759	9,240	14,695	53,046	69,789
B4. Value of averted deaths					
Africa	605	1,820	2,607	7,314	9,360
World	1,035	3,560	5,585	17,566	22,803
B5. Value of time savings					
Africa	3,004	15,877	33,972	33,972	72,293
World	12,022	63,547	229,158	229,158	405,457
C. Africa Totals					
Total Benefits	5,340	22,908	44,036	62,214	108,441
Net Benefits	4,850	20,887	39,993	57,854	83,712
B-C Ratio	10.90	11.33	10.89	14.27	4.39
D. World Totals					
Total Benefits	18,143	84,400	262,879	344,106	555,901
World Net	16,359	73,095	240,270	319,457	419,386
B-C Ratio	10.17	7.47	11.63	13.96	4.07

Notes: B-C Ratio is the ratio of the benefits to the costs of the activity so that a ratio that exceeds 1 is assumed to be worthy of investment in that it yields benefits that exceed costs. The interventions and change in environmental faecal load (Scenarios from Table 8) are:

1. Meet MDG goals for water supply: very high (VI, Vb) or high (Va) go to very high (Vb) or high (Va, IV)

2. Same as 1 but also halving by 2015 the number of people without access to necessary sanitation: very high (VI, Vb) or high (Vb) go to high (IV).

3. Increasing access to WS&S for all: very high (VI, Vb) or high (Va) go to high (IV).

4. Providing for 3. as well as point of use disinfection: very high or high (VI, Va, Vb and IV) go to high (III).

5. Providing regulated piped water supply in house and sewage connection with partial sewerage for all: change from very high, high (VI, Va, Vb, IV and III) to medium to low (II)

3.4 Valuation Studies of Agricultural Water Pollution and Impacts on Domestic Water Supply

The difficulty with the work purely on costs and benefits of health impacts is that these studies do not help identify the costs that are attributable to agricultural water pollution per se. It would be more satisfying to understand the economic costs to human health specifically of poor water quality. It would be even more satisfying then to link such analyses back to ecosystem management, so that the impact of changes in ecosystem management on downstream water quality, the domestic use of water, water treatment costs and human health were clear.

For site-specific analysis of the economic damage caused by poor water quality the application of the valuation methods reviewed above in Figure 1 are well described (Bouwes 1979; Ribaudo, Young, and Shortle 1986; Ribaudo and Young 1989; Lant and Mullens 1991; Bockstael, Hanemann, and Kling 1987; Duda 1985; Smith and Desvousges 1986; Willis and Foster 1983; Young 1996). In a developed country context, there are many such case studies. For example, in a review of studies published during the 1991 to 1997 period that valued freshwater ecosystem services, 21 of 30 studies identified involved analysis of water quality benefits (Wilson and Carpenter 1999). However, these were mostly studies of water quality impacts on recreation, tourism and property values, values associated directly with water quality in streams and rivers.

An additional problem is that these water quality valuation case studies typically are not linked to land use in specific geographical areas, nor do they evaluate damage that is directly and only related to land use change. Oftentimes the measure of water quality that can actually be evaluated (as perceived by recreationalists for example) is extremely crude (i.e. water quality is good or bad), so that associating the measure of damage with a particular type of non-point source pollution is impossible.

By far the largest (though still small) set of studies in the literature have to do with erosion and sedimentation impacts only a few studies make the linkages necessary to quantify these costs in the case of water treatment. A number of these studies demonstrate significant external effects.

For the United States, Clark et al. (1985) gathers related research on practically every conceivable off-site impact of eroding soils and provides a nationwide estimate of the annual monetary damage caused by soil erosion of $6.0 billion ($15 billion in 2006 dollars). Even so Clark et al. concludes that this figure may be severely under-estimated as the impact of erosion on biological systems and subsequently on economic production and consumption is not included. Clark et al. include in their study not just erosion and sedimentation impacts, but the impact of ‘erosion-contaminants’, which are impacts related to water quality more generally, including the effects of pesticides and fertilizers that are used in agricultural production. Based on a range of estimates from other studies of between $50 and $500 million, Clark et al. calculates that water treatment costs due to soil erosion in the United States are $100 million annually ($245 m), of which some $30 million ($73 m) is attributable to cropland. A further $900 million annually, of which $300 million from cropland ($730 m), is Clark et al.’s estimate of the impact of total dissolved solids on municipal and industrial users of water.

The estimates by Clark et al. (1985) serve the purpose of dramatizing the potential magnitude of the off-site damage caused by soil erosion and poor water quality downstream. However, it must be acknowledged that the quality of a majority of the studies drawn upon by Clark et al. is mediocre. Holmes (1988) summarizes this criticism by stating that the Clark et al. (1985) study “is based to a large degree on ad hoc interpretation of a widely divergent group of studies.” The majority of these studies rely on simple damage function estimates of changes in costs or revenues, absent any consideration of optimizing behaviour on the part of consumers and producers as reflected in supply and demand curves.

Holmes' (1988) undertakes a sophisticated study using a firm model and then a hedonic cost function of the nationwide costs of soil erosion to the water treatment industry produces a range of $35 million to $1.37 billion per year ($68m to 2.66b in 2006 dollars). This range is close to that provided by Clark et al. (1985) of from $50 to $500 million, even though Holmes' best estimate from the hedonic model is $353 million ($685m) is three times larger than Clark's best estimate of $100 million. Holmes' best estimate of the costs of turbidity is $0.012/m^3 ($0.023/m^3) and $0.017 per ton of discharged sediment ($0.033/ton). At the same time, it must be acknowledged that despite the sophistication in methods, Holmes methods produce a wide range of results indicating continued uncertainty over the true magnitude of these sorts of damage estimates.

Still a number of useful figures and conclusions for this study emerge from Holmes (1988):

- data from 430 of the largest municipal utilities in the US suggest annual variable treatment costs associated with direct filtration systems are $0.019/m^3 and ($0.038/m^3) and with conventional filtration plants are $0.032/m^3 ($0.062/m^3).
- most utilities with water quality of 10 NTU or greater have adopted the higher cost conventional filtration practices
- total surface water withdrawals for M&I purposes in the US are 5.8 million m^3/day
- source water turbidity level is a significant explanatory variable for utility O&M expenditure, and a 1% increase in turbidity is associated with a 0.07% increase in O&M expenditure
- comparison of regions with the greatest sediment discharge and the greatest benefits induced by a 10% decrease in sediment loading suggest that sediment loading (or erosion levels) will not necessarily locate the areas with the greatest offsite benefits from sediment reduction.

The final point above suggests that allocation of resources to conservation programs based on onsite criteria may be inefficient (Holmes 1988). This conclusion is echoed by the authors of a study of the on- and off-site costs of soil erosion in Southwestern Ontario (Fox and Dickson 1990). In the latter study the authors cite a number of sources, including US EPA, as stating that the cost of water treatment attributable to cropland erosion is $0.012 to $0.050/m^3. Both of these studies suggest that impacts of sediment on recreation are likely to be more significant impacts in economic terms than impacts on water treatment (Holmes 1988; Fox and Dickson 1990).

Economists in Ohio took a simpler but similar route to that taken by Holmes, in examining soil erosion and water treatment costs for 12 small community systems of between 2,000 to 22,000 in population (Forster, Bardos, and Southgate 1987). Erosion rates, turbidity levels, retention time in settling ponds and volume of treated levels all served to explain the level of variable inputs (largely chemicals) in water treatment. Average annual variable costs of water treatment came to $3.83 per person ($7.58 in 2006) and $0.023/m^3 ($0.045/m^3). Similar to Holmes (1988) a 1% increase in turbidity was associated with a 0.12% increase in variable costs. For each 1% change in soil erosion rates a corresponding 0.4% change in water treatment costs was observed. The potential impact of multicollinearity between the independent variables of turbidity and soil erosion is not addressed in the study.

With respect to retention times, Forster et al. (1987) state that larger storage capacities relative to daily throughput imply less cycling through of water and therefore more time for suspended sediment to settle in the reservoir prior to actual treatment. The authors go on to suggest that the more sediment that settles in the reservoir the higher the likelihood that it may be re-suspended by storm events. They therefore posit the longer retention times should increase treatment costs. However, the authors note that in their study all the systems had the same retention time. The lack of variation in this independent variable and the potential for correlation between erosion and turbidity variables raises questions regarding these results. For retention time, it could just as equally be maintained that longer retention times increase the natural removal of sediment (by settling) therefore reducing the variable level of chemicals required in treating the water. However, this may in turn lead to an increase in labour or equipment costs if the sediment needs to be removed from the reservoir. As labour was a fixed costs for these small community systems such an impact was not included in this study. However, it does raise the question of whether investments in storage and settling ponds are cost-effective substitutes for the variable inputs involved in sediment removal.

A third study of a similar nature employs daily data over a four year period from 10 cities selected purposefully to be representative of conditions from amongst 142 cities in Texas that treat surface water separate from groundwater (Dearmont, McCarl, and Tolman 1998). The results are similar in nature to those of the earlier studies and are summarized along with the results from the other studies in Table 12. It may be worth noting that the higher responsiveness of water treatment costs to turbidity (elasticity of 0.27 as opposed to 0.07 and 0.12) from the Dearmont et al. (1998) study may reflect that the Texas sample had twice the average level of turbidity than did the Holmes (1988) study, which in turn had 3 times the average level of turbidity of the sample in the Forster (1987) study. The Dearmont et al. (1998) study also attempted to incorporate the added cost of chemical contamination through a proxy variable that reflected the presence of chemical contamination in local groundwater. This variable had significant explanatory power and suggested a $0.025/m^3 ($0.04/m^3) cost per cubic meter for treatment of waters that are contaminated with chemicals.

In a national level study of the external environmental and health costs of agriculture in the UK figures for the UK that are comparable with those from Clark (1985) and Holmes (1988) are calculated (Pretty et al. 2000). Using data on actual expenses by the water treatment industry between 1992 and 1997, and employing simplifying assumptions as to the percent of source pollution from agriculture the authors calculate that the annualized costs (capital and O&M) associated with mitigating for pesticides, phosphates, soil, pathogens, and nitrate is $427 annually (in 2006 US dollars). The breakdown of results is presented in Table 11. The significance of the capital costs include in this amount is unclear as annual expenditures on capital costs are simply summed for the five-year period instead of taking an annualized value. There may have been an assumption that this is equivalent to the annualized value of all investments in water treatment plants. However, this is not stated and in any event relies on the assumption that the period for which data was collected is typical of historical investments, which may not be the case. In a related article under, the same authors use the cost figures for nitrates as part of estimates of the value of the impact of eutrophication of surface waters in England and Wales (Pretty et al. 2003).

Table 11. Costs of Treating Agricultural Water Pollution in the United Kingdom

Source of Pollution	Annual Costs			O&M Expenditure as Percent of Capital Expenditure	Percent of Pollution from Agriculture
	Best Estimate		Range		
	(£1996)	($2006)	($2006)		
Pesticides	120	240	168-258	8%	89%
Phosphate and Soil	55	110	44-180	9%	43%
Pathogens (esp. Cryptosporidium)	23	46	30-60	All O&M	90%
Nitrate	16	32	16-66	9%	80%
Totals	214	427	258 - 563		

Notes: UK currency converted at 1996 rate and inflated using US CPI

Taken alongside the Holmes study and employing an estimate of urban populations in both the US and the UK leads to the conclusion that the per person costs of water treatment associated with agricultural water pollution varies from $4 (US) to $8 (UK). The higher number for the UK may reflect higher population density in the UK, the later date of the UK study (and increased pollution levels), or perhaps the inclusion of more costs than merely those of sedimentation.

Clearly much work remains to be done in refining such estimates (summarized in Table 12). In particular, one difficulty of many of these studies is that they simply measure existing damage levels and do not consider to what extent these damages could be mitigated by alternative land uses or production technologies. Nor do they subsequently assess the trade-off between alternatives and the existing situation. This may be an important point as even improved technologies will produce some erosion and sedimentation. Of course, oftentimes an understanding of how damage relates to different sediment levels is missing from the studies as well, making it difficult to understand the form of the relationship and how it might be altered by partial reductions in sedimentation rates. The application of a damage function approach that evaluates the choice between the option to undertake conservation and postpone the decision may be worth investigating in this regard (Walker 1982).

Ironically, in a potentially comprehensive and pioneering case study of agricultural costs and benefits of implementing the European Water Framework Directive, a body of research scientist decided to focus on non-marketed benefits such as recreation and to exclude the analysis of water treatment costs as relatively uninteresting (Bateman et al. 2006). Interesting or not, the question is how relevant they are to decision-making. It should also be reiterated that water treatment costs associated with erosion and sedimentation are just one element of the water quality benefits associated with improved land management.

Table 12. Municipal Water Treatment Costs due to Turbidity and Sediment

Study	Location	Total Costs (million)	Variable Water Treatment Costs	Elasticity	Other results
Clark et al. (1985)	USA	$1,075			
Holmes (1988)	USA	$685	$0.023/m^3	0.07 (T)	cost of $0.033/ton (S)
Forster et al. (1987)	Ohio		$0.045/m^3	0.12 (T)	elasticity 0.40 (SE)
Dearmont et al. (1998)	Texas		$0.031/m^3	0.27 (T)	cost of $0.040/m^3 for groundwater contamin.
Pretty et al. (2000)	UK	$427			

Notes: Elasticity figures are the % increase in expenditure associated with a 1% increase in water quality. Water quality as an explanatory variable is measured either as Turbidity (T), Sediment (S) or Soil Erosion (SE) in these studies. All figures in 2006 dollars, adjustments are made based on the year of the data employed, exchange rate at that time and inflation using the US Consumer Price Index.

A number of studies have examined the benefits of protecting groundwater sources from contamination by nitrates and other contaminants. On Cape Cod in Massachusetts, USA a study found that households were willing to pay from $5,000 to $20,000 per household (present value over 30 years) in order to have a 25% chance to a 100% certainty that groundwater contamination by nitrates would not exceed US EPA standards for drinking water of 10 ppm (Edwards 1988). On the other hand, in an ex-post evaluation of the benefits of the Clean Water Act regulations in the Chesapeake Bay watershed from 1972 to 1996, benefits to domestic water systems were not included in the analysis (Morgan and Owens 2001). Health as a category of benefit is discussed but not water quality impacts on household water use. Interestingly Morgan and Owens (2001) do report that municipal water treatment costs are on the order of $650 to $854 million per year in the region. Impacts on property prices due to lower faecal coliform levels are mentioned but are not included in the analysis as the Bay monitoring program does not track this contaminant. Confirming the linkage between water quality and property values a recent hedonic valuation study in a micro-watershed of the Chesapeake finds that changes of one mg/l in dissolved inorganic nitrogen leads to an increase in residential property value of just over $17,000 (Poor, Pessagno, and Paul 2007).

In a developing country context little in the way of comprehensive published valuation studies of the kind reported above for developed countries are available. Aylward (2004) reports on a number of studies in developing countries of linkages between land use, erosion and sedimentation but these are largely related to the use of reservoirs for hydropower and irrigation. An unpublished study from Costa Rica compares water treatment costs in a forested and deforested watersheds and finds that treatment costs are higher in the deforested watershed (CCT and CINPE 1995). However, this cost is just Costa Rican colones 0.04/m^3 or $0.0004/m^3 in 2006 dollars. Based on water use estimates in the paper it can be concluded that the additional costs of cleaning up water in the deforested watershed come to just $0.01/month per household (Rojas and Aylward 2003). Although the reliability of the numbers in the CCT and CINPE (1995) study is open to question the extra water treatment costs forms just 8% of the costs of protecting a forested watershed, as calculated by the same study (Rojas and Aylward 2003).

With regard to water quality issues beyond merely the off-site effect of erosion, Aylward (2004) found no studies in the developing country literature that specifically assess the downstream externalities associated with nutrient or chemical outflows associated with land use change.

Johnson and Baltodano (2004) attempt to use a contingent valuation survey of willingness-to-pay for water quality improvements in a rural micro-watershed in Nicaragua, but find only a marginal interest in such improvements at 0.5% of household income. For a watershed of 25,000 households total willingness to pay was just $10,000 per year or about $0.42 per household. For the roughly 180 km^2 micro-watershed this would amount to $0.58 per hectare or a $100 per hectare payment annually for improvements on 100 hectares or 0.6% of the areal extent of the watershed. The willingness to pay for improvements appears relatively modest in this case (Johnson and Baltodano 2004). In another study in the urban area of Davao, Philippines a contingent valuation survey found that willingness-to-pay for water quality improvements for the purposes of recreation was low, both in absolute terms and as a percentage of income (Choe, Whittington, and Lauria 1996). While pollution in the Davao case is substantial needs other than recreation are obviously more pressing on the population. A World Bank study of the costs of pollution in the Sebou Basin of Morrocco finds that pollution levels in the basin are so high – in part from agricultural water pollution in the lower basin – that water treatment costs are three times the national average (at $2.16/m^3 in 2006 dollars) (Sadoff 1996). The report goes on to tabulate the water treatment savings, health and productivity benefits and fisheries benefits of improving water quality and finds total benefits far exceed the costs of a $240 million program to improve water quality in the basin.

None of these studies directly values the issues confronted in this paper, but they reflect the level of interest and investigation of these topics in developing countries.

4. Water Quality Management

The management of water quality has a long history, however, the tools employed have continued to evolve. This section begins with a review of the arrangements and incentives that are used to govern water quality on behalf of society. A summary of the types of technology that are employed on the ground to manage water quality is then provided. This summary is broken down into two sections, one that considers the water treatment function of domestic water supply systems and one that considers the management of point and non-point sources relevant to agricultural landscapes. Where possible in the two latter sections the unit costs of such technologies are identified.

4.1 Institutional Arrangements and Incentive Mechanisms for Managing Water Quality

As with most types of pollution, economists have long considered water pollution a public bad. In other words, investing in maintaining water quality is a public good. Once water quality is provided it is difficult to exclude potential users downstream from consuming this service. Water quality is also not a separable good in that it is part and parcel of water quantity. The enjoyment by one water user of water quality does not therefore affect the utility derived by other consumers. It is therefore not unexpected that a free market system will lead to an underinvestment in water quality as individual polluters are unable to fully capture any benefits generated by investing in pollution control or mitigation. The production and provision of water quality therefore requires collective action, typically the province of centralized authority in the form of government.

Historically, however, there has been a gradual evolution in the manner in which central authority has sought to regulate and control pollution (not just water pollution). As pollution is produced by large segments of economic activity and by numerous social actors the pollution control problem is one of finding ways of achieving what economists call incentive compatibility. This means that the existing structure of costs and benefits – i.e. the incentives facing the polluter – need to be changed so as to align these incentives with the optimal (or efficient) level of pollution from a societal perspective. Such a perspective takes into account the full range of costs and benefits accruing to different sectors and groups in society. As discussed earlier the efficient level of pollution is that level where the marginal costs of abatement are equal to the marginal benefits of a further reduction in pollution (Mendelsohn 2002).

Four different blends of institutional arrangements and incentive mechanisms can be applied to this problem:

- Centralized Production - Project Investments
- Centralized Regulation - Command and Control
- Centralized Incentive Regulation - Market-Based Instruments
- Polycentric Regulated Markets - Cap and Trade Systems

Each of these is briefly described below along with the advantages and disadvantages of each approach

Project Investments

The centralized authority may choose to provide and produce the public good by simply cleaning up the pollution, in which case it simply needs to raise the funds – which it has the power to do – and proceed to the cleanup. This would of course require the authority to set the desired level of pollution and to expend necessary funds on cleanup. The obvious drawback here is that in many cases it will be less expensive to avoid, control or offset the pollution than to clean it up. The advantage is that the authority is in complete control of the entire process: decisions on targets, and raising and expending funds.

Command and Control

Command and control approaches consist of regulations placed on producers (in this case polluters) which direct them as to what level of pollution control technology they must install and use or what level of pollution they may emit. The advantage of this approach is that it recognizes that avoidance of pollution in the first place is likely to be cost-effective. Technology-based standards specify the methods and equipment that must be used to comply with environmental regulations. Such an approach eliminates the producers' incentive to find a low-cost solution to the problem and may lead to rent-seeking behaviour by government officials that promote a certain type of technology as a means of currying favour or funds from industry. Performance-based standards set uniform control targets for all regulated producers, but unlike technology-based standards the producers are given some choice over how the target is actually met. The advantage here is that the producers will likely minimize the costs of compliance as opposed to a technology-based standard.

Overall the advantage of command and control approaches is that they are a direct approach to resolving the problem (pollution). Both approaches also provide for clear expectations as to the total amount of pollution, with the performance-based standard having a small advantage as the total pollutant load is simply the sum of the performance standards. Disadvantages of the technology-based standards approach are the technology lock-in and for both approaches is the failure to take advantage of differing abatement costs. Even with the performance standard firms are choosing from amongst only their own internal options for pollution control, whereas arguable across most industries there will be a natural variability in the cost of abatement opportunities.

Market-based Instruments: Taxes and Payments

The classic case of regulation in environmental economics is that of a Pigouvian tax placed upon polluters. If the tax is formulated to reflect the marginal external costs of the pollution than society will in effect internalize these external costs and supply and demand will adjust from market levels to levels that produce an efficient allocation of societal resources. Such a tax then is an excellent means of achieving incentive compatibility. Similarly, if property rights rest with polluters and society desires better water quality then it may offer a subsidy or payment to polluters to produce the public good of water quality.

The advantage of using these instruments is that they make the opportunity cost of pollution clear to the polluter. In theory then the polluter will work to reduce his or her pollution to the point where the cost of another unit of pollution reduction is equal to the tax on the pollutant. The polluter thus will likely explore all the different means at his or her disposal to limit their effluent or emission. Another advantage of a tax or payment system is that it can be relatively simple to set up as the tax is set by the central authority – and therefore there is no uncertainty (at least within the current tax period) as to the price of pollution.

An important difficulty with such an approach is that it is an indirect approach to pollution targets and requires technical information to set the amount of the tax or payment so as to achieve an efficient level of pollution. This may also require adaptive management of the tax level over time so as to move towards the pollution target. This may be difficult to do where central authority sets prices administratively. Further, the availability of tax revenue or the need to find tax financing (for payments) can lead to political incentives to overtax polluters or under-tax the general public. And finally, there remains no incentive for those polluters for whom the cost of pollution control exceeds the tax to invest in pollution control. Instead they simply pay the tax and continue on polluting.

Cap and Trade Systems

A Cap and Trade system sets an aggregate rather than an individual cap on pollution (or resource use), and tradable allowances take the form of individual quota shares of the aggregate pollution cap. For example a system of marketable pollution permits involves setting the scale, distribution and allocation of permits in three steps:

- Determine an overall maximum level of pollution (the "cap")
- Assign available pollution permits to polluters
- Allow polluters to buy and sell pollution permits such that there pollution is equal to or less than the permits held

So-called ‘mitigation’ or ‘offset’ programs represent a slight expansion of the traditional marketable permit systems in that they provide for third party non-polluters to enter the pollution market with activities that offset the pollution and generate credits. These credits are then sold to polluters. Credits generated by offsets may represent the same authorized amount of pollution as a permit, but it is useful to consider them as conceptually distinct categories. Irregardless all three steps above apply, it is just that in mitigation programs the emphasis is typically on ‘no net loss’ – i.e. the overall cap is zero. In other words no increase in pollution is allowed – in effect all existing polluters are allocated permits to pollute equal to their current pollution and any new pollution needs to find credits to offset this new pollution, in order to achieve no net loss.

While cap and trade systems have as their primary objective holding pollution to a targeted level, once established they also may be used to lower the overall pollutant load. If third parties are allowed to purchase permits (or credits) then the price of permits will rise and the supply will be less plentiful leading to lower pollution levels. By monetizing pollution these systems allow for a market in not just pollution control, but ecosystem restoration to emerge. While this is always possible through direct funding of restoration project, the existence of a market for permits and offsets provides a higher likelihood that the cost of permits/credits will be minimized as polluters and third party providers all search for low cost solutions.

So the principle advantages of cap and trade systems are that they allow explicit setting of pollution targets and they minimize the cost of abatement. The related advantage they have is that they leave price-setting to the market, i.e. to buyers and sellers, and not to government officials (as with tax and subsidy instruments). The disadvantage of such systems is that they do leave buyers and sellers with price uncertainty, at least at program initiation. This can increase political resistance to such schemes by large institutional players and industry. The programs can also be complex to administer as monitoring, tracking and reporting programs are required to ensure that the program meets its targets and that the participants are following the rules. Still, these are largely up-front issues and on the current evidence a well-designed cap and trade system appears to offer a cost-effective approaches to pollution management (Freeman and Kolstad 2007).

This review of institutional arrangements and incentive mechanisms charts out the evolution that has occurred over the last forty years or so with regard to apply economic principles and methods to water quality management. Developed countries are themselves only now experimenting with the final step towards cap and trade systems. These approaches require technical information, legal and regulatory development, as well as trained human resources to run market systems. In developing regions, such as Africa it is often the case that even command and control approaches to regulating point source water pollution are still not in place of functioning adequately. It is therefore important to consider how far along this evolutionary path a country can venture before it has exceeded its capacity. Still, it may also be the case that a country can leapfrog one or more stages. For example if technical information and modelling capacity is a problem, enhanced technological capabilities developed in the US or Europe might well be readily transferred to other countries. It may also be the case that for some problems – like non-point source pollution only the more evolved approaches of payments and markets really hold the possibility of success and that as these are recognized as problems countries in the developing world can quickly adopt approaches that have been tested and proved elsewhere.

It should also be noted that this topic is discussed further below, specifically as regards definitional issues with respect to Payments for Watershed Services. There a Type 1 Payment for Watershed Services scheme (as defined in Section 5.1) is shown to be an additional, non-regulatory alternative for resolving upstream/downstream hydrological externalities discussed here.

4.2 Mitigation of Poor Water Quality and Avoidance of Health Impacts through Water Treatment

While household point of use disinfection, filtration or other treatment remains an option the standard approach to providing clean drinking water to households and industry is dominated by the installation of water supply and sanitation (WS&S) infrastructure. It is important to recognize that this infrastructure serves two purposes: the delivery of water and the treatment of this water. Therefore there are joint water quantity and water quality benefits from the capital and O&M costs incurred in these facilities. With regard to water quality there is a supply and sanitation component. As the interest in this paper is on how changes in ecosystem management in agricultural landscapes will affect domestic water quality the focus here is on the supply portion of the infrastructure and not the sanitation part. Sanitation is of course a vital part of the overall WS&S system, particularly as it becomes the cause of the next water quality problem downstream if not effectively implemented.

The treatment of water to remove contaminants and raise it to a standard suitable for domestic use and drinking can involve a number of different treatments undertaken in sequence. These include in the typical order in which they occur (WHO 2006; Steel and McGhee 1979):

- Pretreatment for the removal of sediment and material
- Sedimentation, flocculation and/or coagulation for the removal of sediment and biological pathogens through.
- Filtration
- Disinfection

Pretreatment occurs prior to the water entering a treatment plant. Water supply systems may be based on surface and groundwater or a mix of both. If a groundwater system or if the surface flow regime permits confined storage will be part of the distribution and treatment system. Otherwise, reservoirs may be used to smooth out water supply on a daily or seasonal basis. Such storage of surface waters may be used to lower turbidity and bacterial numbers but is not preferred as coagulation and filtration work more rapidly (Steel and McGhee 1979). The exception is where surface waters have extremely high turbidity. In such cases pretreatment without chemicals can settle out much of the suspended matter within 3 to 8 hours (Steel and McGhee 1979). Screens, bankside filtration, roughing filters and microstrainers are other approaches to pretreatment that may be used (WHO 2006; Steel and McGhee 1979). For systems drawing groundwater, wellhead protection includes installing sanitary seals, fencing wellhead area, removing surface water diversion ditches, ensuring quality of concrete works and controlling any wastewater drainage (Howard and Schmoll 2006).

The initial removal of sediment, pathogens and other material takes place in the water treatment plant using a combination of sedimentation, flocculation and coagulation. Sediment tanks are used within the treatment plant and consist of basins (of different shapes) and equipment for mechanically removing the sediment as it settles. Left unassisted settling times for smaller particles is too long for tank volumes, detention times and required flow rates. Thus, coagulation – the use of chemicals to destabilize these colloidal particles, and flocculation – the use of mixing techniques to promote the agglomeration of these particles is required. Chemicals used in coagulation include aluminium sulfate, ferric chloride and lime (Steel and McGhee 1979). Additional coagulant aids such as chorine, clay and activated silica may be used to promote flocculation (Steel and McGhee 1979). These chemicals and any labour required in their application are therefore an important variable input in water treatment. As suspended sediment in the input water rises so increased rates of application of these coagulants and aids is required.

Filtration provides the next level of water treatment for safety concerns, as well as improving colour, taste and odour. Filtration process include slow sand, granular, precoat and membrane filtration (WHO 2006). Filtration is of particular importance for removing microbial pathogens and, in particular, Cryptosporidium where chlorine is the only disinfectant employed.

The final step in comprehensive water treatment is disinfection, which serves as the final barrier to microbial pathogens. Chlorination is the most common disinfection method although ozonation, UV irradiation, chloramination and chlorine dioxide are also available. Disinfection is particularly important in removal of bacteria. Storing of water following disinfection may increase its effectiveness, particularly with respect to *Giardia* and some viruses. Maintenance of residual disinfectant in the storage and distribution system can be important to limit any microbial regrowth. Chlorination may be undertaken prior to coagulation as well as after filtration and is usually applied in amounts of 0.25 to 0.5 mg/l in order to leave a residual of 0.1 to 0.2 mg/l.

The extent to which each of these processes are deployed and the exact choice of technology will vary with source water quality and the economic capability of the water provider. Systems with access to good clean surface and ground water may simply require chlorination for example (Steel and McGhee 1979). For removal of protozoa (including Giardia and Cryptosporidium) coagulation/flocculation, filtration and disinfection are recommended (WHO 2006). However, as with any element of public safety such systems need to be cognizant of and plan for risk factors including extreme events, as for example a fire in a protected headwaters source area (Howard and Schmoll 2006).

As indicated in Section 3.3 the capital and O&M costs of water treatment will be just one component of overall water supply and sanitation costs. The extent of these costs will vary with source water quality and as systems are designed to include some or all of the treatment processes listed here. The figures cited from WHO suggest that water treatment costs in Africa will be half of total costs of full regulated supply or $6.60 per person per year (Hutton and Haller 2004). This is based on annual water treatment costs (capital and O&M) of $0.30/m^3 (for Africa and Latin America). The same study projects annualized water treatment costs of $0.20/m^3 for Asian countries.

By comparison in the UK study, Pretty et al (2000) include both capital and O&M costs in their estimates of the external costs imposed by agriculture on water treatment figures and come up with a figure that translates to $0.16/m^3 using an estimate of per capita household consumption from Scotland of 50 m^3/year (Moran and Dann 2007).

Variation in the quality of source water and the scale (and hence economies of scale) of treatment facilities will amongst other variables determine the capital and O&M costs in specific location. For example the Bureau of Reclamation provides detailed Fact Sheets laying out the capital and O&M costs of technologies for removing specific contaminants (Jurenka, Martella, and Rodriguez 2001). In addition

Reclamations WaTER program in conjunction with the National Institute for Standards and Technology provides an excel spreadsheet that provides cost estimates based on entries of production size and source water quality condition. With figures updated to 2006 the downloaded model suggests that a NF90 reverse osmosis membrane nanofiltration treatment system with a capacity of 378,500 m^3/day (100 mgd) would have a capital cost of $93 million and annual O&M of $17 million. The cost of filtration at an interest rate of 6% and amortization period of 30 years would be $0.18/$m^3$. To give a sense of scale, at approximate use rates for US cities this plant would serve a city of around 500,000 people. The NF90 is not by itself an entire water treatment system as depending on situational requirements it would be integrated with other technologies to fulfil additional treatment processes. Still at $48 a person per year this shows how higher water usage (7 times higher in these calculations for US water users) and the latest technology may increase the costs to users of water treatment.

What exact portion of these costs is as a result of poor ecosystem management and could therefore be reduced or eliminated by improved ecosystem management is a central question. As reviewed in Section 3.4 the literature on how treatment costs are affected by changes in source water quality is thin but yields relatively consistent figures. The more rigorous efforts focus on short-term changes in O&M as a result of erosion and/or turbidity and suggest that variable water treatment costs range from $0.02 to $0.05/$m^3$ as shown in Table 12. This would be around 10-20% of the total of all water treatment costs (using the WHO figures cited above). Using the higher value, for the resident of a developing country – consuming far less water each year than in the US (22 m^3 for example) – the net per capita cost would come to just $1/year. These figures are just a percentage of the full costs because they omit capital costs and, in some of the studies, merely examine a portion of variable costs – though presumably including the costs that influenced by water quality.

A further consideration here is that the studies referred to in Section 3.4 demonstrate that treatment costs are relatively unresponsive to changes in water quality (Dearmont, McCarl, and Tolman 1998; Forster, Bardos, and Southgate 1987; Holmes 1988). The elasticities shown in Table 12 suggest that a 1% change in turbidity or soil erosion appears to result in only around a 0.1% to 0.25% change in cost. In other words if turbidity was lowered by 10% costs would decrease only by 1 to 2.5%. For example if the costs per unit were $0.05/$m^3$ before turbidity was lowered in this fashion, they would be from $0.045/$m^3$. This would be a savings of $0.00045/$m^3$ to $0.00113/$m^3$. In other words, a significant reduction in suspended sediment would save only a tenth of a cent per cubic meter of delivered water.

Another way to look at the significance of these costs is in terms of their relevance in the watershed. Holmes (1988) calculates a damage of $0.033/ton of sediment. Assuming briefly that all soil erosion ends up as sediment and that a range of erosion rates of agricultural landscapes might vary from 5 to 100 tons/ha this suggests damage costs from erosion of between $0.165/ha to $3.33/ha. If only a portion of eroded soils makes it to the water outtake point then these figures would be lower still. As discussed further in the next section this would be unlikely to be a significant source of funding for improved land management practices in agricultural landscapes. In addition with low responsiveness of costs to turbidity (which is likely to be correlated with erosion), marginal changes in erosion would capture only a portion of these damage costs.

Short-run costs of changes in water quality linked to erosion and sedimentation would therefore seem to have only limited impacts on water treatment costs. Clearly the number of studies on which this is based is limited in number and in geographical range. However, it can be argued that the conclusion likely holds given the large number of water treatment plants included in these studies and that the physical processes and chemical engineering involved are unlikely to vary tremendously from a developed to a developing country context. Still, it would be reassuring to see such studies conducted in more erosion prone locations in order to see if the elasticity estimates do increase with the level of turbidity – as commented upon above.

However, water treatment costs – and health impacts – are not merely a function of erosion, sediment and turbidity, but also of chemical and biological threats to water quality. The Dearmont et al. (1998) study did model a proxy of chemical contamination finding that such contamination had an impact in the same range as the turbidity impacts (at $0.040/m^3). Beyond that however there is little in the empirical literature that assesses the economics of these threats.

A major factor in this discussion must also be whether long-run capital costs can truly be ignored in examining water quality and water treatment costs. In the extreme case, a pristine watershed might produce water that requires no water treatment whatsoever, or a minimum level of chlorination at a very low cost. If deforestation and unsustainable agricultural production were to occur, then capital investment in plant and equipment might be required in order to continue providing high quality domestic water supply. In such a case the full costs of water treatment (of $0.30/m^3/yr or thereabouts) would be attributable to the change in land use. However, this probably does not accurately capture most real-life situations. Further, this does not necessarily mean that it is worth taking an ecosystem approach. If the watershed is large, productive land uses very profitable and only a small population drawing their domestic water from the watershed the sum of potential damages may not be significant relative to the costs of protecting the watershed.

In many cases, contaminants from point or non-point sources do exist in the watershed and the capital investment in a water treatment plant is already in place. In this case, improving water quality may indeed lower O&M costs but will have not impact on capital investment (at least until the plant needs replacement). As it appears that these O&M costs are not very sensitive to changes in water quality the additional investment in improving water quality will make only a marginal difference in water treatment costs. Even if water quality improves the water treatment process must still be used as there will still be some (residual) level of contamination and therefore risk if water is not treated.

The alternative case is that there is a high quality water source and no treatment plant. In this case ensuring the protection of the water source may not be sufficient to eliminate the future need to incur the capital cost of a treatment plant. Instead it may merely postpone the date at which such an investment is necessary. If this were true there would be cost savings in the short-term but these might not necessarily be capitalized in perpetuity and, therefore, might not be available in order to make up-front or long-term investments in an ecosystem approach. Whether this would be true or not will depend on the ability to eliminate all risk of a water quality problem developing in the protected, high water quality source watershed.

Engaging in a system of payments for watershed protection with land managers in the watershed might be a way to maintain water quality, but it does not eliminate future risk of degradation in ecosystem functions. This, as such payments are likely to be conditional on performance and represent only temporary, renewable contracts in nature. Alternatively a municipality may invest in protecting the source watershed lands by purchasing the land or a perpetual conservation easement. These instruments can be used to permanently dedicate the land to a particular land use or management regime.

However, water quality regulations change over time as do the threats to drinking water quality. For example, in North America the recent emergence of *Cryptosporidium* as an important threat to drinking water and health has led to new regulations from US EPA and caused many cities relying on high quality natural sources to plan for new treatment plants. As *Cryptosporidium* is carried by wildlife it is quite consistent with the protection of intact natural ecosystems. Investment by a municipality in protecting source watershed lands through purchase or easement does not therefore eliminate future risk to water quality and the needs for improved water treatment facilities. (The New York City case explored in Section 5.3 is a case in point.) And once a decision is made to invest in treatment facilities it is likely that a treatment plant that can accommodate a variety of potential threats and risks will be installed.

The main point here is that health and safety concerns lead public and private water suppliers to err on the side of reducing risk as much as possible. This typically means investing in infrastructure that can successfully accommodate a variety of threats and worse case scenarios. This means that changes in a threat – such as changes in source water quality – lead only to a variation in O&M expenses, not the avoidance of the investment per se. Investments in physical capital are often substitutes for investments in natural capital. Where such investments are in place the impacts of changes in ecosystem services, such as water quality, may be of little consequence. Physical infrastructure has the added advantage of being perceived as eliminating risk whereas an ecosystem approach leaves the prospect of future threats open.

This analysis does not imply that the potential savings from manipulating water quality through an ecosystem approach are negligible or that an ecosystem approach is inherently unworkable. There are many cases where municipalities have already invested in source watershed protection (Ernst 2004). But it is important to explore the risks and limitations of an ecosystem approach as versus a physical infrastructure approach. As highlighted here the issue of what infrastructure already exists combined with an analysis of risk and irreversibility of investment may be important not only in the choice between infrastructure or ecosystems, but in the choice between what type of incentive mechanism can be best used in an ecosystem approach.

Still, the conventional approach at present is simply to treat the water at the point it enters the system (or at later points in the distribution system) so as to eliminate health risks. Understanding the health impacts of poor water quality is useful in choosing the level of pollution that is worth avoiding. However, the key question for this paper is whether ecosystem management is a cost-effective way of generating water quality benefits. While there remains more to be learned about the full impact of poor water quality on conventional water treatment approaches the evidence that does exist suggests that the costs of treating low quality water are relatively low where investments in treatment facilities have already been made. In cases where treatment facilities do not exist or provide only minimal treatment the costs appear to be much higher but their full extent will depend on the nature of water quality threats and the regulatory environment.

4.3 Avoidance through an Ecosystem Approach to Watershed Source Protection in Agricultural Landscapes

In the 2nd World Water Development Report, WHO and UNICEF state that 'the evidence base for associations between natural ecologies, biodiversity conservation and human health still requires substantial development' but then go on to conclude that 'health can be a key motivator in mobilizing communities to participate in nature conservation and environmental management' (WHO and UNICEF 2006) The literature reviewed so far in this paper makes it clear that while the economic framework needed to underpin the linkages between ecosystem management, water quality, domestic water systems and human health can be discerned, hard quantitative evidence of the economic significance of these linkages remains formative and not always supportive to the case at hand. This result supports the statement by WHO and UNICEF in the WWD Report but stands in contrast to the confidence expressed by others in the 'obvious' merits of taking an ecosystem approach.

For example, a recent report by Forest Trends to China on the lessons learned from international experience suggests as a primary conclusion that payments between downstream users and upstream watershed service providers are 'immediately applicable' (Scherr et al. 2006: iv). The report bases this call for action on later statements that there is a 'huge' demand for clean water and that 'investments in sustainable watershed management are often substantially cheaper than investments in new water supply and treatment facilities.' Unfortunately the evidence offered consists of unpublished papers not subject to peer review and funded by conservation organizations. As will be seen in Section 5.3 even the primary evidence offered, that of the New York City watershed story is not quite as it is related in the report. The Forest Trends report is probably accurate to state that already billions of dollars are being spent globally on water-related ecosystem services. However, this begs the question of whether these expenditures are achieving meaningful outcomes in terms of environmental and health objectives. Nor is it clear what portion of these funds are actually direct payments from downstream users to upstream providers.

The most thorough work to date on ecosystem services was undertaken by the Millennium Ecosystem Assessment. The Health Synthesis Report of the Assessment strongly asserts the role of ecosystems in human health but on the topic of water can muster little in the way of concrete linkages (Corvalan, Hales, and McMichael 2005). The report documents that the decrease in annual availability of freshwater per person (globally) from 16,800m^3 in 1950 to 6,800m^3 in 2000 was driven simply by the growth in population relative to a fixed (but renewable) supply. The widespread microbial contamination of drinking water is mentioned but is not linked to ecosystem degradation. The report does state that human pollution of the environment – including pollution from cultivated systems – impairs the ability of ecosystems to provide clean and reliable sources of water, but no statistics are available to qualify the statement (Corvalan, Hales, and McMichael 2005). The Director-General of WHO acknowledges in the foreword to the report that 'Nature's goods and services are the ultimate foundations of life and health, even though in modern societies this fundamental dependency may be indirect, displaced in space and time, and therefore poorly recognized' and goes on to note the need to 'look at environmental health through a broader lens' (Corvalan et al. 2005: iii). The Millennium Assessment thus appears to approve of the sentiment that an ecosystem approach is valuable but in the case of water quality and human health is likewise short of hard evidence to document and quantify the linkages involved.

Indeed, health and water quality professionals are not agnostic on the issue of whether upland ecosystem management should be a concern to those in the water supply and sanitation sector (and those concerned with drinking water and human health). In a rather large volume published by WHO and dedicated to groundwater drinking quantity there is an entire section on 'Understanding the Drinking-Water Catchment' and numerous references to the importance of watershed management in protecting groundwater quality (Schmoll et al. 2006). The volume also contains chapters on the relationship between agricultural management relates and groundwater quality, and the establishment of groundwater protection areas (Chave et al. 2006; Appleyard 2006).

As laid out earlier, the key question is one of cost-effectiveness. How cost-effective are measures to reduce water pollution from agricultural landscapes versus water treatment at the point of withdrawal and abstraction? To the extent that both approaches will improve water quality for domestic water supply they will both serve to alleviate the economic impacts on human health as described in Section 3.3. To foreshadow the conclusion reached below it is unlikely that there can be a general answer to the question of which measure is most cost-effective. This, given the site-specific nature of the economics of land use, agricultural production and water pollution. Rather, here an effort is made to summarize available information on the type, effectiveness and cost of the measures in order to understand the range of such costs and to see what the literature finds as to the factors that influence these costs.

But although cost-effectiveness will be an important indicator in considering and developing payment systems it will also be important to consider the joint benefits that an ecosystem approach will provide. While that is not the focus of this paper it is an important practical consideration that needs to be kept in mind in planning and implementation at the site level.

Best Management Practices

It is important to recognize that in any situation there will be a number of activities or practices that can be undertaken to improve water quality. Activities that are candidates for selection as part of a cost-effective program of water quality management are often called BMPs or best management practices (Guiling and St. John 2007). A FAO sourcebook identifies a wide range of potential methods for controlling water pollution, specifically for reducing erosion and sediment, fertilizers and pesticides (Ongley 1996). Generalizing across these categories there are two levels of methods identified. The first relates to the choice of arrangement and incentive mechanism, as previously outline in Section 4.1. For example, with regard to pesticides one method of reducing use is simply to ban the production or import of selected pesticides (Ongley 1996). This is a higher-level societal choice than that of the operational-level choice of which BMP to apply at the farm or watershed level. This paper is focussed on the use of PWS schemes and, therefore, is more concerned with the operational choices that face farmers and how these choices alter downstream water quality.

A final caveat is that the BMPs considered here are ones applicable to non-point sources rather than point sources. Point sources may also be important sources of poor water quality in agricultural landscapes and may well be worth including in PWS schemes. However, the technical side of BMPs for limiting point source pollution are rather more straightforward. Reducing the use of chemicals and or cleaning up facilities and treating effluent prior to discharge are by their very nature more direct (and technically less complex) approaches than managing diffuse sources across a large agricultural landscape. It is also true that setting up a PWS for point as versus non-point sources will also be correspondingly less complex an endeavor. For this reason the emphasis here is on non-point source control.

Based on a number of sources the FAO sourcebook lists a wide range of BMPs in agricultural landscapes that address water quality problems (Ongley 1996):

- Avoid over-use of fertilizers, pesticides and irrigation through:
  - Management planning
  - Training and licensing of individuals applying pesticides (and fertilizers)
  - Testing and approval of equipment for applications
  - Improved scheduling of applications
  - Requiring maintenance of records regarding applications
  - Limiting aerial applications
  - Adoption of mechanical and biological alternatives including organic agriculture
- Avoid or reduce soil erosion, runoff and reduce nutrient leaching through:
  - protect retired, highly erodible or fallowed land with conservation cover, perennial vegetation and green manure crop
  - increase organic matter through conservation cropping and tillage
  - terracing and contour farming
  - use crop residue to protect fields during critical erosion periods
  - delay ploughing of debris into soil and seedbed preparation
  - strip cropping

- sediment basins and infiltration trenches/diversions
- buffer strips, field borders and filter strips

- Avoid stockpiling and leaking/dumping of excess pesticides
- Reduce or eliminate fertilizer and pesticide applications in critical areas
- Limit stocking rates based on acceptable levels of manure
- Limit use of sewage sludge and manure from confined animal feeding operation
- Manage composition of feed to reduce nutrient content of manure

Clearly there are a wide range of possible BMPs for water quality improvements and their outcomes and their direct and opportunity costs will vary tremendously from one situation to the next. While site-specific analyses are therefore required it is also important to recognize that these BMPs will have additional impacts on other natural resources, ecosystem services and biodiversity. It is therefore essential in the planning of projects or incentive systems in specific locations to take an integrated (or holistic) approach to natural resource and ecosystem management. That said, it remains useful to continue to hone in on the specific water quality impacts and costs of most concern in this paper.

Another important omission to notice is that much of the literature on agricultural BMPs for water quality does not raise the issue of biological pathogens. As describe in Section 2 of this paper this aspect of the water quality problem does appear important if not critical for domestic water supply. The example of dysentery and Shigella illustrates the difficulty of clearly defining the link between land and water management in agriculture, downstream water quality and health impacts. For example, if 90% of agricultural lands were managed so as not to provoke an outbreak of these diseases, the risk from the remaining 10% might be sufficient for authorities to recommend water treatment. To some extent suspended sediment is indicative of biological activity and so the removal of suspended sediment will lower the risk from biological pathogens (Steel and McGhee 1979). Further consideration, however, is needed as to how agricultural BMPs affect biological pathogens. It may be that the response in terms of water treatment operations may be of a different character with respect to the proportional application of BMPs and resulting reductions in downstream nutrient concentrations as versus the same for biological pathogens. The relationship between water quality at the intake, the costs of water treatment and the risk of health impacts may be more linear with respect to chemical contaminants than biological contaminants.

There are two steps in calculating the cost-effectiveness of BMPs. The first step is specifying the relationship between the BMP and specific water quality parameters. This will involve estimation of the loss of parameters of interest (e.g. Nitrogen, Phosphorus, sediment) under existing and BMP management, as well as delivery ratios to points of impact downstream. A number of models suited to this purpose exists as described briefly below. The second step is to calculate the cost of the BMP (also described further below). Of course the relationships involved may not be linear so that the relationship between level of investment in the BMP and water quality outcome will not be constant as the investment level changes.

Models and Tools for Estimating Non-Point Source Water Quality Improvements from BMPs

There are a large number of models, techniques and tools linking ecosystem and watershed processes with water quality (for example see the Register of Ecological Models online database). These models typically involve an integration of simulation models and GIS technology so as to provide decision support systems that provide for spatial interpretation of cause and effect. Many simulation models exist. A few examples of such models include those that simulate soil erosion (USLE, RUSLE, WEPP, SDE), nitrogen (NLEAP, ANIMO), and pesticides (RICEWQ, VARLEACH). A number of models exist for modeling one or more pollutants, typically at the field (CREAMS, GLEAMS, LEACHM) or watershed scale (AGNPS, SCUAF, BASINS, SWAT). A useful summary of a number of key elements of agricultural non-point source models is provided in Table 13.

These models may be turned to different purposes. A general application is to simulate how changes in BMPs lead to changes in specific water quality parameters downstream. This can assist in targeting where in a watershed BMPS may be employed so as to obtain the best outcomes. Other models are used to define the maximum allowable load of nutrients for sustainable source water quality (TMDL Models) and are employed across the US in implementation of the Clean Water Act. New applications emerge although the core elements are typically established simulation models or simulation models integrated with GIS. For example, WRI has developed an online estimator for nitrogen and sediment and USDA-NRCS is developing a Nitrogen Trading Tool that relies on NLEAP an existing nutrient model (Lal 2007; Guiling and St. John 2007).

The proliferation of models for examining the environmental impacts of BMPs is heartening. Still improvement is needed. The World Resources Institutes suggests three ways that further investment could improve the estimation of these outcomes (Guiling and St. John 2007):

- increasing site-specific research on estimating environmental outcomes, as well as other related environmental benefits produced by the BMP
- development of a monitoring framework that would allow the validation of estimation methodologies and the testing of their accuracy
- creation of a central repository of estimation methodologies and monitoring data

The USDA's Conservation Effects Assessment Project (CEAP) is mooted by WRI as a potential candidate for the monitoring framework and, if expanded, could fill the other missing functions as well. Consideration would be required of how to achieve these objectives at a global rather than just a national scale. Scientists at the USDA have also recently developed a database of studies measuring nutrient load data in the United States (Harmel et al. 2005). The MANAGE database – an activity affiliated with CEAP was initiated with over 160 records from 40 published studies representing over 1,100 watershed years of data. The data came from farms using conventional practices as well as conservation practices. Even with such a large data set the different crop types and differing site-specific geomorphological characteristics meant that little in the way of firm conclusions could be taken from the meta dataset in terms of the relationship between tillage and conservation practices and nutrient loads – even though strong results were found in some of the individual studies themselves (Harmel et al. 2005)

Table 13. Agricultural Non-Point Source Models

NAME	APPLICATION	TIME SCALE	SPATIAL SCALE
A. Low to medium data needs			
Unit area loads (statistical prediction)	Sediment loss□Nutrient loss	Long-term averages	10's to 100's km2
NOTE: Statistical models use aggregated data for comparable conditions. Predictive power is low but can be useful for screening purposes or where no field data are available; or where the spatial scale is so large that field data are uneconomical.			
USLE (Universal Soil Loss Equation)	Average soil loss for specific crops, etc.	Annual	Plot/field
RUSLE/MUSLE□(Revised/Modified USLE)	Average soil loss for specific crops, etc.	Annual	Plot/field
NOTE: Empirical USLE-type models have been applied to large area analysis, using remote sensing data, etc. for regional estimates of soil loss (e.g. Brazil). USLE-type models are often incorporated into more detailed hydrological models below.			
B. Data intensive modeling (process-oriented)			
ACTMO (Agricultural Chemical Transport Model)	Hydrologic processes□Water quality	Event, continuous	Field
AGNPS (Agricultural Non-point Source Pollution)	Hydrology, erosion, N, P and pesticides	Event, daily, continuous	Grid cell, field scale
ANSWERS (Areal Non-point Source Watershed Environment Response Simulation)	Hydrology, erosion, N P and pesticides	Single storm	Grid cell
CREAMS (Chemical, Runoff and Erosion from Agric. Management Systems)	Hydrology, erosion, N, P and pesticides	Daily, continuous	Field scale
EPIC (Erosion-Productivity Impact Calculator)	Hydrology, erosion, nutrient cycling, crop and soil management and economics	Event, daily, continuous	Field scale
HPSF (Hydrologic Simulation Program-Fortran)	Hydrology, water quality for conventional and toxic organic pollutants	Event, daily, continuous	Watershed
SHE (Système Hydrologique Européen)	Hydrology, with water quality modules	Event, daily, continuous	Watershed
SWAM (Small Watershed Model)	Hydrologic processes, sediment, nutrients and pesticides	Daily, continuous	Watershed
SWAT (Soil and Water Assessment Tool	Hydrologic processes, sediment, nutrients and pesticides	Event, daily, continuous	Simultaneous simulation for hundreds of sub-basins
SWRRB (Simulator for Water Resources in Rural Basins)	Water balance and hydrologic processes and sedimentation	Event, daily, continuous	Watershed
WEPP (Water Erosion Prediction Project)	Hydrologic processes, sediment processes	Single storm, daily, continuous	Hillslope, watershed, grid cell

Source: Ongley (1996)

The use of models to examine potential water quality benefits of widespread application of BMPs at the regional or national scale is a very large undertaking. Recently, scientists from the USDA completed an assessment of the potential water quality gains from a limited set of the BMPs in listed above. The study simulated conventional and alternative practices on a large percentage of US croplands. The results for tillage suggested that existing use of mulch tillage (21% of sample) and no-till (21% of sample) has had the following water quality benefits over a scenario in which 100% of the sample is conventional tillage:

- 32 percent reduction in sediment loss (0.8 ton/a/yr or 0.3 tons/ha/yr reduction, on average)
- 26 percent reduction in wind erosion rates (0.3 ton/a/yr or 0.11/tons/ha/yr reduction, on average)
- 7 percent reduction in nitrogen loss (3.2 lb/a/yr or 0.59 kgs/ha/yr reduction, on average)
- 3 percent reduction in phosphorus loss (0.4 lb/a/yr or 0.07 kgs/ha/yr reduction, on average)

The results for conservation practices, including contour farming, stripcropping, and terracing (found currently on 10% of lands or 13 million hectares) suggested that the gains from these practices produced the following water quality benefits:

- sediment loss was reduced 54 percent (1.8 ton/a/yr or 0.66 tons/ha/yr reduction, on average)
- nitrogen loss was reduced 6 percent (7 lb/a/yr or 1.3 kgs/ha/yr reduction, on average)
- phosphorus loss was reduced 28 percent (1 lb/a/yr or 0.18 kgs/ha/yr reduction, on average)

The potential changes in erosion and sediment loss are substantial. Interestingly these BMPS had a proportionately less significant impact on nutrient loss. Clearly issues remain in the measurement, modelling and monitoring of the effects of agricultural BMPs on water quality. Nonetheless, the discussion above highlights that there exist a host of studies, protocols and methods that can be brought to bear on this problem in a given site.

Cost-Effectiveness of BMPs

The economic analysis of BMPs involves two principle types of costs: direct costs and opportunity costs. Each BMP will carry with it certain capital and variable running cost related to actual implementation of the BMP. Likewise a given BMP may impose an opportunity cost on farmers as they change from their prior production practices to the BMP. For example, the implementation of buffer strips may reduce the extent of arable land on the farmer's property and therefore reduce production and income. Obviously, the ideal BMP is one that has low direct costs and low (or negative) opportunity costs. Negative opportunity costs imply that with the BMP the farmer actually realizes net increase in farming returns, either through production cost-savings or through higher production or higher prices for farm output.

No general conclusions on the effectiveness of specific BMPs can be made at this point, particularly given the objective of this paper in exploring how PWS schemes would fare in Africa and the difficulty of transferring developed country estimates to developing countries. Nonetheless, a number of case studies have examined the cost-effectiveness (and in some case the economic benefits) of BMPs in specific places. These are reviewed below to provide an initial indication of how these BMPs fare in terms of water quality improvements and costs and what issues arise.

A study in the Mississippi River Valley examined the cost-effectiveness of controlling nutrient loads through two methods: reducing fertilizer use or using wetlands as filters (Ribaudo et al. 2001). The choice of wetlands was motivated by the finding that wetlands far outperformed buffer strips in terms of nitrogen reductions (wetlands remove 10-20 g N/m2/yr). The fertilizer standard was found to be more efficient than wetlands up to a particular point of total reduction in nutrient load, at which point the capital restoration and easement costs of wetlands are overcome and wetlands are more efficient.

Although the cost-effectiveness curves are not linear a few indicative point results include (Ribaudo et al. 2001):

- a loss of $109 million ($124 million in 2006 dollars) in net social welfare accompanies a 244 thousand tonnes (10%) reduction in fertilizer user for a cost-effectiveness of $0.0005/g N
- restoring one million hectares of wetland leads to 97 thousand tonnes of reduction at a cost of $1 billion ($1.14 billion) or $0.011/g N.
- the point at which the cost-effectiveness turns to favoring wetland restoration is at a reduction of 1,250 tonnes (or 25%) of nutrient loading or a rough cost-effectiveness of $0.06/g N

The important point made by this study is that in a given location the choice of cost-effectiveness will vary according with the scale of the objective. Also, to reach a given pollution reduction target it is likely that a number of BMPs may come into play as their scale-dependent cost-effectiveness come into play. So it is not even a question of which BMP is the most cost-effective put rather what is the least-cost package of BMPs that will enable reaching a given pollution reduction target.

Another interesting study this time from northeast Netherlands examines the cost-effectiveness of BMPs to reduce phosphorus loading and lake eutrophication (Hein 2006). The results show that due to thresholds between loads and eutrophication there is no cost-effective solution between leaving the situation as is (with eutrophication) and investing in sufficient remediation to reach the clear water threshold. In the latter case an investment of 5 million Euros is required to reduce phosphorus loading of 3 tons P/year or 1.3 Euros/g P ($1.63/g P in 2006 dollars). In comparison Faeth (2000) found a range of from $0.015/g to $0.042/g (in 2006 dollars) for phosphorous reduction from mulch tillage, no-till and nutrient management BMPs in the USA. Note that Hein's result relies on a rough assumption about the potential benefits of eliminating eutrophication so the results are not strictly speaking a cost-effectiveness analysis. This study does however suggest the potential importance of accounting for threshold effects when the impact of water pollution on domestic water quality is routed through intermediate impacts in terms of eutrophication.

A study of a small watershed in the northeast of France, simulates six nutrient management scenarios each characterised by a set of conventional and BMP farm practices using a biophysical model that reflects variability in climate (Lacroix, Beaudoin, and Makowski 2005). The scenarios included a set-aside of marginal land, reduction of fertilizer by 20% below the agronomic optimum, the sowing of post-harvest 'catch' crops and 'integrated fertilization,' which involves limiting fertilizer use in vulnerable zones. The following results are noteworthy:

- costs in all scenarios represent less than 10% of crop gross margin, with the exception of set-aside scenario where a set-aside of 17% of cropped area led to a cost equal to 48% of the gross margin
- the costs of the scenarios with reduction in fertilizer use are the most costly (after set asides) due to loss in production value and combining the use of 'catch' crops with integrated fertilization is only slightly less costly
- integrated fertilization is by far the least costly method but generates the least amount of pollution reduction when compared to conventional practices
- when variability in climate is included in the modelling none of the practices achieve a 100% probability of reaching the European concentration standard of 50 mgNO3/l/yr in the short run.

- integrated fertilization with early sowing of catch crops is the optimal approach in the long run

The conclusion of the study is that standards can only be met over the longer-term. Interestingly Lacroix et al (2005) go on to compare the costs of the optimal BMP with that of standard water treatment costs for nitrates. Citing average French water treatment costs (capital and O&M) of from 0.27 to 0.5 Euros/m^3 (\$0.34 to \$0.64/m^3 in 2006 dollars) the authors suggest that the cost of the optimal scenario in their study is 0.06 to 0.08 Euros/m^3 (\$0.076 to \$0.10/m^3) for nitrate pollution. The authors cite a few other French studies that have produced similar results showing that in the long run agricultural BMPs are cheaper than water treatment (Lacroix, Beaudoin, and Makowski 2005). While the comparison is very useful for the purposes of this paper and the margin is substantial between the cost of the two approaches as cited it needs to be questioned whether it is correct to assume that the entire cost of water treatment should be compared with just the application of BMPs to nitrate pollution reduction. Further, the lack of any presentation of actual cost-effectiveness numbers in the study or the inclusion of the method by which the reductions were converted into equivalent per cubic meter figures makes it difficult to verify the results.

Another European study examines the impact of lowering livestock density in the Rhine on nitrogen levels (Gomann et al. 2005). The study found the costs to be 20 Euros per kg N (\$0.025/g N). However, an alternative policy, taxing nitrogen use was found to be 20 times more cost-effective, as limiting livestock to 1 livestock unit per hectare let to large percentage reductions in stocking rates (up to 80%) dramatically affecting opportunity costs. Long-term costs of reducing nitrogen loads by one-third in the Ems basin come to 9.5 Euros per kg N (\$0.011 per g N). Even with these reductions the load would still exceed the new European standards.

The study by Gomann et al. (2005) not only examines cost of operational-level BMPs but also explores the cost-effectiveness across different policy-level choices as to how to implement non-point source reduction. This raises the bar from a simple comparison of on-the-ground BMPs to the higher-level question of how best to regulate and control agricultural water pollution (as discussed in Section 4.1). In some cases or locales, the choice may be between payments to farmers for implementing BMPs, or an entirely different method (bans, taxes, subsidy removal, trading) of creating the necessary incentive compatibility (Branca, Cory, and Monke 2004).

For example, in a study of methods for reducing nitrate water pollution in Texas, USA it was found that incentives for modernizing irrigation systems (and reducing water use) were more cost-effective for farmers (in terms of farm income) and society (in term of use of public funds than restrictions on per acre use of nitrogen, taxes on nitrogen use or taxes on water use (Wu et al. 1995). Growing evidence supports the contention that use more sophisticated incentive mechanisms will achieve higher levels of efficiency (Freeman and Kolstadt 2007). Ribaudo et al (2001) report on studies that confirm that for non-point sources performance-based policies (i.e. specific nutrient concentration targets) are less effective those design-based policies that provide incentives to change specific practices. In particular the practical cost and difficulty in directly observing performance (the reduction in nutrient levels) at the farm level is prohibitive.

A study of three midwestern watersheds compares different policy-level options and finds that trading with performance based conservation subsidies is considerably less expensive than a conventional program of subsidies for agricultural BMPs (Faeth 2000). This is due to the tendency of such subsidy programs to fail to be cost minimizing. Setting agricultural BMPs in a trading framework where point source polluters are seeking least-cost solutions achieves considerable cost-efficiency. An important question remains as to the feasibility of performance-based systems and this will be taken up in the next section where the application of PWS to agricultural water pollution and domestic water supply is taken up.

5. Payments for Watershed Services

It is often useful to clearly define terms in environmental management, particularly when referring to tools that are new and quite popular. Following a review of definitional issues a definition of PWS is proposed with two variants so as to be inclusive of observed experience in the field. Examples of PWS cases relevant to water quality provision in agricultural landscapes are then summarized.

5.1 Definition of PWS

Payments for Watershed Services is a name that has come to refer generally to efforts to internalize the external effects of upstream-downstream hydrological effects of ecosystem management (the management of land and associated resources). The general idea being that a downstream use of watershed services pays the upstream producer and therefore achieves incentive compatibility. A more precise, agreed-upon definition probably does not exist. This lack of a definition is not unique to PWS – as of 2005 the term 'Payments for Environmental Services', was also un-defined in the literature (Wunder 2005). At an expert meeting on PWS held in Bellagio in March of 2007 a group of PWS experts and practitioners began debating a precise definition. The definition mooted was (more or less) as follows: a PWS is the voluntary exchange of money or in-kind goods and services from a downstream user of watershed services to an upstream producer for the purpose of improving downstream watershed services, often facilitated through an intermediary. This led to much discussion with the result that the group decided it did not have time to try to reach consensus on a specific definition. A number of the points of potential disagreement are discussed below.

Voluntary Exchange? Including a requirement that payments be entered into voluntarily by buyers would limit the definition of such programs as it would rule out programs that are funded through taxes and fees, whether direct or indirect. In these cases funds are sourced administratively according to a formula rather than through a voluntarily negotiated arrangement between buyer and seller or between buyers, sellers an an intermediary. However, PWS involves the term 'payments' not 'markets' and therefore the idea of each and every buyer voluntarily purchasing services is not essential. In point of fact any direct fee – such as a mandatory surcharge on a water bill – or indirect tax – such as revenues sources from consumption or income taxes is ultimately subject to the discretion of the payer. So that such payments be voluntary is not an essential part of a broad definition. Rather is it an important caveat, in that payments voluntarily agreed to be buyers are not subject to the same regrets as payments determined and assessed by an agency, utility or other tax-generating authority. A hydropower company that makes a payment and ultimately does not receive the expected services has no one to blame but itself (and its consultant scientists). A household that makes payments to a utility that receives no benefit from subsequent payments to landowners may feel unjustly done by and seek redress.

Payment or Incentive? The concept of 'payment' may be limited simply to a cash payment or run the gamut through to any type of incentive that promotes incentive compatibility. Including at a minimum 'payments' that take the form of in-kind goods and services' is logical as the seller is still being paid, its just not in money per se. There may be good reasons, particularly in less developed settings, where an affluent downstream buyer may want to compensate upstream communities in goods or services, rather than in cash. This would be particularly true if cash payments might lead to corrupt behaviour due to its fungible nature. In a Winrock International project in India, participating communities were compensated with saplings for planting in the watershed (Agarwal 2007). In the Working for Water Program in South Africa, contract labor is supplied to remove alien invader species from the property of participating landowners (Marais 2007).

However, to the extent that 'payment' refers to the compensation that internalizes an externality it is not that much more of a stretch to expand the concept of a 'payment' to that of reward or incentive. In this case more creative financial arrangements such as tax credits and property rights would qualify – for example a program could 'reward' land managers with more secure property right or land tenure. The difficulty is that the term incentive can mean both positive and negative incentives, whereas PWS clearly refers to positive incentives only. It is probably best therefore to stick with 'payments' as the term but define 'goods and services' broadly. Obtaining a property deed for example would be equivalent to obtaining a 'good' and a tax credit is simply a form of cash payment.

Watershed or Hydrological Services? Services should be understood here as shorthand for goods and services in the economic sense. This poses a problem as the economic services provided by watersheds could potentially include the value of all ecosystem services and biodiversity. However the general intent of PWS as focussing on services associated with the hydrological cycle or hydrological function is clear. This is particularly true given the common use of the acronym PES to refer to variously payments for environmental or ecosystem services. Therefore hydrological services would be a more precise phrasing of the sense of watershed services as used in PWS. Unfortunately, this point is from time to time forgotten or mischaracterized, but generally the term PWS is in circulation and it should simply be recognized to apply to hydrological services of ecosystems.

Buyer a Downstream Beneficiary? Whether the buyer is the downstream beneficiary or whether the funds originate with a downstream beneficiary raises again whether there must be a direct connection between the recipient of the hydrological service and the payment that is rendered to the upstream buyer. If a third party pays upstream landowners to produce hydrological services this is clearly a payment for watershed services. Has an externality been internalized? It might seem not since the beneficiary has made no contribution to the payment. Has incentive compatibility been achieved? For the upstream landowner the payment does provide the proper incentive to produce socially desirable levels of watershed services. For the downstream beneficiary though the true external cost is not realized as the third party has made the payment. The downstream beneficiary will see a lower, 'subsidized' price for the service and tend to over-consume. Thus this does not really achieve incentive compatibility and thus is not true to the general sense of PWS Still this may be largely an academic distinction so it might be useful to posit two types of PWS – one where the beneficiary is the buyer and one where the buyer may only be a beneficiary in some indirect or diffuse way.

Do Intermediaries Matter? The short answer is that intermediaries are just a feature of differentiated production and will emerge as dictated by transaction costs. Whether they are present or not is not a key element of whether an approach is a PWS or not. However, it does not hurt to be clear that intermediaries may be present. Interestingly, the larger the scale of the PWS and the more diffuse the connection between the payee and the beneficiary the more likely the need for an intermediary.

Taking this discussion into consideration two variants of a working definition of PWS for this paper can be offered[1]. Type 1 PWS demonstrate a direct linkage between the source of the payment and the beneficiary. Type 1 PWS would be 'pure' PWS. The definition would be that a PWS is the provision of a payment of money or in-kind goods and services from a downstream user of watershed services (understood as hydrological services) to an upstream producer for the purpose of improving downstream

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[1] In the context of the PESAL project, for which this paper was prepared, a broader definition is used: "PES transactions refer to voluntary transactions where a service provider is paid by, or on behalf of, service beneficiaries for agricultural land, forest, coastal or marine management practices that are expected to result in continued or improved service provision beyond what would have been provided without the payment."

watershed services, which may be facilitated by an intermediary. In other words, the externality is directly internalized between the beneficiary and the land manager.

A Type 2 PWS is one where the connection between the source of funds and the beneficiary is indirect. In Type 2 PWS the payment is serving the purpose of internalizing the externality for the upstream land manager but not for the downstream beneficiary. A definition of Type 2 PWS would be the provision of a payment or in-kind goods and services to an upstream producer of watershed services (understood as hydrological services) for the purpose of improving said services, likely facilitated by an intermediary. In other words, society at large internalizes the upstream externality on behalf of the beneficiary.

A further observation is that as the linkage between the source of the payment and the beneficiary becomes less direct the more likely an intermediary will be involved. Where the linkage is direct and immediate the need for an intermediary is less likely.

What both Type 1 or 2 PWS have in common is that they provide direct incentives to land managers to alter their expected behaviour. PWS are therefore either payments to land managers not to do something (that they otherwise would have absent the payment) or they are payments to do something (that otherwise would not have happened). In this manner such payments create additional benefits over the status quo and thus fulfil the additionality criterion typically expected of PES schemes. Note that this implies that the land manager who is not managing the land so as to avoid downstream externalities or the land manager that is considering such a course of action effectively are acknowledged as having the 'right to pollute' in this case.

Other typologies or characterizations have been offered. A number of observers have defined 'Markets for Ecosystem (or Water) Services' or 'Payments for Watershed Services' as consisting of three types of 'deals' (Powell, White, and Landell-Mills 2002; Johnson, White, and Perrot-Maitre 2001; Mulder, Tassone, and Perrot-Maître 2006):

- Self-organized private deals
- Trading schemes
- Public payment schemes

The distinction between private deals and public schemes is alternately attributed to a difference in ownership and in level of government involvement (Powell, White, and Landell-Mills 2002). But this distinction may not be as simple as it seems. For example, Mulder et al. (2006) describe the Heredia, Costa Rica PWS as a private payment scheme. In this scheme, Empresa Servicios Publicos de Heredia (ESPH), a utility providing water and other services, charges its water ratepayers a hydrologic tariff. The funds obtained are then expended on forest conservation and BMPs in the watershed above the water source. In fact ESPH is a parastatal organization (an autonomous public institution created under Costa Rican law) and not a water utility owned by a private company or individual (Kosoy et al. 2007). ESPH ratepayers are the ones that pay the hydrologic tariff and they are indeed private parties. But the tariff is mandatory not voluntary and ESPH management, not the ratepayers, made the decision to undertake the PWS. So is this a private deal or a public scheme? The larger question is does it really matter? Does resolving the question in the case of the Heredia case provide any useful information for categorizing the deal (or the scheme)?

It would appear that whether the entity engaging in a PWS is private or public is not as important an element of characterization of PWS as whether the buyer is the beneficiary or some other stakeholder or entity. For this bears on the fundamental economic issue involved in PWS of how the externality is being internalized. It seems more useful to focus on the linkage between the externality and how it is internalized rather than by whom.

The term 'trading schemes' – as in a water quality trading scheme – is also used in setting forth a markets and payments typology (Powell, White, and Landell-Mills 2002; Johnson, White, and Perrot-Maitre 2001; Mulder, Tassone, and Perrot-Maître 2006). Such schemes are the same as the cap and trade systems as explained in 4.1. The defining feature of a cap and trade system is the cap, which places the burden of further use or pollution on the users or polluter. In a cap and trade system for water quality that allows offsets, point sources polluters would pay non-point source polluters (in this case land mangers) to implement BMPs in order to offset point source pollution. Thus cap and trade systems are an extension of the 'polluter pays principle'. Placing the burden of payment on the polluter is achieved through regulation – i.e. government action. This is conceptually distinct from the central idea underpinning PWS schemes, as elaborated above, that the polluter is paid not to pollute by the downstream party that bears the cost of the pollution. While offset systems do involve a payment to a land manager for reducing pollution these are not actually for the purpose of internalizing the land managers pollution, but rather that of the non point source polluter. For this reason, 'trading schemes' are really not examples of PWS as defined above at all. Rather they are simply cap and trade systems that can be employed to limit and reduce pollution and or resource use. They are an example of markets created by regulation.

Another type of PWS mooted in the literature is certification schemes (Mulder et al. 2006). In these schemes, consumers of products such as coffee or salmon pay extra for the product based on product labelling information certifying that the raw material was produced by farmers that are simultaneously providing hydrological services. Here the externality of downstream hydrological impacts of land use is being internalized through the payment, it is just that the buyer is not the beneficiary. Thus, certification schemes fit the criteria for a Type 2 PWS. They are just another way to fund the provision of hydrological services.

Given this discussion it might be asked if PWS schemes represent a new tool in achieving incentive compatibility? Within the context of the different policy-level strategies for managing water quality outlined in Section 4.1, PWS Type 2 fits clearly in the category of market-based instruments. Typically such instruments are employed by society as a whole to encourage or discourage use of a resource (or pollution) using taxes or subsidies. Type 2 PWS are simply payments by society (or a portion thereof) to economic actors to generate positive externalities. These positive externalities are then consumed by another portion of society. Just as the production of ethanol might be subsidized by government in order to provide benefits in terms of reduced carbon emissions and energy security, so government subsidizes the production of a set of hydrological services that may benefit municipalities, irrigators, ecosystems and other water users.

What distinguishes PWS Type 1 from PWS Type 2 and the typical market-based instrument is the direct nature of the relationship between the beneficiary and the land manager. So PWS Type 1 are not so much subsidies aimed at producing positive externalities for society but rather payments by those experiencing (or facing the possibility of experiencing) negative externalities. And these payments are made to those holding property rights to reduce these externalities (or to postpone them).

This raises the distinction between a payment and a subsidy. Conventional rhetoric amongst those working in the field holds that the use of the term 'payments' in PWS or PES is distinct from a subsidy because the funds are going to purchase something specific. However, this ignores the point that a subsidy is also offered in the hopes of acquiring some benefit. The benefit may be somewhat diffuse, as in energy or food security, but can also be very specific, such as health care subsidies. Political economy dictates that subsidies are not provided for no gain whatsoever. While it is true that some subsidies do not produce positive benefits for society, but simply are a means for acquiring votes (as in the concept of 'pork-barrel spending') the general concept of a subsidy as a market-based instrument is to provide a positive externality. So it would seem that a more useful definition of a payment system is one where the funding is provided by a buyer to a seller in exchange for the rendering of a service. So, Type 2 PWS would be correctly characterized as subsidies. In other words the key distinction between a payment and a subsidy is not the existence of the benefit but the degree to which the entity providing the funding is the entity receiving the benefit.

But what about Type 1 PWS, where the buyer is clearly the beneficiary? Is this a new concept or a new tool? Certainly it is not reflected in the policy-level approaches to managing water quality outlined in Section 4.1. This is because these approaches are all policies that describe how government can regulate the production of water quality. Pollution is typically considered a public good that is not amenable to provision by the market but that requires collective action, in other words government regulation of some kind. However, economists have long acknowledged that in certain situations the recipient of pollution and the polluter will be able to negotiate an arrangement that leads to a reduction in the level of pollution (Coase 1960). Coase's Theorem suggests that regardless of the original allocation of property rights between the two parties such an arrangement will emerge, subject only to transaction costs. In other words if the costs of arriving at a transaction exceed the net benefits of such an arrangement then the parties will be better of living with existing level of pollution and doing nothing. If the transaction costs are low, the parties can negotiate and enter into an agreement and still generate net benefits. Such a voluntary arrangement is called a 'Coasian Bargain.'

Some economists have taken Coase's Theorem as suggesting that pollution (and other natural resource and environmental problems) can be solved by simply assigning and enforcing property rights (Anderson and Leal 2001). Externalities therefore are eliminated as agreements are reached for 'efficient' levels of pollution. However, the problem of exclusion and rivalry remain, as fundamental defining features of public goods and, thus, market failure and collective action remain a necessity (Randall 1983, 1988). Where a Coasian bargain occurs, such as in a Type 1 PWS, residual market failure may remain and might still be subject to government regulation. Note that this is not a matter of 'public' involvement as a public drinking water agency may in fact enter into a Coasian Bargain. It is therefore not a matter of whether the entity is private or public but rather that the PWS is entered into on a voluntary basis, without requiring collective action or government regulation. In other words Coasian Bargains are examples of unregulated markets. PWS Type 2 then are simply an example of a Coasian Bargain and the unique circumstances where markets can resolve environmental problems without regulation.

It is also worth recognizing that in the case of upstream/downstream hydrological services there are at least two forms that a Coasian Bargain may take. The downstream buyer has a choice between paying an upstream land manager to provide the services or acquiring the upstream property so as to undertake provision directly (Déprés, Grolleau, and Mzoughi Undated). By in essence undertaking vertical integration of the upstream and downstream portions of the provision of hydrological services the downstream buyer may also solve the externality problem. From an efficiency point of view this may not be the best alternative and in some cases may not be feasible for a variety of reasons. The difficulty is that the buyer really desires to acquire only the hydrological services associated with the land and not the full suite of goods and services the land provides. Therefore, it may be more efficient to 'rent' just these services through a payment system.

Many M&I water suppliers have long relied on protected watersheds for their water supplies, in some cases owning the watershed or in other cases partnering with public or non-profit agencies to ensure there protection (Ernst 2004; Dudley and Stolton 2003). Protection of source watersheds by M&I water suppliers is therefore not a new concept or a new application of the Coasian Bargain. However, the concept of an M&I water supplier entering into agreements to 'rent' hydrological services from numerous private landowners in the source watershed is new. That said, PWS, and therefore by implication PES, are merely the latest application of an economic idea conceived some 50 years ago.

This definition and characterization of PWS suggests the following conclusions:

- Economists have a well-ordered set of tools that can be used in environmental management and, therefore, any characterization of PWS should take advantage of this typology rather than seek to replace it
- Type 1 PWS, where the buyer of services is the beneficiary is not a novel tool, being generally an example of what economists would call a Coasian Bargain
- Type 2 PWS, where the buyer of services is not the beneficiary but society (or a portion thereof) more generally is simply a case of a market-based instrument: a subsidy for the generation of a positive externality
- Water quality trading schemes are therefore not PWS, narrowly defined, but simply examples of cap and trade schemes.

In sum what is new in PWS is the application of the existing set of regulatory and non-regulatory tools for achieving incentive compatibility with regards to the specific problem of hydrological externalities of land use. PWS are also therefore a subset of ‘Markets and Payments for Hydrological Services,’ which would be a more robust descriptor of these regulatory and non-regulatory tools (absent command and control).

5.2 Summary of Relevant PWS Experiences

Below a summary is presented of experience and cases where PWS have been employed to remedy quality problems related to water for household/domestic use, particularly with respect to water quality impairment from agricultural landscapes. Examples of relevance can be sorted into three groups:

- Cases where the downstream domestic water quality beneficiaries have paid upstream land managers in agricultural landscapes for water quality improvements (i.e. Type 1 PWS)
- Cases where the downstream domestic water quality beneficiaries have paid upstream land managers (in any ecosystem) for water quality improvements (i.e. Type 1 PWS)
- Cases where system of payments have gone to reward upstream land managers in agricultural landscapes for water quality improvements (Type 2 PWS)

Table 14 presents a number of PWS experiences that meet these criteria as seleceted from a number of review documents that have sought to present PWS case studies (Landell-Mills and Porras 2002; Perrot-Maître and Davis 2001; Smith, de Groot, and Bergkamp 2006). Additional cases and key data is provided from other papers and sources as necessary.

The focus here is on nonpoint sources. Point sources can be included in payment schemes in agricultural landscapes as pointed out in the New York case reviewed below. In such a case payments are provided for both BMPs and point source control. Integrated programs are likely to be needed where agriculture or livestock processing, or other M&I uses are employing the waterway as a pollution sink. However, PES and PWS really originated as a means of funding forest conservation and reforestation (Johnson, White, and Perrot-Maitre 2001; Powell, White, and Landell-Mills 2002). For this reason the emphasis in the literature is on ecosystem solutions, not point source control. As a result the literature on PWS does not contain case studies of payments for point source control – even thought the New York case, which is easily the most cited, is in fact an example of an integrated program.

An interesting parallel between PWS for forest conservation and payments for point source controls exists. Most countries have legislation that regulates point source controls. However, these are often not enforced. Similarly, most countries have fairly proscriptive laws on the books that limit land use change and productive uses of land, particularly in forested areas. Oftentimes these laws are also not enforced. Thus, payments schemes for forest conservation and point source control find themselves in the awkward position of offering incentives for people not to undertake actions that are, in fact, against the law. This raise the problem of moral hazard. Why would any landowner or municipality obey the law when not doing so might lead to an offer of a payment for doing so? In both cases there may be an argument that the imposition of new regulations and laws has forced people to give up a practice they previously enjoyed with impunity. However, it is important to note the potential for abuse of positive incentive systems like payments and the need to clearly define a baseline for what behaviour is allowed or accepted before entering into a payment scheme.

Leaving further discussion of point source payment schemes to the New York case, the summary below examines PWS with respect to non-point sources. To date, little to no external evaluation work of PWS schemes has been carried out. Thus, much of the available information comes in the form of review or policy documents that provide varying levels of detail. In some cases these documents are clearly intended to promote PWS (or PES) and may not necessarily be the most reliable source of information (Ernst 2004; Dudley and Stolton 2003; Johnson, White, and Perrot-Maitre 2001; Scherr et al. 2006). In others the intent is to study the development of payment schemes in order to learn from experiences and provide guidance to practitioners (Smith, de Groot, and Bergkamp 2006; Rojas and Aylward 2003; Landell-Mills and Porras 2002). Often one such review (such as in the present case) merely cites another review. This is unfortunately the case with the table presented below. This of course makes triangulation a dubious proposition as a means for validating information. A small but growing resource are working papers, books and peer-reviewed articles that have explicitly set out to evaluate one or more PWS, typically using a few case studies that are examined in more detail than they receive in the the broader reviews (Salzman 2005; Pagiola 2002; Postel and Thompson 2005; Kosoy et al. 2007; Miranda, Porras, and Moreno 2003; Pagiola, Arcenas, and Platais 2005; Déprés, Grolleau, and Mzoughi Undated). Finally, the work by the International Institute for Environment and Development reflected in Landell-Mills and Porras (2002) continues with a forthcoming review of cases of payments for watershed services in developing countries. A prototype website for the IIED project (www.watershedmarkets.org) with case study details was also consulted for information on cases, particularly in Africa.

Still, there are few PWS cases that have been in existence for long enough to have been subjected to the scrutiny of a formal evaluation. This is true in the case of Type 1 PWS, particularly those where domestic water suppliers are funding BMPs in agricultural landscapes. The exception is the New York City partnership with farmers in the Catskills watershed. As this case is the oldest, most-cited and most-discussed case of this nature it is discussed below in detail as it yields many important lessons about Type 1 PWS.

Other major Type 1 PWS recognized in the literature include the Vittel-Nestle program in France and the program set up in the city of Quito, Ecuador. The Vittel case involves bottled water rather than M&I supply but is an excellent example of the use of both land purchase and payments for services. In this case, the multinational relied on natural springs as an important source of natural mineral water, labelled under the Vittel name. The Vittel springs are located in an agricultural watershed and from the 1970s on showed an increasing increase in contaminants, in particular rising nitrate concentrations. As mineral waters are marketed at least on their promise of being low in nitrates this led the company to take action. Beginning in XXX Vittel began purchasing lands in the watershed. Ultimately Vittel acquired 45% of the lands at a cost twice the market value (Déprés, Grolleau, and Mzoughi Undated). Due to restrictions on the use of agricultural lands, these lands could not be retired and were subsequently leased at no cost back to farmers willing to use BMPs required by Vittel.

For the remaining lands Vittel pursued contracts with each farm leading to the implementation of recommended BMPS. Farmers were compensated based on the expected costs of adjusting to these new practices. However, given uncertainty on these costs, farmers probably received a share of the gains that Vittel would otherwise have reaped from the contracts (Déprés, Grolleau, and Mzoughi Undated). These contracts are from 18 to 30 years in length given the lag time required to see results in terms of spring discharge. Such a solution was feasible given that the farmers were easy to identify and not too numerous; initially there were around 40 farmers, although the number has since shrunk to 26 (Perrot-Maitre 2006). It was also possible to clearly define the rights that were exchanged; in this case the contract specifies the BMPs that each farmer shall undertake. The time required from the initiation of work by the research team that pulled together the BMPs and water quality information to the signing of the first contract in 1992 was a period of 4 to 5 years (Déprés, Grolleau, and Mzoughi Undated). By 2005 all but three farmers had signed contracts and 96% of the targeted area was under contract, although subsequent reports suggest that all 26 farmers are now enrolled (Déprés, Grolleau, and Mzoughi Undated; Perrot-Maitre 2006).

The Vittel case is of interest as it shows how land purchase and payments can successfully be integrated in a plan for improving water quality in a watershed dominated by agriculture. Reforestation and control of point source pollution were also included but were of minor import (Perrot-Maître and Davis 2001). The case is also useful for showing how a cooperative, learning-by doing approach can lead to successful negotiations (Perrot-Maitre 2006). However, it is worth noting that an important factor driving success in this case was the overwhelmingly favourable economics of the arrangement. Vittel stood to lose the annual production of 1 billion bottles of water from the Vittel springs (Perrot-Maitre 2006). None of the studies found quantified this loss in dollar terms but the revenue alone to the parent company Nestle Waters must exceed $1 billion per year. Stacked up against the best estimate of what the deal cost Vittel – about $24 million in land purchase, equipment provision and payments – there can be little doubt that a successful deal was likely. The only question was how much of the benefits of the agreement the landowners would succeed in extracting from Vittel.

In the case of the Quito example, funds paid by the municipality and other sources have been deposited into a trust fund (FONAG) since 2000 (Echavarria 2002). The intent was to create a non-declining endowment fund that can receive funds from different sources and use the investment returns to fund watershed protection activities. However, a recent decision by the Quito municipality to guarantee the payments by water users into the future are leading FONAG - the non-profit in charge of the program – to change its strategy and to begin funding projects right away (Arcila 2007). The program has only recently begun funding projects so to date it has very little experience with contracts and payments for watershed services. Again the time lag between the idea for a payment scheme and the onset of implementation is considerable, around 5 years in this case.

The Nicaraguan and Honduran examples cited in Table 14 are notable for their small, community-level scale. These examples involve communities of just a few thousand people acting together to protect critical areas in watersheds that range in the hundreds to the thousands of hectares (Kosoy et al. 2007). For example in the Jesus de Otoro watershed a town of 5,200 is aiming to protect just 200 critical hectares. In doing so they are levying a small charge of $0.06 per household per month. With 1,269 households this is approximately $100/year in funding. With incentives for forest conservation and agricultural BMPs being paid at from $3 to $17/ha these funds were sufficient so far to implement payments on 74 of the hectares targeted by the project. Interestingly, in both of these Central American cases the water boards are local, citizen run groups, which ended up arriving at charges that were only a fraction of their willingness-to-pay as determined by consultant studies. Perhaps this is a benefit of working at a small, local scale. Such a program can be initiated at a modest level and then increased, as it is perceived to generate benefits. This conclusion is not far off that of a World Water Council document on financing the MDGs that suggests that WS&S project involve local communities so as to keep costs down of meeting rural needs (Toubkiss 2006).

Other PWS are listed in Table 14, including a number of Type 1 schemes that are underway as mechanisms more for forest conservation and management than for payments to farmers in agricultural landscapes. These provide additional examples of how payment schemes involving M&I water suppliers have emerged. In the ESPH Heredia case in Costa Rica, a mandatory surcharge is added to water bills and then used to pay landowners to conserve forest areas (Guzmán Gómez Undated). In Coatepec, Mexico a voluntary surcharge was instituted (for the same purpose) and then replaced with a mandatory surcharge (Guzmán Gómez Undated). It is worth noting that the PWS schemes listed in Table 14 just reflect a number of more well-known cases not the full universe of cases. For example, Nestle Waters has worked to replicate the Vittel experience in other watersheds in which it has an interest and other European cities and companies have pursued PWS schemes (Déprés, Grolleau, and Mzoughi Undated; Perrot-Maitre 2006). The IIED website on PWS records over 100 cases in developing countries, all at some stage of development or implementation. Thus, there is much experimentation underway in this field.

With respect to the Type 2 PWS schemes listed in Table 14 these represent a number of the large country-wide payment or incentive programs for watershed management such as those in the US, UK, Costa Rica and Mexico. As these are reviewed elsewhere and are effectively agricultural, reforestation or forest conservation subsidy programs unrelated specifically to the issue of domestic water quality they are not discussed further here.

Table 14. Payment for Watershed Service Cases

Name of Program (Municipality)	Country	Buyer Type	Seller Type	Contracted Activity	Desired Function	Desired Service	Intermediary None	Amount Paid By Buyer	To Manager
Type 1 Deals Between M&I Buyers and Providers (in Agricultural Landscapes)									
New York City	USA	M&I Water Supplier (M&I)	Farmers	Land Purchase, BMPs, Riparian & Forest Management	Reduce Phosphorus & Eutrophication	Avoid Filtration & Public Health	Catskills Valley Corp.		
Vittel-Nestle	France	Bottled water	Farmers	BMPs, Land Purchase	Maintain Low Nitrates	Avoid loss of Brand	Agrivair	$1.50/m^3$	$230/ha/yr
Quito	Ecuador	M&I, brewery, grants	Protected Areas & Landowners	Reforestation, Forest Protection, BMPs	Maintain Water Quality	-	FONAG		
[a]Jesus de Otero	Honduras	M&I	Landowners & Farmers	Forest Protection, BMPs	Improve Water Quality	n/a	None	$0.06/hh/ month	
[a]San Pedro del Norte	Nicaragua	M&I	Landowners & Farmers	Forest Protection, SFM, BMPs	Improve Water Quality and Flow			$0.31/hh/ month	$26/ha/yr
Other Type 1 Deals Between M&I Buyers and Sellers (in any Ecosystem)									
[b]ESPH	Costa Rica	M&I and brewery	Landowners & Park	Forest Conservation, Reforestation	Maintain Water Quality, Regulate Flows	Avoid Filtration & Public Health	PROCUENCAS, FONAFIFO	$0.008/m^3$	$30 to $110/ha/yr
[c]Coatepec	Mexico	M&I	Landowners	Forest Protection	n/a	n/a	FIDE-COAGUA	$0.10/hh/ month	$50 to $100/ha
[d]Cauca Valley	Columbia	Voluntary fees from Farmers	Landowners	Land Purchase, SFM	Improve Base and Peak Flow & Flood Control	Water Supply and Flood Control	Cauca Valley Corporation	$0.0005 to $0.002/m^3$	
Type 2 Deals for Water Quality in Agricultural Landscape									
Conservation Reserve Program	USA	General Fund Taxes	Farmers	Land Fallowing, BMPs	Improve Water Quality and Flow Regime, Reduce Erosion	n/a	NRCS and Soil & Water Districts	$3/ton of erosion reduction	$6 to $26/ha/yr rental plus other incentives
Nitrate Sensitive Areas Scheme	UK	Government: General Funds	Farmers	BMPs	Reduce Nitrate Levels	Reduce Treatment Costs	None		
[e]Mexico PWS	Mexico	M&I Water	Landowners	Forest Protection	Watershed	n/a	CONAFOR –		

Name of Program	Country	Buyer Type	Seller Type	Contracted	Desired Function	Desired Service	Intermediary	Amount Paid	
(Municipality)				Activity			None	By Buyer	To Manager
		User Fees			function		Forestry Agency		
[a]Costa Rica PES	Costa Rica	Tax on Fossil Fuels /Water Fees	Landowners	Reforestation, SFM, Forest Protection	Biodiversity Conservation & Ecosystem Function	Carbon, Water, Biodiversity and Scenery	FONAFIFO	$\$0.005/m^3$	\$40/ha/yr
Salmon Safe	USA	Food Consumer	Farmers	BMPs	Habitat for Salmon	Restore Fishery	Salmon-Safe		
Pirnampiro	Ecuador	Municipality	Landowners	Forest Protection & Land Management	Improve flow	n/a	n/a		

Notes: Ag BMP = Agricultural BMPs for water quality, SFM = sustainable Forest Management

Source: Information from Perrot-Maître and Davis (2001) and Smith et al. (2006), supplemented by [a] Kosoy et al (2007), [b] Gamez (2007), [c] Guzmán (Undated), [d] Landell-Mills and Porras (2002), and [e] Muñoz-Piña et al (In Press).

5.3 The New York City Watershed Management Program

Perhaps the most cited and talked about PWS case is also potentially the most relevant to this paper. It is also widely referenced and examined in the literature on PWS (Appleton 2002; Perrot-Maître and Davis 2001; Landell-Mills and Porras 2002; Dudley and Stolton 2003; Salzman 2005; Postel and Thompson 2005; National Research Council 2000; Sagoff 2002; Ernst 2004). As related by Sagoff (2002) this deal first received attention in a Nature commentary in which two economists from Columbia University wrote that 'In 1996, New York City invested between $1 billion and $1.5 billion in natural capital, in the expectation of producing cost savings of $6 billion - $8 billion over 10 years' (Chichilnisky and Heal 1998). The project in question involved investments in the Catskills watershed – an agricultural landscape from which New York City obtains its water supply. The economists went on to say that this demonstrates 'how New York City realized billions of dollars in economic benefits by sustaining the Catskills watershed as a water filtration system, rather than . . . building a new filtration plant.'

As Sagoff (2002) points out these figures have been widely repeated and taken as fact. Indeed, with the exception of Postel and Thompson (2005) the reviews and policy documents reviewed for this paper do not attempt to verify these figures. Sagoff (2002) remains the lone critical voice on the New York case, although his objective is really to question whether tying conservation of ecosystems to utilitarian values is a wise choice for conservationists to make. Fortunately, a number of sources exist in this case including a full report by a committee of the National Research Council, a first-hand account of the inception of the deal and a recent effort to collect additional information on progress of implementation of the case (NRC 2000; Appleton 2002; Postel and Thompson 2005). More recent programmatic reports by the New York City Department of Environmental Protecion (DEP) also provide a voluminous source of information, although no clear, comprehensive review and analysis of the environmental outcomes and corresponding program expenditures was found on the DEP website.

In the 1990s the City of New York relied on a water supply of 4.9 million m^3/day (National Research Council 2000). The City obtains 90% of its water supply from surface water in the Catskills and Delaware (Cat-Del) watersheds that extend some 200 kms north of the City (Appleton 2002). The remaining 10% comes from the Croton watershed (Appleton 2002). Water use by the city from the Cat-Del watershed would then be 4.4 million m^3/day and 0.49 million m^3/day from the Croton. Historically, New York's water was unfiltered but by the end of the 1980s the City had decided it needed to build a filtration plant for the Croton. Appleton (2002) reported that the facility was scheduled to be completed in 2009 at a cost of $500 million with annual running costs of $50 million. The O&M for this plant would then be $0.28/$m^3$ and the total annual cost (spreading capital costs over 40 years at 3%) would be $0.40/$m^3$. This is within the range identified previously for water treatment costs in this paper.

The potential cost of having to build a similar plant for the Cat-Del watershed led the City to explore a watershed management approach to the problem and to work with farmers and other landowners in the watershed to reduce non-point source pollution (see Appleton 2002 for details). In January of 1997 the City signed a Memorandum of Agreement (MOA) with 76 organizations including the State, EPA, local governments and environmental groups (Ernst 2004). The MOA committed the City to spending $1.5 billion over ten years (Postel and Thompson 2005). The MOA included a voluntary agreement between the farmers and the City for the adoption of a package of BMPs called the 'Whole Farm' program. In addition, the City undertook to acquire additional lands for protection purposes, implement a forestry management program based on the same ideas, engage in stream corridor management and create the Catskills Development Council to improve urban land management and limit sprawl and development in the watershed. The MOA also set forth plans for investments in sanitation infrastructure in the Cat-Del watershed to reduce pollutants entering rivers.

In the Cat-Del watershed the city taps a land area of 414,000 hectares, of which 68% is forested, 16% is low density residential, 9% is vacant land and 5% is agricultural land (National Research Council 2000). Agricultural use tends to be confined to the valleys in the Delaware watershed and dairy farms predominate. Existing forest is of high quality and has its hydrologic protection function intact (National Research Council 2000). Prior to 1997 land ownership was largely in private hands at 76% of land area. Of the 24% of land held by the public the City owned 14,600 hectares (3.5% of the land area). Between 50,000 and 200,000 people inhabit the watershed, depending on the time of year.

The National Research Council (NRC) provides a thorough review and accounting of the background and early implementation of the MOA and the City's Watershed Management Program (NRC 2000). The NRC report provided the following findings:

- land use and population pressure in the watershed are limited (a net increase of 235 persons residing in the watershed was observed between 1860 and 1990), such that the future threats to water quality from development could be offset by careful planning, more extensive environmental regulation and wastewater management
- water quality in the Catskills did not show signs of deterioration in the decades leading up to the agreement, other than the increasing risk from microbial pathogens, particularly *Cryptosporidium* and *Giardia* (which demonstrate resistance to chlorine disinfection)
- under the new Surface Water Treatment Rule (SWTR) issued by EPA in 1989 the City would have been required to install filtration to deal with these newly recognized microbial threats but it obtained a Filtration Avoidance Determination (FAD) waiver from EPA. One element in the granting of the waiver was the plan for the Watershed Management Program represented by the MOA.

As pointed out by Sagoff (1992) the NRC report suggests that the threat to New York City ratepayers in the 1990s was not so much water pollution from rampant development and agricultural source pollution but the promulgation of the new SWTR regulation and the re-emergence of wildlife in the watershed that serve as a source of microbial pathogens (waterfowl in particular were cited in the NRC report). The growing risk of microbial contamination is precisely why the SWTR was initiated and why the NRC report suggests that the City program refocus on control of microbial pathogens as its primary objective (rather than phosphorus and eutrophication). Sagoff (1992) goes on to critique the emergence of a new conventional wisdom that somehow NYC is spending $1.5 billion on protecting natural ecosystems so that they may filter and purify the water supply for the City. He points to limited expenditure by the City of a sum on purchasing land and easements on by early 2002 as evidence that the program is not quite what the Columbia economists thought. Since that time, however, the land acquisition program has met its target of making offers on 143,000 acres of land and has come relatively close to spending its allocated budget, as described further below.

Various reviews and reports of the program carried out by the City in 2006 are supplemented by information from the City website to provide an overview of progress and expenditure on the Watershed Management Program. Beginning with non-point source management:

- Land Purchase: 31,000 hectares acquired via fee simple purchase or easement at a cost of $194.8 million (NYC DEP 2007)
- Whole Farm Program: 94% of farmers signed up, 4,268 BMPS on 295 farms financed to the total of $28.2 million (NYC DEP 2007) and total expenditures of $40 million (NYC DEP 2007)
- Stream Management Program: Approximately $3 was originally allocated to this program but the Department says commitments by the City are now $31 million (NYC DEP 2007), to date all

planning is complete and with 4 of 12 stream restoration projects complete $2.6 million has been spent on project implementation (NYC DEP 2006)

- Watershed Forestry Program: Over 290 management plans completed by April 2003 for 18,200 hectares of privately owned forests, with a commitment by the City of $4 million and matching federal and state grants of another $4 million (NYC DEP 2007).

Thus it appears that as of the end of 2006, expenditure on non-point source management totalled at least $245 and a total of $274 million is committed. Of funding to date only about $80 million (30%) or so is committed as payments to farmers for watershed protection, as opposed to land purchase. The other half of the Program is effectively sanitation infrastructure investment program designed to limit point source pollution in the watershed. Program status to date include:

- City-owned Wastewater Treatment Plants (WTPs): the City completed upgrades of the six WTPs it owns and operates in 1999 at a cost of $240 million (NYC DEP 2007)
- non-City owned WTPs: by the end of 2005, 25 of 37 City partners had completed WTP upgrades and the City amended the MOA to increase funding by $192 million for a total of $272 million for the regulatory upgrades (NYC DEP 2007)
- Septic: system replacement and maintenance expenditures of $13.6 million were committed under the MOA and a further $15 million authorized under the 2002 amendment of the MOA in order to obtain the 2002 FAD (NYC DEP 2006)
- Stormwater: initial funding for stormwater retrofits was $7.5 million and another $7.5 million was committed in 2002, with approximately $10 million expended or approved by the end of 2005 (NYC DEP 2006, 2007); expenditure on stormwater control projects was $1.9 million of $31.7 million initially allocated with $7.9 million transferred over to other programs (NYC DEP 2007); and local technical assistance fund of $1.25 million was unspent as of end of 2005 (NYC DEP 2007)
- New Sewage Treatment Infrastructure Program: Seven new WTPs and upgrades to facilities linked to municipal plans are either underway or complete and are financed by the city through a $75 million allocation under the MOA and a $12 million expansion of the program as part of the 2002 amendment of the MOA (NYC DEP 2006)
- Community Wastewater Program: As part of MOA amendments in 2002 and 2006, six communities requiring intermediate wastewater treatment facilities were identified and will be financed by up to $16 million in City funds through 2012 (NYC DEP 2007, 2006)
- Sewer Extension Program: 14 communities with failing septic systems will be connected to City-owned WTPs at a cost of $10 million; and all but 1 have signed agreements with the City (NYC DEP 2006).
- Sand and Salt Storage Facilities: Sand, salt and de-icing facilities for winter road maintenance will have been brought up to regulations through the expenditure of $10 million by 2004 (NYC DEP 2007).

The total expenditure on infrastructure is thus approximately $700 million for a rough total of expenditure under the Watershed Management Program (either completed or planned) of roughly $975 million. This figure does not include the $60 million that apparently has been granted to a trust fund for the watershed for environmentally sustainable economic development projects (Postel and Thompson 2005).

It is somewhat surprising that no complete accounting of expenditure on this program was available from the reports published on the New York City, EPA or the State of New York websites. Postel and Thompson (2005) cite the DEP commissioner as stating in 2004 that more than $1 billion had been spent.

In all likelihood the figures listed above exclude the City's management and administrative costs for the program so these figures may be considered roughly consistent. It should also be noted that on the watershed management side of the equation state and federal cost-share contributions in the tens of millions of dollars are noted in the sources cited above. In the sources located for this review it was not possible to find any evidence of expenditures, planned or actual, in the vicinity of $1.5 billion, this despite a statement by DEP in its 2006 FAD renewal application that this amount has been spent (NYC DEP 2006).

Clearly – and this can be observed directly from budget figures available – the majority of the Watershed Management Program funds go to infrastructure (Salzman 2005). The numbers cited above suggest that so far only about 25% of program expenditure has gone to ecosystem management as opposed to sanitation infrastructure. And of this amount only a small portion has gone to BMPs on agricultural land. Sagoff (2002) argues that the original comments by Chichilnisky and Heal (1998) have led many to believe that New York City was spending the $1-$1.5 billion on purchasing and preserving land in the Catskills order to 'purify' the City's water supply. As correctly noted by Sagoff, a watershed does not serve to purify the water supply. The input of rainfall is typically quite pure. Rather it is the interaction of rainfall with land, vegetation, soil, animals and humans that lowers the water quality of the precipitation input. As the NRC report notes, significant contaminants are sourced from forest, agricultural and residential lands in the Cat-Del watershed. Thus, the objective of improved watershed management is not to 'purify' the water but instead to limit the leaching and contamination that might be caused by alternative, less-environmental forms of land use and management.

What is most instructive about the Catskills case is that in a mosaic landscape that includes agricultural land, forests and small communities, approximately 75% of the expenditure has gone to minimize and reduce the impacts from point sources associated with communities and their residential, commercial and industrial activities. The remaining 25% has gone to non-point source watershed management activities. Of particular relevance for this paper is that of the funds dedicated to these activities some 70% have gone to land purchase and just 30% to BMPs on farms and forests and in stream corridors. This information could mean one of two things: that these BMPS are relatively inexpensive or that they are of relatively little consequence. Another way to look at it is that these lands are just 5% of the area in the watershed. Taking the City's estimates of $40 million for the Whole Farm Program this would be about $2,200 in expenditure per hectare. By comparison the land acquisition program has protected land at a cost of $6,000 per hectare. It may therefore be more efficient to purchase only the critical lands and simply 'rent' hydrological services from the remaining lands. It appears likely that farming is just one of a number of potential threats to water quality in the watershed, and one of limited extent and therefore a minor component of the overall program .

In actuality it is the sum total investment in the Watershed Management Program by the City that has allowed it to garner a number of successive five year FADs from EPA and, recently, the first ever 10-year FAD (granted by EPA in 2007). Whether any one component of the Program was critical to obtaining the waiver and thereby postponing the date a filtration plant would be required is not known. Nor is it known to what extent each element of the investment program contributed to the reduction of pollution or the maintenance of the waiver. However, it is important to recognize that watershed management in this program consists largely of investment in sanitation infrastructure to ameliorate the impacts of human sewage and is not an exclusive investment in protecting ecosystem services or natural capital.

Another question emerging from this case study is the issue of thresholds in BMPs from agricultural lands. In order to accept the agreement the City required a commitment from farmers that 85% of farms would be enrolled within 5 years (Appleton 2002). Presumably, the rationale was that without the ability to scale the program to this size the City might be left investing large amounts of money but falling short of its clean water objectives and, therefore, having to install a filtration plant. Although the NRC (2000) states that agricultural water pollution is a ‘potentially significant’ contributor of microbial pathogens, nutrients and pesticides the exact nature of, and argument for, a threshold in regard to farmer participation is not clear.

The widespread claim that the Watershed Management Program (correctly understood) avoids a $6 to $8 billion expenditure can be looked at in different ways. NRC (2000) reports that there were estimates that a plant for the Cat-Del would have cost just half of this number. It is also the case that this estimate reflects the replacement cost approach rather than the avertive expenditures approach as reviewed in Section 3.2.1. As such caution in using this number as a benefit estimate is required. On the other hand, a recent newsletter by an environmental advocacy group for management of the Croton watershed now reports that the Croton treatment plant may cost up to $2.9 billion and that the contractor originally selected to build the project has abandoned the project (Croton Watershed Clean Water Coalition 2007). This raises the question of whether the $6 to $8 billion dollar price tag for the Catskills treatment plant would be an under-estimate given that it would need to process nine times the volume of water as the Croton plant.

With regard to the Croton plant, it is interesting to note that recent data from DEP suggests that City water conservation efforts continue to make headway. Since 2002 water use has remained below 4.3 million m^3/day for the entire city (NYC DEP 2007). As noted earlier, in the 1990s the Cat-Del watershed was producing 4.4 million m^3/day. Whether someday the City’s reliance on the Croton watershed might become obsolete is not a question that this paper can answer, however, this development does suggest that the City’s water conservation program would be generating exceedingly high monetary benefits if it could lead to avoiding the costs of the Croton plant.

It is also true that technology evolves such that yesterday’s cost for treatment may not be today’s cost. The City’s overarching strategy is one of pursing a dual track approach (NYC DEP 2007). To confront the risk of ever-stricter regulation and the possibility that the investment program would not be sufficient to maintain the FAD, the City has also undertaken to be in a state of readiness to construct a filtration plant should it be needed. In economic terms the decision the City faces is to assess the costs, benefits and risks involved. In other words the investment program remains viable as long as the marginal costs of continuing the investment program are less than the marginal costs of the filtration plan. Or in other terms as long as the benefits of postponing investment in filtration exceed the ongoing costs of the Watershed Management Program the program will continue to be viable. However, risk is an essential element here. A large disease outbreak from *Cryptosporidium* or *Giardia* in the City – such as that occurring in Milwaukee in 1993 that sickened over 400,000 residents – could greatly affect the risk-return calculation.

The city continues to develop plans for future water treatment, although plans have now shifted from a traditional coagulation/filtration plan to a UV-disinfection facility (NYC DEP 2007). A pilot facility was almost complete as of the end of 2006 and design, contracting and permitting work for a full facility were either complete or underway as of that time as well. The 2006 submission for FAD renewal to EPA suggests that all Cat-Del water will be treated by this facility by 2012 (NYC DEP 2006). No cost estimates for this plant were found in the documents from DEP.

In sum, it is likely that investing in the Watershed Management Program did avoid a larger investment in filtration, in the sense that it bought the City time to reduce water quality threats in the Cat-Del watershed. However, if the City had the option of spending $1 to $1.5 billion to postpone filtration and provide quality water to its citizens or spending $6 to $8 billion to provide quality water it is not the case that the savings or the economic benefits of the Watershed Management Program were the difference between these two amounts. The real savings will be the difference in present value between undertaking the Program for 15 years and then installing a UV-plant (apparently) and building a conventional plant in year 0. The answer to this question will depend on how accurate was the $6 billion estimate, how much less expensive is the UV technology and how much have the costs of construction risen relative to the general rate of inflation. The latter is a particular concern going forward as noted by DEP in its latest report (NYC DEP 2007)

For the purposes of this paper, however, the interesting lessons learned from the New York experience come from the integrated nature of the program – combining payments to point sources with land purchase and payments to farmers for BMPS. Also, the cooperative process of designing the Whole Farm program and the implementation of the pollution reduction program through a system of payments to farmers are instructive. The NRC report applauds the effort by the City to reach this agreement with farmers and agrees that this is a 'reasonable strategy' for the City and 'possibly' other suppliers. In this case the threats may have been limited and not of primary importance but nature of the negotiation is instructive. In particular, the threshold level the City insisted on warrants further investigation in considering the scale of a useful PWS program in an agricultural landscape. The additional lesson learned from this case is that human sewage and sanitation and the need for point source pollution prevention infrastructure will need to be considered as part of any watershed management plan.

6. Payments for Water Quality Services from Agricultural Landscapes in sub-Saharan Africa

The paper reviews and summarizes the state of knowledge with regard to agricultural landscapes, water quality and domestic water supply and the management of the relationship between these constituents through the use of Payments for Watershed Services schemes. The review is informs a scoping-level assessment of the potential contribution that PWS could make in the context of sub-Saharan Africa (hereinafter referred to simply as "Africa"). In this section the sources, information, key elements and lessons learned in the prior Sections are drawn upon in responding to four scoping questions:

1. Identify current strategies to improve quality of water for household/domestic use and estimate spending to meet water quality objectives, including both national public sector spending as well as overseas development assistance.

2. Identify the potential role in improving quality of water for household/domestic use for improved ecosystem management in agricultural landscapes and assess the potential of environmental services to reduce current costs society is bearing regarding the provision of clean water for household/domestic use, including assessment of potential willingness to pay for these services and the costs of alternatives

3. Identify the potential for developing innovative institutional arrangements and financial mechanisms for developing payment programs for water services from agriculture landscapes (WSAL).

4. Assess potential legal and regulatory barriers to the establishment of payment programs for WSAL and identify likely solutions.

In addition a short conclusion on the state of the knowledge and prospects for PWS in Africa is provided.

6.1 Current Strategies to Improve Domestic Water Quality

The discussion above and the results from earlier sections suggest that current strategies in Africa (and in many other regions) will likely rely on engineered water treatment approaches going forward. Notwithstanding, the enthusiasm of US EPA and the US National Research Council for the New York City example of Type 1 PWS, it is questionable how much attraction there is in Africa for a watershed approach to water quality improvement. There is discussion in this regard with respect to intact, forested watersheds, but this idea is probably less well appreciated in the case of agricultural landscapes. This would be true generally, given the emphasis on the use of these payments in the conservation of forest ecosystems. Nonetheless, if improving the input water quality to regulated water systems results in cost savings in the U.S. it will likely result in cost savings in Africa as well.

It is important to note that costs for water treatment are unlikely to be lower in Africa than in developed countries. Although a unit of pollution is likely to incur lower social cost in terms of human health and productivity impacts (due to lower personal productivity) in developing countries, the corresponding cost savings of avoiding treatment costs will likely not vary. In Africa the investments in water treatment facilities has in many cases not yet been made. Indeed they may not be occurring because their costs are high and their benefits are low – when compared to the situation in developed countries. African countries are implicitly choosing to avoid water treatment costs and instead bear the costs to human health. In developed countries the decision has been to pay the (relatively small per person) costs of installing water treatment infrastructure – given the high costs of the human health impacts (due to higher personal productivity). Perhaps in developing countries with less extensive agricultural production and lower labour costs, the alternative of avoiding the water quality problem through improved ecosystem management in agricultural landscapes would be a more appropriate solution, or at least one worthy of further investigation.

Despite this interest and activity it is clear that the traditional approach to water supply of dealing with pollution at the point of diversion or extraction prevails at present. In Africa with low existing numbers served by water supply infrastructure, it is probable that much new infrastructure will be required simply to provide water supply when and where it is needed. Along with this supply infrastructure is likely to come water treatment facilities intended to protect water supplies from contaminants and pollutants in source waters. As described earlier in this paper based on estimates derived for the World Health Organization, large amounts of money are destined to be spend this effort in Africa, as agencies work to achieve or surpass the Millennium Development Goals for water and sanitation. Depending on the level of service provided the annual expenditures range from 500 million a year to meet the MDGs for water supply to $4.3 billion to meet basic access to water supply and sanitation for all (Hutton and Haller 2004). Full provision of in-house regulated water supply would cost $24.7 billion per year. Put in context the WHO estimates are on the low end of the range of six studies that have attempted to calculate the global costs of achieving the MDGs (Toubkiss 2006). A number of other studies suggested costs three times that of the WHO study. No matter the exact figure, the expectation is that in Africa much of this expenditure will rely on external financing (Toubkiss 2006).

In light of the earlier analysis of the New York City case it is also of interest to note that global studies suggest that meeting the MDG sanitation goals will be 2 to 5 times more expensive than meeting the water supply goals (Toubkiss 2006). This is a reminder that the potential applicability of the ecosystem management portion of working towards water quality will be limited – the major elements in terms of expenditure will be provision of sanitation infrastructure followed by the provision of water supply infrastructure. Efforts to provide high quality water inputs to supply systems will be the third component. As noted earlier in this paper the WHO estimates suggested that for the most expensive option of in-house regulated water supply the water treatment costs were roughly half of total costs. And as explored, it can be expected that better ecosystem management will serve to only offset a portion of those costs, possibly just the variable O&M costs.

It should be noted that these conclusions come from studies examining large, capital-intensive supply and treatment infrastructure designed to serve large cities. The provision of water supply and sanitation to smaller, rural communities may not exhibit the same technological and economic characteristics. Therefore it is worth noting that the extent of urban and rural needs in Africa differ. In Africa just over 50% of urban populations have access to sanitation; while in rural areas the number falls to 28%. Similarly, with drinking water some 270 million rural people do not have access to improved drinking sources while just over 50 million urban inhabitants lack this access. Thus, although PWS Type 1 are often conceived as large transactions between municipalities and large-scale watersheds it will also be important to consider the potential of PWS at the community and small watershed scale.

6.2 Potential for Ecosystem Management to Improve Domestic Water Quality

The next three sections explore three different questions related to the potential of PWS in Africa:

- does ecosystem management in agricultural landscapes have the potential to lead to significant and quantifiable improvements in water quality, thereby lowering water treatment costs and promoting human health?
- are payment systems a potential tool for incentive compatibility in undertaking these improvements?
- What legal, regulatory and contractual conditions are likely to be required to foster PWS?

With regard to the potential of ecosystem management the question is in what context and to what extent and scale does an ecosystem approach have potential to make a difference for domestic water supply. In the discussion below, material provided in earlier sections is used to explore these issues in the African context.

Physical, chemical and biological pollutants affect water quality. African agricultural landscapes are likely to be susceptible to all three problems. Land degradation due to forest clearing and poor soil husbandry will increase erosion and downstream levels of suspended sediment and sedimentation. Withdrawal of waters for irrigation and other uses will decrease water quantity in rivers and streams, leading to higher temperatures and, other thing equal, higher concentrations of other contaminants. Runoff of chemicals used on lands as fertilizer or pesticides is also a potential risk. For Africa this problem may be less severe in areas that are sufficiently impoverished that agriculturalists have little access or ability to purchase such chemicals. It is also probable though that the emerging concern in developed countries regarding residues of veterinary medicines affecting water quality will be less of a concern in Africa. On the other hand, in places where these chemicals are subsidized and/or where training in their use is not in place there may be excessive rates of runoff of these chemicals.

Biological pathogens are likely a significant concern in the African context. The use of animal manure in place of inorganic fertilizers, the lack of human sanitation for small farming communities spread across the landscape, the widespread maintenance of livestock as part of the household livelihoods, and the probability than any confined animal feeding operations have unregulated discharges all serve to accentuate this risk in a developing, as opposed to developed, country context.

In terms of impacts on human health, the prevalence of suspended sediments and biological pathogens are probably of primary concern in the African context. Water carrying large amounts of sediment provides a suitable environment for microbial activity and survival. As emphasized earlier, 40% of global morbidity and mortality from diarrhoea occurs in sub-Saharan Africa. The social and economic costs of such waterborne diseases are extremely high in Africa. A WHO study estimates that providing access to basic water supply and sanitation means that the population would have just a high (instead of very high) risk from environmental faecal-oral pathogen load. Even this apparently marginal improvement would yield economic benefits of $44 billion/year (Hutton and Haller 2004). The same study suggest that taking this risk down to a medium or low level through the provision of full piped water supply to the household (and partial sewerage) would generate over $100 billion/year. While these estimates may be on the high side (as discussed in Section 3.3.2) they serve to convey the enormity of the problem. Still, source water quality is only one cause of these diseases – as sanitation and personal hygiene play an important role. It is not possible, therefore, to attribute a specific benefit (or willingness-to-pay) to improving poor source water quality in Africa. Nor are there valuation studies of a case study nature that can be used to understand these benefits in specific locales.

The other impact of poor source water quality comes directly from impacts on the storage, withdrawal and treatment of water for domestic use. Sediment can negatively affect production of treated water through physical impacts on reservoirs, settling ponds and outtakes; as well as through raising the costs of treating water to remove sediment and biological and chemical materials affixed to sediments. Africa as a whole has a far smaller proportion of its rivers dammed for storage, thus some of the concerns manifested in other regions about the impacts of sedimentation on dams will be of less consequence, or at least prevalence.

Chemicals and biological pathogens also impose higher treatment costs, either in the form of enhanced treatment facilities or variable treatment inputs. Of the different regions, Africa, is probably the one most likely to face difficulties in responding to these situations. Reliance on external finance, internal red tape and human resource limitations may mean that changing needs for treatment facilities are not accommodated in an efficient fashion. Similarly, while shortage of variable inputs – particularly chemicals used in treatment – would not be a concern in the developed world they may well be in parts of Africa. In such cases, the costs of poor water quality may indeed feed through in terms of increased disease incidence rather than treatment costs. This, as facilities either must limit their production of treated water, forcing consumers to lower quality supplies, or simply push lower quality water through the system to the end users.

These impacts on treatment costs apply not only to large M&I systems but also to community systems. To the extent that community systems are under-capitalized, changes in water quality may have a proportionately greater impact on them as they may not have the buffering provided by large dams, modern treatment systems, large amounts of funding or significant socio-political clout.

The literature review provided earlier suggests the following findings with regard to the relationship between BMPs, water quality and water treatment:

- Water treatment costs will be a significant component of overall water supply and sanitation costs
- In agricultural landscapes, improving water quality will involve a mix a mix of point source controls (including sanitation) and BMPs
- Where water treatment infrastructure is in place the savings of improved water quality will only be the reduction in the variable costs of treatment, and these will be relatively small
- Where water treatment infrastructure is not yet in place the savings from improved water quality will be the reduction in the variable costs of treatment plus the savings from postponing capital investment in enhanced water treatment infrastructure
- The costs of an ecosystem approach in the provision of clean water cannot be assumed to be less than the cost of the conventional alternatives
- The costs of an ecosystem approach may be large or small with respect to the accompanying investment in point source controls

Africa has more people living in rural areas than other regions and, thus, proportionately more Africans may be relying on community systems rather than large M&I systems. It is also true that Africa is starting from a much lower installed base in terms of water treatment facilities. In agricultural landscapes it is likely that non-point agricultural and point community sources will be important components of water quality problems. It is also likely that point sources will stretch up and down rivers and stream. This suggests that integrating point and non-point efforts will be critical. It also suggest that merely focussing on one or the other might lead to failure. Clearly, if existing poor water quality can be rectified or if existing good water quality can be maintained the potential gains in terms of potentially postponing investment in ever more sophisticated treatment methods can be realized. Protecting good quality water may take less time (and effort) than improving already poor water quality and, therefore, might be prioritized. Another factor here is that the risk of failure in turning around a situation with poor water quality looms large. This risk may argue in favour of simply installing the necessary treatment facilities given the need to protect the public and the uncertainty about outcomes with an ecosystem approach.

Conjecture on general trends and possibilities aside, the potential for improving domestic water supply and human health through an ecosystem approach in Africa will vary from one area to the next depending on a number of site-specific factors. In working to assess the potential of ecosystem management to make an important contribution to water quality and domestic water supply key information in scoping for opportunities include:

- Geography. Are community and/or municipal systems downstream from agricultural landscapes?
- Water Quality. What are the primary threats to domestic water quality: physical, chemical and biological?
- Pollutant Sources. To what extent do threats arise from non-point agricultural sources or point industrial, livestock or human sources? Are the threats from point or (agricultural) non-point sources? Do they stem from communities, agricultural activities or other source activities?
- Agricultural Practices and BMPs. What agricultural practices are contributing pollutants? Are there more sustainable practices that will reduce source pollution?
- Engineering. Is the downstream water supply source groundwater or surface water? Is storage involved? What level of supply, treatment and sanitation infrastructure is already in place?
- Health. What is the level of disease burden (morbidity and mortality) associated with waterborne disease and, more specifically, poor source water quality?

Finally, the literature is fairly clear on the linkages between land use and physical and chemical pollutants. Although limited, most of the economic analyses also examine the impacts of these pollutants on water treatment facilities and human health. In the case of biological pollutants there is a clear linkage to disease and human health. What was not found in the literature surveyed for this paper was a clear indication of the contribution of source water quality to this problem. Nor is the linkage between land use in agricultural landscapes and biological pathogens abundantly spelled out. Along the same lines there is quite a bit in the literature about how standard agricultural BMPs affect physical and chemical pollutant loads, but not so much about how these same BMPs would affect biological pollutants loads. Thus, further reconnaissance is needed regarding how prevalent and how solvable a problem biological pollutants are in agricultural landscapes in Africa.

6.3 Potential for PWS in Agricultural Landscapes in Africa

In this section the potential for developing PWS in agricultural landscapes is discussed with reference to information on existing and proposed PWS schemes, as well as likely future directions in this field.

While the focus is on Type 1 PWS between entities engaged in M&I or community water supply and farming communities, this is set off against the potential for Type 2 PWS.

6.3.1 Existing Schemes and Proposals

To date most of the experience with PWS comes from either developed regions – in particular the U.S. and Europe – or Latin America. In the most thorough global review to date, Landell-Mills et al. (2002) report on 61 cases of PWS. Of these 26 are classed as contracts for watershed protection or BMPs reflecting what is likely the set of cases, which meet the definition of Type 1 and Type 2 PWS above (Landell-Mills and Porras 2002). Of those listed, and particularly for those in developing countries, many are only at the proposal or pilot stage.

Five of the 61 cases of PWS found globally were from Africa (Landell-Mills and Porras 2002). One of these five cases is the stream flow reduction tax found in South Africa. As a tax on evapotranspiration the program is an economic incentive for the management of hydrological services but it is not a PWS as defined above. Four cases are watershed protection contracts: three in Malawi and one in Zimbabwe. Two of the Malawi PWS were described as pilots and one as proposed. All three appear to by Type 1 PWS. The first of the pilots involves payments from Escom, Malawi's national energy provider. The payments would go to local non-profits and government agencies that work through government and private forestlands to improve watershed management so as to reduce sedimentation of downstream facilities. The second pilot involved a number of water boards (including the large cities of Blantyre and Llilongwe) that proposed a similar set of payments for forest protection. The third Malawi case is a proposal for a Forest Department fee on five water boards, again for forest protection. The Zimbabwe case is a proposal for a number of downstream water users, largely irrigators, to pay watershed protection in headwater areas of the Runde and Save watersheds of south-eastern Zimbabwe. Recent follow-up work by IIED could not confirm that any of these initiatives are operational and most had not received support to move forward (Porras 2007).

Further research by IIED suggests a number of other PWS schemesthat are now underway or development (IIED 2007). The Working for Water program in South Africa was not included in the original set of PWS cases, but may be considered an example of Type 2 PWS. Since 1995 the Department of Water and Forestry (DWAF) has worked with local communities in the removal of over 1 million hectares of invasive aliens with a goal of restoring native vegetation and increasing low flows to the countries streams and rivers (Department of Water and Forestry 2007). The program receives a budgetary allocation of approximately $1 billion and has employed over 20,000 workers (IIED 2007; Department of Water and Forestry 2007) While the programme does not target water quality per se, increasing flow levels should improve water quality.

IIED also suggests that payments for watershed protection are being made by a utility in the Maloti-Drakensberg watershed although no further details are provided. Apparently efforts are underway to expand this scheme as part of the Maloti-Drakensberg Transfrontier Project, a World Bank funded project (IIED 2007). The GEF-funded Western Kenya Integrated Ecosystem Project may also constitute a Type 2 PWS, although no clear linkage to domestic water quality is made (IIED 2007). IIED also lists two other African initiatives involve wetland restoration, one paid for by Uganda Breweries and the other by South African government agencies (including DWAF) but these are not examples of payments for BMPs in agricultural landscapes and are not relevant to this paper.

Another source of information on African PWS cases comes from two meeting of the Katoomba Working Group on Environmental Services held in Africa in 2005 (Uganda) and 2006 (South Africa). These meetings were well-attended, suggesting considerable interest in the concept of PES in the region (The Katoomba Group). Countries involved in these meetings and case inventory work sponsored by the

Katoomba Group included Kenya, Tanzania, Uganda, Malawi, South Africa and Madagascar. Still, even at the 2006 meeting, the attendees were largely those working for African environmental non-profits and their developed country counterparts, as well as a number of representatives of US and South African environmental, forestry and water agencies. No representatives of African water boards, municipalities or health agencies were present.

The Katoomba meetings and their associated presentations, proceedings and inventories provide a useful swathe of information about the current extent of, and plan for, PES development in southern and eastern Africa, including PWS. Generally, the documents reveal little in the way of actual implementation of PES or PWS. Rather, there is much discussion of a conceptual approach towards PES and identification of the components or phases in developing a PES. The 2005 meeting concluded that efforts to date in East and Southern Africa occur in an ad hoc fashion typically in the form of small scale pilot projects (The Katoomba Group 2005). Out of 45 PES cases identified by Katoomba inventories in Uganda, Kenya, South Africa and Tanzania, money has changed hands in just nine cases (The Katoomba Group 2007). Of the ten 'water' cases identified only two are in operations and these are the Working for Water and Working for Wetlands project from South Africa covered above (The Katoomba Group 2007). Another two cases that are in planning in Madagascar should be added to this list for a total of 12 PWS cases identified by the Katoomba Group (Randimby and Razafintsalama 2006).

In South Africa where efforts are relatively well-advanced, PES is described as 'on the agenda' and in planning and design but with limited implementation (King 2006). Water is mentioned as a focus in South Africa, with the public sector engaging through the Working for Water initiative and the private sector involved to date only in project planning. The science behind PWS is described as 'good' unlike with biodiversity. Eight of the twelve PWS cases identified by the Katoomba Group inventories are in South Africa. In Kenya, in contrast only one PWS is mentioned – the GEF-funded project mentioned above – which is described as in the planning stage (Mwangi and Mutunga 2007). Two projects in Tanzania are also in planning. One is part of an IUCN project in the Pangani river basin and would potentially include payments by M&I water suppliers. The other project is a collaboration between CARE, WWF and IIED that would involve payments by, amongst others, M&I water suppliers for BMPs to land managers in the Ruvu and Sigi river basins. (Scurrah-Ehrhart 2006). Of the Madagascar projects one involves potential payments by the Eau Vive water bottling company to the Communal Authority of Andranovelona for watershed protection (Randimby and Razafintsalama 2006).

Additional efforts were reported at the Bellagio meeting to design PWS schemes for downstream water quality in the Aberdares watershed that supplies water to Nairobi, Kenya and for the Klein Berg River Catchment that supplies water to a reservoir used, amongst others, by the City of Cape Town in South Africa (Marais 2007; Pagiola 2007). Independently, it also appears that the City of Cape Town itself is interested in working with Cape Nature to pursue PWS opportunities to improve its M&I situation(Clarke 2007).

In sum, it appears that limited experience with PWS exists in Africa. Further, there are no successful examples of Type 2 PWS of the type envisioned here: agreements between domestic water suppliers and farmers in an agricultural landscape. Nonetheless, a number of efforts have been made and various proposals are underway. In addition, interest appears high amongst African and donor organizations, and for the region PWS seems to be generally agreed to be an important focus within broader efforts on PES.

6.3.2 Obstacles

Drawing on the Katoomba presentations and proceedings a rather long list of obstacles and barriers to the development of PES (and PWS) in Africa emerge (The Katoomba Group 2005, 2006, 2007):

- limited information including scientific understanding (biodiversity), market information on likely buyers and sellers, and economic information on costs and benefits
- no centralized information on PES, no determination of 'best practice' and no technical backstopping
- lack of capacity to design and manage projects, capacity-building not built into projects
- government does not have capacity to determine where, when and in what form PES are appropriate
- absence of institutions and policies to support on-the-ground implementation
- minimal government participation and weak legal and regulatory frameworks
- land ownership and tenure issues
- lack of requisite partnerships
- high transaction costs
- lack of existing, viable models and
- poor articulation and marketing of the business case for investors

Overcoming these obstacles will not be easy. One difficulty faced in the current environment is moving the project development and implementation out of the policy and research community and into the business, finance and legal community. The tools required to conceive of and promote PES and PWS as a useful approach are not necessarily those required to develop on-the-ground projects. This is particularly true for Type I PWS. These involve appraisal, negotiation and legal agreements between buyers and sellers. This is somewhat akin to traditional international development projects except that this involves convincing a municipality (for example) to buy a service, rather than convincing a government to accept a long-term loan or a grant. This is also different than the traditional conservation project that involves international donors, government agencies, and communities and often involves grant money. In particular, the experience with GEF project grants is not applicable here. While GEF grants may be justified on the basis of global payments for conservation of in-country biodiversity they are in some sense the antithesis of a PWS project. PWS projects are completely local and Type 2 PWS projects rely on agreements between local entities. Local government entities may therefore be confused between such radically different ideas, all marketed under a single PES slogan. All the more reason for working to separate out PWS Type 1 efforts from broader efforts at promotion of PES.

Until a transition happens in the style through which PWS schemes are developed many are likely to continue to be 'projects' consisting of proposals and studies, and not 'transactions'. This may be part of what is meant by Katoomba participants when they comment that the 'business case' is lacking. It is not just the 'case' that may be lacking, but an understanding of the project cycle and requisite skill set for developing what is essentially a voluntary business transaction.

6.3.3 Prospects

Likely prospects for Type 1 PWS in Africa probably can be divided into two types: large M&I systems and small community-based water supply systems. These are discussed below with a view to identifying characteristics that would need to be assessed in order to prioritize likely prospects.

Large urban areas and M&I water suppliers are prime prospects for engaging in payment programs with land managers in the upstream watershed. While there is a distinction between surface and ground water systems in terms of susceptibility to physical pollutants there is no reason to suppose that groundwater systems are necessarily protected from chemical and biological pollutants. Still, surface water systems might be prioritized on this basis. The existence of a sole M&I provider would also be a preferred attribute in the development of a PWS, particularly in the development of a country or regional pilot project.

Due consideration of point and non-point sources will be required in designing a project that will make a measurable difference in water quality. Care and thought is required in deciding what policy approach will be taken towards point sources. The full range of approaches provided in Section 4.1 should likely be considered within the bounds of the local context. It is unlikely that merely setting up a PWS for non-point sources would be a useful endeavour, unless the upstream watershed is largely uninhabited. As discussed, scoping for such projects might well first focus on municipalities that have limited infrastructure for water treatment at present.

For a large urban area it is likely that the corresponding source watershed will be of significant size, and likely contain a variety of land uses. For this reason, PWS related to non-point sources may require an intermediary that specializes in working with farmers and foresters. The City of New York did largely implement its Whole Farm Program on its own. Whether African municipalities would want to take on this role and build their capacity to do so is questionable. To the extent that existing extension agencies exist they might be better placed to work through the implementation of PWS schemes in the field. Research and experimentation on BMPs and their impacts on water quality will also likely be needed, depending on prior experience. Agency, university and research center expertise will be required and partnerships with developed country research centers will likely be of value. It is also advisable to ensure that other economic incentives in the agricultural landscape act in concert with the PWS and selected BMPs. These may consist of bundled payments for other ecosystem services or other government subsidy or funding programs. Bundling these into a single package of payments to program participants would be useful.

The other type of PWS that could be promoted would revolve around small-scale local community systems. Important considerations in this case are likely to be geographic scale, population density and existing level of water treatment. Geographic scale may be an important factor in facilitating a clear connection between BMPs and water quality. Working in smaller headwater catchments has an obvious attraction in terms of limiting the spatial extent for source pollution. Narrow river valleys where pollutants move to the river quickly and can be monitored along its length might also be an attractive option. A large scale, open floodplain system would probably be the least desirable in terms of being able to track and monitor cause and effect.

Density of population has two aspects worth consideration. First, is density of the general population. Areas where the population is largely centered in towns and villages, with larger, commercial agricultural operations in the rural areas would serve to reduce transaction costs. The numbers of farmers or operators that would need to be incorporated into a meaningful PWS scheme would be reduced over an area with many smaller farms. Similarly, the concentration of population in town centers would mean larger numbers of people could be served by centralized improvements in WS&S. A key factor here is the sanitation piece. Where the population is largely rural, living on the small farm, with few town centers inadequate household sanitation would suggest the need for an integrated program that treats both agricultural practices and household wastes. The costs of latrines and other approaches to what might be called rural 'non-point' source sanitation problems may not high when compared to the capital costs of larger scale sanitation infrastructure, but the transaction costs (and risks of failure) of having to mount extension campaigns to encourage adoption of new technologies would be substantial. Of course if a BMP and sanitation extension programs are combined this might lower transaction costs.

This discussion highlights a major issue on the sell side of a PWS scheme in Africa. In the typical agricultural landscape farms dot the landscape and in some areas are very small in size. PWS schemes therefore will often be subject to the large numbers problem, raising the transaction costs of implementation. Whether or not an area has the requisite social capital to enable the spread and adoption of a new approach, such as payments for BMPS, will be an important question in determining feasibility. This applies for both Type 1 and Type 2 PWS. This is not just an issue with recruitment and contracting, but also with efforts to monitor compliance. While PWS schemes that revolve around land cover and vegetation (such as the Mexican national level PWS) can carry out monitoring using remote sensing this may not be feasible with agricultural BMPs that seek to lower non-point source pollution.

On the buyer side the numbers problem can also occur. A Type 1 PWS would likely require grouping communities in a downstream zone of influence and ensuring that all subscribe to a PWS scheme. A strategy for coping with the problem of free-riding due to the large numbers problem would then be necessary. Just as with large scale PWS involving major urban M&I suppliers, if there are only two or three entities involved and one decides not to participate that may well end the initiative. But if there are 50 communities involved it is inevitable that some will choose to free-ride by not joining an agreement. While this may not be terminal to such an agreement, free-riding can be corrosive – particularly if the benefits of a PWS scheme take years to materialize there will be pressure over time as communities considering pulling out of the agreement.

This also raises the question of whether in most African rural areas it is reasonable to even consider a Type 1 PWS? Do such communities understand the issues and are they well enough organized and do they have the financial or in-kind resources to enter into a payment scheme? The examples cited earlier from Honduras and Nicaragua suggest that this should not be dismissed immediately. However, it is important to stress that the research shows that people in those communities are predisposed to believe that protecting forests improves water quality and quantity – this is probably an important factor in achieving adoption of a Type 1 PWS (Kosoy et al. 2007). While African populations may have the same beliefs about forests and water it would be useful to survey public attitudes towards agriculture and water quality to establish the likely credibility of the sorts of claims that would be made in marketing a Type 1 PWS to a community.

But more to the point, the question of resources is an important one. If African communities are in general not already providing basic WS&S to their constituents is it likely they would wish to make payments to farmers in the hopes that over a period of years these payments would be adopted and reduce (but not eliminate) water quality threats? This may not be a reasonable proposition in many places and, therefore, a Type 2 PWS scheme may be a more likely alternative. Comparing major urban centers with rural towns it may make sense to suggest that Type 2 PWS schemes are more appropriate in rural areas, while Type 1 schemes may be more likely to be adopted in urban centers.

A final question concerns the implications of the level of existing water supply and treatment infrastructure for rural PWS schemes. As stated throughout this paper, where treatment is non-existent or minimal and water quality remains high, every effort should be made to maintain water quality. A Type 2 PWS scheme may help by instituting payments to maintain existing land uses (forest or agricultural). The difficulty of course is defining the threat (of change in land use or practices) that merits the payment of an incentive. Apart from this simple, ideal case however would be the bulk of situations.

In the case where there is effectively no water treatment – meaning either that there is no centralized supply and water is taken directly from water bodies, or where there is piped supply (in common or to the household) but no treatment facilities – changes in pollutant levels can be expected to directly impact human health. Whether this is a linear relationship or not is a central question. In other words does a 50% reduction in biological pollutants lead to a 50% drop in incidence of diarrhoeal disease? Or does it merely result in a 10% drop in incidence? Most importantly, are there thresholds where these relationships change so that up to some point improvements in water quality make little difference but once a threshold is reached incidence of disease tails off quickly? Answers to these questions will be important to informing likely targets of a PWS scheme and the likelihood of success in terms of required adoption rates for payments and BMPs. In other words in the case where there is no water treatment the risk and the potential reward of engaging in a PWS scheme would both be high.

A further consideration would be the time frame for results from the adoption of BMPs. If the time scale is fairly short then undertaking a PWS scheme may be a reasonable short-term strategy, even if in the long term the community is likely to receive improved WS&S facilities. And, if substantial progress is made prior then the requirements of such facilities in the water treatment area may be reduced.

For communities where water treatment infrastructure is already in place it would seem that the reverse applies; the risk and the potential reward of investing in a PWS scheme would be low. This, given that marginal improvements in water quality are likely to only reduce variable water treatment costs – and at a decreasing rate. Therefore, communities with water treatment facilities would not be a priority in scoping out likely places to develop PWS schemes.

Given the knowledge and financing required it is likely that rural PWS schemes in Africa will not arise in the absence of outside intervention. For example, the community-level schemes in Central America cited earlier and in Table 14 were developed with development assistance from the Swiss government (Kosoy et al. 2007). Transaction costs of such schemes can be high – in the Jesus de Otrero case the development costs of the program were thirty times the annual funds collected (Kosoy et al. 2007). Given the obstacles cited earlier and the resulting transaction costs this conclusion is probably also true of PWS schemes in urban areas. For example, the Nairobi scheme mentioned earlier is receiving technical support from the World Bank. The Cape province and Cape Town examples in South Africa involve a number of government and regional conservation agencies. All of this suggests the possible need for organized intervention instead of ad hoc intervention.

Specific questions that will need to be answered in evaluating the social and economic feasibility of developing a PWS scheme include:

- Cost. What is the incremental cost to farmers to switch to BMPs?
- Relevance. What is the existing level of investment in water treatment infrastructure?
- Willingness to pay. Is there evidence that downstream utilities, water boards or other users are willing and able to pay to improve water quality?
- Complexity. Is the number of buyers small enough or are they socially cohesive enough to eliminate free-riding behaviour? Is the number of farmers small enough or does enough social capital exist to suggest that widespread adoption of a payment system for BMPs?
- Capacity. Is there the human resource and organizational capacity to develop a payment scheme? Does a likely intermediary already exist or will it need to be created?

6.4 Legal, Regulatory and Contractual Elements of PWS

The institutional arrangements and incentive mechanisms for regulating water quality are presented in Section 4.1. No thorough review of current laws and regulation in Africa was possible for this paper. However, generalizing, it is likely that most countries are at the beginning stages of such efforts, perhaps with laws on the books that proscribe a command and control approach to regulating point source effluent and in some cases perhaps with market-based instruments implementing the polluter pays principle. Just as likely is that implementation, monitoring and evaluation vary with the overall level of economic development and human resources of the country. It is also unlikely that regulation and implementation of such is present for non-point agricultural source pollution.

In this regard the proposition of a Type 1 or Type 2 PWS will represent an evolution of current thinking and approaches to pollution management. That said, as long as the approach is feasible there is no reason why a country cannot leapfrog to a more efficient arrangement or mechanism. One of the advantages of a Type 1 PWS – where upstream and downstream entities negotiate a mutually satisfactory arrangement – is that no special legislation is required. All that is required is for the two parties to negotiate an arrangement. While (as described below) it would be good practice to arrive at a written contract there is no requirement of this in the basic approach. A handshake agreement would in theory be sufficient.

The Bellagio expert group debated the need for special policy and legislative development to underpin PWS (including Type 1 and Type 2) and largely concluded this was not necessary. While it may be advantageous to make PWS part of government policy, specific legislation is not necessarily required to launch a PWS scheme (Appleton and Mayers 2007). The Katoomba inventories of PES in Africa note that policies that establish the right to buy and sell 'ecosystem stewardship services' have not been essential in pilot projects – though they might be limiting in expanding these pilots (The Katoomba Group 2007). The Bellagio group suggests that there are two specific exceptions to this rule. First, at both the local and national levels legislation or regulations that provide funding for a PWS may be necessary and advantageous. For example, the Mexican national payments system involved protracted political discussion and in the end the Federal Rights Law was amended to permit the assignment of a specific amount of the water fees collected by the government to the PWS scheme (Muñoz-Piña et al. In Press). In the case of the Quito Water Fund, recent passing of a local ordinance has succeeded in securing the funding stream from the municipality – causing this scheme to change from being an endowment fund employing earnings to one that can spend funds as they are received (Arcila 2007). The other exception is when legislation is required to create a proper intermediary organization or to legally develop consortia of buyers or sellers (Appleton and Mayers 2007).

Property rights for specific hydrological services produced by land management typically do not exist. Property or use rights regarding land management typically do exist. Thus, the key legal element in PWS is the contract, either between the water supplier and the land manager in a Type 1 PWS or between the intermediary (or other entity doing the purchasing) and the land manager in a Type II PWS. A PWS contract typically calls for the land manager to undertake a specific land use and/or land management activity. In Mexico, participants in the national CONAFOR scheme are paid to not deforest their land. In Costa Rica, participants in the national FONAFIFO protection program are paid to not deforest and to undertake a number of specific land protection activities, including patrolling and managing fire breaks. An alternative is to specify specific indicators of performance in terms of downstream services. One scheme under the RUPES program is working on a contract which specifies that payments will be made when downstream sediment rates reach a specified level (van Noordjwik 2007). It is likely that application of PWS in Africa would pursue a similar logic of the buyer paying the seller for implementation of agricultural BMPs, rather than for services per se.

Given the nature of the relationships between buyer and seller, or between intermediary and seller, it is therefore good practice for the buyer and seller to arrive at a contract, i.e. an agreement between the buyer and seller (the parties to the agreement) on the terms of the transaction. Contracts clarify the roles and responsibilities of the parties. The advantages of having a written contract include:

- buyer and seller have a clear and physical record of the terms of the deal, and can revisit the agreement to refresh their memory or renegotiate as necessary
- buyer has physical evidence of the transaction to offer to the ultimate provider of funds

- third parties (not involved in the transaction) and evaluators are informed of the key elements of the deal
- the agreement can be recorded in the relevant property registry, an essential step for longer-term contracts held with landowners in jurisdictions where property registries exist – this notices potential buyers of the property

The seller does not need to be the landowner, but should at a minimum be the 'proprietor' of the land meaning that the seller has the de facto right to manage the land and exclude others from the lands that will be under contract (Schlager and Ostrom 1992). This means they have the practical ability to undertake all management actions required under the contract and can exclude others from accessing and using the land and its resources. Note that the right of alienation (right to sell) is not required.

These contractual requirements may not always be easy to implement in rural parts of Africa. However, they should probably still be requirements, particularly as they provide the buyer with a minimum level of assurance that the seller understands and will comply with the program.

As the agricultural BMPs are the cause of the desired effect (hydrological services) these are contracts not for services but for the performance of activities that cause (or produce) the services. In such contracts the risk is fully borne by the buyer. Typically, it is the buyer that is 'selling' the program to land managers and, therefore, presumably in command of knowledge regarding the cause and effect of the land/water interactions produced by the BMPs. The distribution of risk therefore seems appropriate. This lesson, while learned in the context of developed countries and (largely) Latin America remains applicable in Africa. When an intermediary is involved, if the intermediary is the proponent or entity that has provided the motivation behind the PWS then it would be advisable that the intermediary make sure that the ultimate buyer (i.e. the entity providing the funds) is on notice that there is no guarantee of service provision (only implementation of the BMPs).

For both national-level and site-specific schemes negotiations between buyers and sellers have often taken place in advance of the onset of a program. Prices are often thus set through political process (national) or cooperative negotiation (local). At the national level, political or legal concerns over price discrimination across different regions or lands have led to the selection of standard fixed price offer systems for Type 2 PWS/PES, or at most a limited tiered pricing systems. Political debate over the Mexican scheme ultimately led to the selection of a two-tiered system where cloud forest lands are paid at a different rate to all other lands (Muñoz-Piña 2007). Such systems are 'sticky' given that once agreement is reached, there is often a high barrier to reopening such high level political negotiations. For example, in Costa Rica when the country switched from a system of subsidies for forest conservation and reforestation to one of payments for environmental services the per hectare payments remained effectively the same (Rojas and Aylward 2003).

With regard to determining the amount of payments most schemes to date have focused on identifying the opportunity cost of net benefits foregone by the landowner from their prior use of the land. In theory this represents a floor for payments. A theoretical ceiling for payments would be the value of the hydrological services – though in most cases this will be unknown. Most programs have relied on estimates of opportunity cost as these are far easier to calculate (Rojas and Aylward 2003).

For site-specific schemes with well-defined service buyers, efficiency becomes more critical to success and buyers and sellers have been more willing to differentiate price based on the service potential of lands. In a Fundación Natura project in Los Negros, Bolivia a fixed price system was negotiated in the first year, but once that was successful Natura was able to differentiate price based on type of forest and expected service benefits in subsequent years (Asquith, Vargas, and Wunder 2007).

Anecdotal reports suggest that in some cases – typically after a program has been up and running – sellers have been willing to accept less than opportunity costs. It is hard to evaluate these claims given that the actual opportunity costs for a given landowner will vary around the costs as calculated by consultants. With regard to PWS in which payments are made for forest conservation it may be the case that the land manager really had no intention of engaging in other land uses and thus is willing to take a payment less than the opportunity cost of idling land.

There is little or no experience that defines an optimum duration for a PWS contract. There are however some important points to consider about the length of the contract which must be considered in planning the PWS initiative and during any negotiation process. These include:

- There maybe a significant time lag between implementation of the BMP and the emergence of improved water quality. This suggests that where possible contracts need to be longer rather than shorter, and ensure that the maturity of the contract matches that of the expected time frame for provision of the services.
- Prices for agricultural commodities and inputs change over time. This suggests that long-term contracts run the risk of becoming redundant.
- Negotiating contracts costs money. This suggests that medium to long-term contracts are more desirable.

Contracts should also explicitly allocate the risk from a natural catastrophe on the land in PWS schemes - fire, flooding, disease, etc. In the Costa Rican national environmental services program, for example, the risk is borne by the landowner, i.e. following a verified event the contract is terminated. This can pose hardships for those landowners whose alternative source of income would have come from the (now degraded) lands. One solution is for such programs to provide an insurance program that the landowner can buy into, pooling their risk with others in the program.

Designing clear and effective contracts that avoid the exploitation of the seller by the buyer (and vice versa) is of crucial importance as PWS programs are intended to be long-term, i.e. where the buyer will want to maintain existing contracts and sign new contracts over time. Buyers of water quality are likely to desire PWS contracts in perpetuity, where land purchase is not a practical alternative. As such the perceived fairness of agreements by sellers will be an important determinant of future outcomes and buyers will want to make every effort to ensure that contracts are both fair and efficient.

Ensuring that the legal regime is sufficient to support such contracts will be important. However it may be debated to what extent a buyer in a context such as Africa is likely to take legal action to enforce the terms of a contract. More likely it is incumbent on the buyer to understand the cause and effect relationships involved, to monitor the performance of the seller and to ensure that the contract specifies the consequences of a failure to perform. The most likely recourse upon non-performance is likely to simply be cancellation or non-renewal of the contract, rather than some type of punitive action. This is, of course, in the case where sellers are small-scale agriculturalists. Should PWS be developed with larger landowners or corporations, the posting of a performance bond may well serve to provide the necessary assurances to the buyer that the seller will perform.

Obviously, the application of PWS in Africa faces significant challenges in terms of developing what are relatively sophisticated contracts with rural landowners that may have limited education. However, the great benefit of PWS is that the action required is that of implementing a BMP. In this regard, there is considerable experience in Africa, as in other developing countries, with the propagation of agricultural innovations. That landowners will be provided a cash or in-kind payment in return should only help to motivate landowners to adopt and implement these BMPs.

6.5 Conclusions

The gaps in knowledge regarding the topics considered in this paper are considerable. The key to the further development of PWS and in particular PWS in the case of agricultural landscapes and domestic water quality will be a clear conceptual and analytical framework that can accommodate evidence as it becomes available. Only in this manner can the evidence be placed into an evolving picture of where and how and to what degree the PWS approach is applicable. This paper is not an exhaustive review of the evidence but it takes a step in that direction and in so doing suggests that facile assumptions regarding the (complex) linkages along the way from field to tap and ignorance regarding the economic evidence that is available may lead to an excessive optimism about the prospects for PWS.

On the central question of whether ecosystem management is a cost-effective alternative to water treatment the paper (and the evidence) remains inconclusive. While it is likely that ecosystem management will be preferable in some cases, so it is likely that in others that an infrastructure approach is of merit. Even the suggestion that PWS will be more attractive at smaller-scales and at the scale of large municipalities is largely a hypothesis at this point. There are simply not enough cases of thorough cost comparisons between mitigation and avoidance approaches. It is therefore not possible to generalize about which approach is to be preferred. Instead it will be necessary to compare costs of reaching different benefit levels in particular sites in order to conclude whether ecosystem management is an attractive alternative to water treatment.

As the saying goes, ‘find a big enough hammer and everything looks like a nail.’ PWS (and PES generally) is clearly a tool in the toolkit for land and water managers. The overall conclusion of the paper must be that when it comes to downstream water resource management PWS is not the big hammer, but a tool in the toolkit – and like all tools will be appropriate in some situations and inappropriate in others. While much work remains to better define the correct use of the tool, the review suggests that there is merit in the approach and that more experimentation is called for to better calibrate the tool. Further, there seems no reason to think that an approach that appears to be working in Latin America would not work in Africa. Still, lacking any thorough evaluation of the many incipient PWS schemes ongoing caution is required prior to any massive upscaling and replication. PWS remain very much in their pilot phase of development.

It is also important to stress a key difference between the bulk of PWS in developed countries and Latin America and that considered here for Africa. There is a difference between paying land managers to continue to maintain forest cover in urban watersheds and paying farmers to adopt BMPs. In the first case, there is a high degree of confidence that forest protection will avert a major decline in water quality and avert or postpone water treatment costs. The question in this case is whether the payment is really needed since the forest was being protected anyway. In the second case, it is not as clear that the contracted-for-activity will have enough impact on water quality to have a major impact on water treatment costs. Again, much will depend on the context but it is worth being cautious on this front. Site-specific investigation and modelling may be warranted in order to provide assurances on cause and effect. This, as PWS schemes will be asking farmers to alter their farming systems, some times substantially, and it will be important that this be done based on credible evidence rather than mere hypothesis or conventional wisdom gleaned from an international workshop on environmental services.

Finally, this review suggests the need for a comprehensive, planned approach to developing PWS in a region, rather than an ad hoc approach. Existing reviews by those working in this field to date suggest that in Africa there is a lack of clear examples, or models, to follow. The global review of the literature and PWS cases presented here suggests this is as much a failure of the global brain trust on PWS as it is any lacking on the part of African efforts. The conceptualization of what is called a Type I PWS here – were a prospective beneficiary pays land managers to alter their behaviour in the expectation of receiving downstream hydrological services – has been largely mangled and misunderstood by enthusiasts. This paper suggests that such PWS schemes can be placed in an already existing typology of methods that has a long history within environmental and natural resource economics. Proper understanding of the economic basis of a Type 1 PWS leads then to a more reasoned approach to assessing when a PWS scheme is the right tool and when one of the other tools might be the better choice.

The principal implication of understanding that what is 'new' about PWS is a Type 1 PWS or Coasian bargain, is that this approach is a voluntary, not a regulatory approach. It also connects PWS firmly with a large literature and experience on transaction costs and, makes it clear that if transaction costs are high this approach will not work. So if examples of Type 1 PWS are to be developed in Africa it is abundantly clear that transaction costs need to be lowered. The absence of Type 1 PWS simply reveals that transaction costs are currently so high as to impede development of such agreements. An ad hoc approach might well find a case – such as the Vittel-Perrier case in France – where the net benefits of arriving at a voluntary agreement are so huge that any amount of transaction costs can be accommodated. But this will not help in the development of the 2nd, 3rd and subsequent cases. To maximize the chances of being good at developing PWS (instead of merely being lucky) at least three elements would be useful:

- a comprehensive plan for identifying and classifying potential sites,
- a clear set of methods for assessing costs, benefits and transaction costs
- a clear conceptual framework for the development of buyer-seller agreements (Type 1) or generalized BMP subsidy programs (Type 2), including a focussed plan for how to lower transaction costs

Clearly this paper implies the need for the development of institutional capacity that could help interested parties develop PWS Type 1 (and PWS Type 2) in African agricultural landscapes with a focus on domestic water supply. Nonetheless, it is important to stress that the topic (water quality for water supply), the context (agricultural landscapes) and the geography (sub-Saharan Africa) covered in this paper are merely subsets of the larger global issue of how to develop PWS. It would of course likely be more efficient to first create a global institutional capacity, and then work on how to devolve or deploy this capacity to work on the issues confronted in this paper. However, the effort is somewhat divisible and an effort at focussing on, say, Type 1 PWS in large urban areas could be a discrete task for which the creation of specific capacity and pilot projects would be useful. Regardless, it would be advisable to create this capacity away, or distinct, from similar capacity for the other major ecosystem service categories, such as carbon sequestration, and biodiversity. Carbon and biodiversity are major global issues and can taken on geopolitical overtones. Further they largely involve the transfer of foreign capital – and in the case of carbon lend themselves to considerable participation by the private sector with all that brings. PWS is a distinctively local problem, will often involve the participation of public entities and as such requires a much more local approach and solution.

7. References

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