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Wetlands as natural assets

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Abstract As the services of wetlands are uniquely related to hydrological processes, they result in a wide range of benefits to humankind. Although this facilitates the characterization of wetlands as natural assets, there are measurement problems. Because wetland services are nearly always non-marketed, they need to be explicitly valued to determine the trade-offs between development and conservation of wetlands. Two case studies, a floodplain in northern Nigeria and mangroves in southern Thailand, illustrate the issues involved. In the case of Nigeria, the natural capital is the river floodplain, and the trade-off is the upstream water diversion compared to the downstream flooding benefits to farming, fishery and forestry, as well as groundwater recharge. In the case of Thailand, the natural capital is the mangrove system, and the trade-off is the conversion to shrimp farms, as opposed to the mangrove benefits of locally harvested products, habitat-fishery linkages and storm protection.

Key words economic valuation; ecosystem services; floodplain; mangrove; hydrological services; natural asset; Nigeria; Thailand; wetlands

Les zones humides en tant que biens naturels

Résumé Bien que les services des zones humides soient particulièrement associés aux processus hydrologiques, ils produisent toute une gamme de bénéfices pour l'humanité. Si cela facilite la caractérisation des zones humides en tant que biens naturels, il y a cependant des problèmes de mesure. Parce que les services rendus par les zones humides ne sont presque jamais sur des marchés, ils doivent être évalués de façon explicite pour déterminer le compromis entre le développement et la conservation des zones humides. Deux études de cas, une zone inondable au nord du Nigéria et des mangroves au sud de la Thaïlande, illustrent les problèmes rencontrés. Dans le cas du Nigéria, le bien naturel est la zone inondable et le compromis est le détournement d'eau en amont comparé aux bénéfices, apportés par les inondations en aval, pour l'agriculture, la pisciculture et l'exploitation forestière, ainsi que la recharge des nappes. Dans le cas de la Thaïlande, le bien naturel est le système de mangroves et le compromis est la conversion à l'élevage de crevettes opposé aux bénéfices venant des mangroves : récoltes locales de produits, liens entre habitat et pisciculture et protection contres les tempêtes.

Mots clefs évaluation économique; services écologiques; zone inondable; mangrove; services hydrologiques; bien naturel; Nigéria; Thaïlande; zones humides

INTRODUCTION

Because ecosystems generate services that contribute to human welfare, they can be considered a form of wealth. The recent literature on ecological services also implies that ecosystems are assets that produce a flow of beneficial goods and services over time (Daily 1997, Daily *et al*. 2000, World Resources Institute 2001, Pagiola *et al*. 2004, Heal *et al*. 2005, MEA 2005, Barbier 2007). For example, as Daily *et al*. (2000, p. 395) state, "*the world's ecosystems are capital assets. If properly managed, they yield a flow of vital services, including the production of goods (such as seafood and timber), life support processes (such as pollination and water purification), and life-fulfilling conditions (such as beauty and serenity)*." Ecosystems should therefore be treated as an important asset in an economy, and, in principle, ecosystem services should be valued in a similar manner as any form of wealth. That is, regardless of whether or not there exists a market

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for the goods and services produced by ecosystems, their social value must equal the discounted net present value (NPV) of these flows (Barbier 2007, 2009).

The concept of ecosystems as natural assets is already having an influence on how policymakers view wetlands. For example, the Changwon Declaration of the 10th Conference of the Parties of the Ramsar Convention states that "wetlands are vital parts of the natural infrastructure we need for addressing climate change" (Ramsar Convention 2008). Given that wetlands, which comprise coastal wetlands, freshwater swamps and marshes (including floodplains), and peatlands, amount to 6–8 million km^2 globally (Mitsch *et al*. 2009), these ecosystems are an abundant source of natural capital.

The purpose of this paper is to develop further this approach of viewing wetland ecosystems as natural assets. Because they provide a vast array of hydrology-related services that contribute to human welfare, wetlands should be viewed as "natural infrastructure", just like other wealth in an economy. As a consequence, when wetlands are converted or exploited for various economic activities, an explicit trade-off is being made between, on the one hand, the loss of hydrological services from degraded or destroyed wetland ecosystems, and, on the other, the commercial and other benefits gained from the new economic activities. What is needed, therefore, is to make this trade-off explicit by measuring, or valuing, the loss in benefits that occur when wetlands are converted or damaged. The aim of this paper is to explain this approach, to examine the special challenges involved in valuing wetland ecosystem services, and to illustrate the issues involved with two case study examples, mangroves in Thailand and a floodplain in northern Nigeria. In the case of Thailand, the natural capital is the mangrove system, and the trade-off is the conversion to shrimp farms as opposed to the mangrove benefits of locally-harvested products, habitat-fishery linkages and storm protection. In the case of Nigeria, the trade-off is between water allocated for upstream developments and maintaining a downstream floodplain that yields benefits to local communities through flood recession agriculture, fishing, fuel wood and groundwater recharge. In both cases, valuing hydrology-related wetland services is instrumental in determining whether wetland environments should be protected or restored as natural capital.

VALUING WETLAND ECOSYSTEM SERVICES

In identifying the ecosystem services provided by natural environments, such as wetlands, a common practice is to adopt the broad definition of the Millennium Ecosystem Assessment (MEA 2005) that "ecosystem services are the benefits people obtain from ecosystems". Thus the term "ecosystem services" is usually interpreted to imply the contribution of nature to a variety of "goods and services", which in economics would normally be classified under three different categories (Barbier 2007): (i) "goods" (e.g. products obtained from ecosystems, such as resource harvests, water and genetic material); (ii) "services" (e.g. recreational and tourism benefits or certain ecological regulatory and habitat functions, such as water purification, climate regulation, erosion control and habitat provision); and (iii) cultural benefits (e.g. spiritual and religious beliefs, heritage values).

To assess the contribution of nature in providing such "goods and services", one needs to measure its impact on human welfare, or, as Freeman (2003, p. 7) succinctly puts it: "*The economic value of resource-environmental systems resides in the contributions that the ecosystem functions and services make to human well-being*", and consequently, "*the basis for deriving measures of the economic value of changes in resource-environmental systems is the effects of the changes on human welfare*." Similarly, Boyd and Banzhof (2007, p. 619) state that "*final ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being*."

Regardless of how one defines and classifies ecosystem services, as a report from the US National Academy of Science has emphasized, "*the fundamental challenge of valuing ecosystem services lies in providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e. goods and services) derived by humanity, and their subsequent values*" (Heal *et al*. 2005, p. 2). Table 1 provides some examples of how specific wetland ecosystem services are linked to the underlying ecological structure and functions underlying each service. It also cites, where possible, economic studies that have estimated the values arising from the service. The list of studies in Table 1 is not inclusive; for more comprehensive summaries of the literature on economic valuation of wetlands, see e.g. Barbier (1997),

Table 1 Examples of wetland ecosystem services and valuation studies.

Ecosystem structure and function	Ecosystem services	Valuation examples
Attenuates and/or dissipates waves, buffers wind	Coastal protection	Badola and Hussein (2005), Barbier (2007), Costanza *et al*. (2008), Das and Vincent (2009).
Provides sediment stabilization and soil retention	Erosion control	Sathirathai and Barbier (2001)
Water flow regulation and control	Flood protection	Brouwer and van Elk (2004)
Groundwater recharge/discharge	Water supply	Acharya and Barbier (2000, 2002).
Provides nutrient and pollution uptake, as well as retention, particle deposition	Water purification	Breaux *et al*. (1995), Gren (1995), Byström (2000), Yang *et al*. (2008), Jenkins *et al*. (2010)
Generates biogeochemical activity, sedimentation, biological productivity	Carbon sequestration	Jenkins *et al*. (2010)
Climate regulation and stabilization	Maintenance of temperature, precipitation	
Generates biological productivity and diversity	Raw materials and food	Ruitenbeek (1994), King and Lester (1995), Naylor and Drew (1998), Janssen and Padilla (1999), Sathirathai and Barbier (2001), Islam and Braden (2006).
Provides suitable reproductive habitat and nursery grounds, sheltered living space	Maintains fishing, hunting and foraging activities	Freeman (1991), Swallow (1994), Bell (1997), Barbier and Strand (1998), Janssen and Padilla (1999), Johnston *et al*. (2002), Barbier (2007), Smith (2007), Aburto-Oropeza *et al*. (2008), Sanchirico and Mumby (2009).
Provides unique and aesthetic landscape, suitable habitat for diverse fauna and flora	Tourism, recreation, education, and research	Creel and Loomis (1992), Bergstrom and Stoll (1993), Cooper and Loomis (1993), King and Lester (1995), Bateman and Langford (1997), Hammitt *et al*. (2001), Johnston *et al*. (2002), Carlsson *et al*. (2003), Othman *et al*. (2004), Brouwer and Bateman (2005), Birol *et al*. (2006), Birol and Cox (2007), Do and Bennet (2008), Jenkins *et al*. (2010).
Provides unique and aesthetic landscape of cultural, historic or spiritual meaning	Culture, spiritual and religious benefits, bequest values	Bateman and Langford (1997), Naylor and Drew (1998), Kwak *et al*. (2007).

Woodward and Wiu (2001), Brander *et al*. (2006) and Turner *et al*. (2008). Nevertheless, the valuation studies are representative of the literature, and thus instructive.

For one, as the studies in Table 1 indicate, wetland valuation studies have tended to focus on only a few ecosystem services, such as recreation, coastal habitat-fishery linkages, raw materials and food production, and water purification. In recent years, a handful of more reliable estimates of the storm protection service of coastal wetlands have also emerged. But, for a number of important wetland ecosystem services, very few or no valuation studies exist.

In addition, current valuation studies also illustrate the extent to which wetland ecosystem services are uniquely related to hydrological processes. As emphasized by Mitsch *et al*. (2009, p. 2), these processes are key to the functioning and structure of wetland ecosystems: "*The hydrology of the landscape influences and changes the physiochemical environment, which in turn, along with hydrology, determines the biotic communities that are found in the wetland*." For example, hydrological processes, such as seasonal soil–water regimes, surface inundation and maintenance of water quality, critically determine wetland ecosystem structure and function, and thus influence the type ecosystem goods and services provided. Similarly, changes in water regime will affect different wetland services significantly, resulting in many possible trade-offs and synergies among these services within different wetland scenarios and water regimes. The consequence is that the ecosystem services provided by wetlands are driven by hydrology, and understanding how changes in hydrological processes affect the delivery of these services is critical to determining the impact on human welfare (Bullock and Acreman 2003, Brauman *et al*. 2007, Emerton and Boss 2008, Mitsch *et al*. 2009).

Because the structure and functions of many wetlands can be uniquely defined by hydrological processes, it is possible to identify the spatial unit, or natural landscape, that is distinct to each type of wetland. In particular, different aspects of the hydrological system underlying wetlands and their services operate at different scales, e.g. surface inundation (flooding), water quality and biodiversity. Thus, as a wetland landscape varies in scale, due perhaps to conversion, draining or other human-induced disturbances, the impact on the provision of and synergies between wetland services can be substantial. Such a landscape approach is being used increasingly for assessing the cumulative effects of wetland loss and degradation, characterizing wetland boundaries and identifying restoration or mitigation opportunities (NRC 1995, Bedford 1996, 1999, Gwin *et al*. 1999, Mitsch and Gosselink 2000, Simenstad *et al*. 2006). It follows that the various goods and services provided by a wetland will also be tied to, and thus defined by, its landscape extent; i.e. "*wetland values depend on the hydrogeomorphic location in which they are found*" (Mitsch and Gosselink 2000, p. 27).

If the hydrology-related services of wetlands are related to their landscape extent, then characterizing wetland ecosystems as natural assets is straightforward. In other words, as there are "*reciprocal interactions between spatial pattern and ecological processes*" (Turner 2005, p. 319), it is the spatially heterogeneous area of a wetland landscape that is the fundamental to its ability to provide the various wetland ecosystem services listed in Table 1. It follows that, if for each wetland ecosystem we can define its corresponding landscape in terms of a quantifiable "land unit", which is defined as "a tract of land that is ecologically homogeneous at the scale level concerned" (Zonneveld 1989, p. 68), then we have a representation of the wetland ecosystem as a natural asset in the form of this unit of land, or ecological landscape.

WETLAND ECOSYSTEMS AS NATURAL ASSETS

For example, let us suppose that the flow of services provided by a wetland ecosystem in any time period, t, can be quantified, and that we can measure what each individual is willing to pay for having these services provided to him or her. If we sum up, or aggregate, the willingness to pay by all the individuals benefiting in each period from the wetland services, we will have a monetary amount (call it B_t) that indicates the social benefits in the given time period t of those services. It is assumed that there will be a stream of such benefits generated by the ecosystem, from the present time and into the future. Because society is making a decision today about whether or not to preserve the wetland, we want to consider the flow of benefits of these services, net of the costs of maintaining the ecosystem, in terms of its present value. To do this, any future net benefit flows are discounted into present value equivalents. In essence, we are treating the wetland as a special type of capital asset, a kind of "natural wealth", which, just like any other asset or investment in an economy, is capable of generating a current and future flow of income or benefits.

Compared to conventional economic or financial assets, the wetland asset has some special characteristics.

For one, like all ecosystems, a wetland is a special type of "non-reproducible capital good". If the wetland ecosystem is left relatively undisturbed, then its flow of services is not affected by the rate at which they are used. The wetland can go on providing the various services listed in Table 1. Although, like other assets in the economy, a wetland ecosystem can be increased by investment, e.g. through restoration activities, ecosystems can also be depleted or degraded, e.g. through habitat destruction, land conversion, pollution impacts, etc. Land-use change, in particular, poses the greatest danger to wetland landscapes, and thus the services they provide. As summarized by Bockstael (1996, p. 1169), "*because landscape pattern and ecological processes are closely linked . . . land use change at one scale or another is perhaps the single greatest factor affecting ecological resources*."

The problem is exacerbated by the fact that most wetland services are not marketed, whereas the services of most other assets in the economy, including land appropriated through converting wetland landscape, are marketed. The failure to consider the values provided by key wetland ecosystem services in current policy and management decisions is a major reason for the disappearance of many global wetlands (MEA 2005). The failure to explicitly measure the aggregate willingness to pay for otherwise non-marketed ecological services exacerbates these problems, as the benefits of these services are underpriced and may lead to excessive land conversion, habitat fragmentation, harvesting and pollution caused by commercial economic activity undertaken by humans.

Figure 1 illustrates the difficulty that the above challenges pose for managing a wetland ecosystem landscape among competing uses. In this figure, the marginal social benefits of ecological services at any time t are represented by the line MB_t for a coastal ecosystem of given area $\bar{A}$. For the purposes of illustration, this line is assumed to be downward sloping, which implies that, for every additional square kilometre of wetland landscape area, A, preserved in its original state, more ecosystem service benefits will be generated, but at a decreasing amount. Note that it is straightforward to determine the aggregate willingness to pay for the benefits of these services, B_t, from this line; it is simply the area under the MB_t line. If there is no other use for the wetland, then the opportunity costs of maintaining it are zero, and B_t is at its maximum size when the entire wetland ecosystem is maintained at its original land area size $\bar{A}$. The ecosystem management decision is therefore simple; the wetland landscape should be completely preserved and allowed to provide its full flow of services in perpetuity.

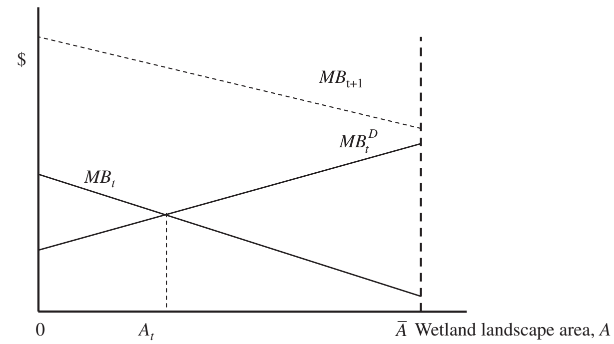

Fig. 1 Wetland landscape conversion to development.

However, population and economic development pressures in many areas of the world usually mean that the opportunity cost of maintaining wetland landscape is not zero. The ecosystem management decision needs to consider these alternative development uses of wetland landscape, which should be included in Fig. 1. For example, suppose that the marginal social benefits of converting the wetland for these development options is now represented by a new line MB_t^D in the figure. The result is that $\bar{A} - A_t$ of wetland landscape should be converted for development, leaving A_t of the original ecosystem undisturbed.

Both of the outcomes discussed so far assume that the willingness to pay for the marginal benefits arising from wetland ecosystem services, MB_t, is explicitly measured, or valued. But if this is not the case, then these non-marketed flows are likely to be ignored in the land-use decision. Only the marginal benefits MB_t^D of the marketed outputs arising from wetland economic development activities will be taken into account, and, as indicated in the figure, this implies that the entire ecosystem area $\bar{A}$ will be converted for development.

A further problem is the uncertainty over the future values of wetland landscape. It is possible, for example, that the benefits of ecosystem services are larger in the future as more scientific information becomes available over time. For example, suppose that in the subsequent period $t+1$ it is discovered that the value of wetland ecosystem services is actually much larger, so that the marginal benefits of these services, MB_{t+1}, in present value terms is now represented by the dotted line in Fig. 1. If the present value marginal benefits from wetland development in the future are largely unchanged, i.e. $MB_t^D \approx M_{t+1}^D$, then, as the figure indicates, the future benefits of ecosystem services exceed these costs, and the natural landscape should be restored to its original area $\bar{A}$, assuming of course that it is technically feasible and not excessively expensive to do so. Unfortunately, in making development decisions today we often do not know that, in the future, the value of ecosystem services will turn out to exceed development benefits. Our simple example shows that, if we have already made the decision today to convert the $\bar{A} - A_t$ area of the wetland, then in the future we should restore the original wetland ecosystem.

The following two case studies, a floodplain in northern Nigeria and mangroves in southern Thailand, illustrate the approach to viewing wetlands as natural assets that is highlighted by Fig. 1. In the case of Nigeria, the natural capital is the river floodplain, and the trade-off between developing and conserving the wetland is the upstream water diversion compared to the downstream flooding benefits to farming, fishery and forestry, as well as groundwater recharge. In the case of Thailand, the natural capital is the mangrove system, and the trade-off is the conversion to shrimp farms as opposed to the mangrove benefits of locally harvested products, habitat-fishery linkages, and storm protection.

CASE STUDY: HADEJIA-JAMA'ARE FLOODPLAIN, NIGERIA

In northeast Nigeria, an extensive floodplain has been created where the Hadejia and Jama'are rivers converge to form the Komadugu Yobe River, which drains into Lake Chad. Although referred to as wetlands, and designated as a Ramsar site, much of the Hadejia-Jama'are floodplain is dry for some or all of the year. Nevertheless, the floodplain provides essential income and nutritional benefits in the form of agriculture, grazing resources, non-timber forest products, fuel wood and fishing for local populations, as well as groundwater recharge of the Chad Formation aquifer and many shallow aquifers throughout the region, and "insurance" resources in times of drought (Hollis *et al*. 1993, Thompson and Hollis 1995, Thomas and Adams 2000). In addition, the wetlands are a unique migratory habitat for many wildfowl and wader species from Palaearctic regions, and contain a number of forestry reserves (Hollis *et al*. 1993, Lemly *et al*. 2000, Thompson and Polet 2000).

However, the Hadejia-Jama'are floodplain has come under increasing pressure from drought and upstream water developments. Due to past water diversion, the maximum extent of flooding declined from between 250 000 and 300 000 ha in the 1960s and 1970s to around 70 000–100 000 ha in the 1990s (Thompson and Hollis 1995, Thompson and Polet 2000, Jacobs 2002), which is about the present level of flooding. Drought is a persistent, stochastic environmental problem facing all sub-Saharan arid and semi-arid zones, and the main cause of unexpected reductions in flooding in drought years. But the main long-term threat to the floodplain is water diversion through large-scale water projects on the Hadejia and Jama'are rivers. Upstream developments are affecting incoming water, either through dams altering the timing and size of flood flows, or through diverting surface or groundwater for irrigation. These developments have been taking place without consideration of their impacts on the Hadejia-Jama'are floodplain, or any subsequent loss of economic benefits that are currently provided by use of the floodplain.

In addition, the diminishing floodplain is also worsening downstream conflicts and ethnic tensions over water use, such as between sedentary agricultural communities and nomadic herders (Lemly *et al*. 2000, Thompson and Polet 2000, Schuyt 2005). In addition, high-quality floodplain land has been appropriated for pump-irrigated wheat production by local military garrisons. Changing agricultural practices and agricultural intensification has also led to the removal of habitat for wildlife and natural vegetation. As noted by Schuyt (2005, p. 185), "*the poor flooding of wetland due to dams, diversions and climatic changes*" is not only affecting floodplain agriculture, fishing and fuel wood, but also a wide range of other wild resources that "*provide materials for utensils and construction and contribute to improved diets and health, food security, income generation and genetic experimentation*."

The largest upstream irrigation scheme at present is the Kano River Irrigation Project (KRIP), which currently amounts to 22 000 ha (Sangari 2006). Water supplies for the project are provided by Tiga Dam, the biggest dam in the basin, which was completed in 1974. Water is also released from this dam to supply Kano City. The second major irrigation scheme within the river basin is the Hadejia Valley Irrigation Project (HVIP), which irrigates 8000 ha (Balmisse *et al*. 2003). The HVIP is supplied by Challawa Gorge Dam on the Challawa River, upstream of Kano, which was finished in 1992. Challawa Gorge also provides water for Kano City water supply. A number of small dams and associated irrigation schemes have also been constructed or are planned for minor tributaries of the Hadejia River. In comparison, the Jama'are River is relatively uncontrolled, with only one small dam across one of its tributaries. However, plans for a major dam on the Jama'are at Kafin Zaki have been in existence for many years, and would provide water for an irrigated area totalling 84 000 ha. Work on Kafin Zaki Dam has been started and then stopped a number of times. In 2008, the Bauchi State government announced plans to proceed again with construction of the dam, although work has yet to start.

The current and planned water diversions in the Hadejia-Jama'are River basin are, unfortunately, an example of the classic case of ignoring the benefits provided by a natural asset—in this case the downstream floodplain landscape. As shown in Fig. 1, against the benefits of these upstream water developments must be weighed the opportunity cost of the downstream floodplain losses. Otherwise, too much water is diverted upstream, and the floodplain landscape will diminish excessively.

For example, economic valuation studies have focused on three types of floodplain benefits that are affected by the impacts of upstream water diversion on the floodplain:

- Flood-recession agriculture, fuel wood and fishing in the floodplain (Barbier *et al*. 1993).
- Groundwater recharge of domestic water supply for household use (Acharya and Barbier 2002).
- Groundwater recharge that supports dry season irrigated agricultural production (Acharya and Barbier 2000).

Barbier and Thompson (1998) simulated the impacts of various upstream water diversion scenarios in the Hadejia-Jama'are River basin on the flood extent that determines the downstream floodplain area. The economic gains of the upstream water projects were then compared to the resulting economic losses to downstream agricultural, fuel wood and fishing benefits. All scenarios were compared to a baseline simulation without any of the large-scale water resource schemes in place within the river basin.

Table 2 summarizes the estimated gains in irrigation benefits upstream with the downstream losses from agricultural, fuel wood and fish production in the floodplain for the different upstream dam and water release scenarios. Given the high productivity of the floodplain, the losses in economic benefits due to changes in flood extent for all scenarios are large, ranging from US$4 to 23 million. As expected, there is a direct trade-off between increasing irrigation and dam developments upstream and impacts on

Table 2 Losses in floodplain benefits *versus* gains in irrigated production, net present value (US$ 1989/90 prices), Nigeria (source: Barbier and Thompson 1998).

Scenarios[c]	Irrigation value[a]	Floodplain loss[b]	Net loss	Irr. value as % of floodplain losses
	(1)	(2)	(2) – (1)	(1)/(2) × 100
Scenario 1	682 983	−4 045 024	−3 362 041	16.88
Scenario 2	354 139	−2 558 051	−2 203 912	13.84
Scenario 3	682 963	−7 117 291	−6 434 328	9.60
Scenario 4	3 124 015	−23 377 302	−20 253 287	13.36
Scenario 5	556 505	−15 432 952	−14 876 447	3.61

[a] Based on the mean of the net present values of per ha production benefits for the Kano River Irrigation Project (KRIP), and applied to the gains in total irrigation area for each scenario.

[b] Based on the mean of the net present values of total agricultural, fuel wood and fishing benefits for the Hadejia-Jama'are floodplain, averaged over the actual peak flood extent for the wetlands of 112 817 ha in 1989/90 and applied to the declines in mean peak flood extent associated with each scenario.

[c] Scenario 1: Tiga Dam only. KRIP at 27 000 ha; no regulated water releases.
Scenario 2: Tiga Dam only. KRIP at 14 000 ha; regulated water release of 400×10^6 m^3 in August.
Scenario 3: Tiga Dam, Challawa Gorge, and small dams on Hadejia tributaries. KRIP at 27 000 ha; regulated water release from Challawa Gorge of 348×10^6 m^3 $year^{-1}$.
Scenario 4: Tiga Dam, Challawa Gorge, small dams on Hadejia tributaries, Hadejia Valley Irrigation Project (HVIP), Kafin Zaki Dam. KRIP at 27 000 ha; 84 000 ha of irrigated agriculture from Kafin Zaki; HVIP at 12 500 ha; regulated water release from Challawa Gorge of 348×10^6 m^3 $year^{-1}$.
Scenario 5: Tiga Dam, Challawa Gorge, small dams on Hadejia tributaries, HVIP, Kafin Zaki Dam. KRIP at 14 000 ha; HVIP at 8000 ha; regulated water releases of 350×10^6 m^3 in August from Tiga Dam, 348×10^6 m^3 $year^{-1}$ and 100×10^6 m^3 in July from Challawa Gorge, 100×10^6 m^3 $month^{-1}$ in October–March and 500×10^6 m^3 in August from Kafin Zaki, and Hadejia Barrage open in August.

the wetland benefits downstream. Scenario 2, which yields the lowest upstream irrigation gains, also has the least impact in terms of floodplain losses, whereas Scenario 4 has both the highest irrigation gains and floodplain losses.

Although Scenario 2 is the preferred outcome, as it produces the lowest net loss overall, it is clearly unrealistic. Challawa Gorge was completed in 1992, small dams have been built on the Hadejia's tributaries, and the HVIP has been implemented. In fact, Scenario 4 is already on the way to being implemented, although when the construction of Kafin Zaki Dam might occur is presently uncertain. The only alternative to Scenario 4, which assumes full implementation of all upstream water projects and dams without any releases for the downstream floodplain, is Scenario 5, which also assumes full upstream development, but with less irrigation to allow regulated water releases from the dams to sustain inundation of the downstream floodplain.

The results confirm that, in all the scenarios simulated, the additional value of production from large-scale irrigation schemes does not replace the lost production attributed to the wetlands downstream. Gains in irrigation values account for, at most, 17% of the losses in floodplain benefits. Interestingly, even in Scenario 4 that allows for full development of all planned upstream dam and irrigation projects, the losses to floodplain agriculture, fishing and fuel wood benefits are so large that the additional irrigation values gained compensate for only 13% of the losses. Further expansion of the KRIP and HVIP, as well as the construction of Kafin Zaki Dam and additional upstream irrigation schemes, are not appropriate developments in the river basin. Further upstream water diversion for irrigation is also questionable, given the serious concerns about the inefficient use of water for crop production by farmers in the KRIP and HVIP (Balmisse *et al*. 2003, Sangari 2006). However, as an alternative, if Kafin Zaki dam were to be constructed and formal irrigation within the basin limited to its current extent, the introduction of a regulated flooding regime (Scenario 5) would reduce the net losses from around US$20 million to just under US$15 million. Scenario 5 may therefore be the most efficient outcome for allocating water between the floodplain and upstream dam developments.

Such a regulated flooding regime could also produce additional economic benefits that are not captured in our analysis. Greater certainty over the timing and magnitude of the floods may enable farmers to adjust to the resulting reduction in the risks normally associated with floodplain farming. Enhanced dry season flows provided by the releases from Challawa Gorge and Kafin Zaki dams in Scenario 5 would also benefit farmers along the Hadejia and Jama'are rivers while the floodplain's fisheries may also experience beneficial impacts from the greater extent of inundation remaining throughout the dry season (Neiland *et al*. 2005). Thus, the introduction of a regulated flooding regime in conjunction with upstream water developments may be the only realistic hope of minimizing floodplain losses.

Some of the upstream water developments are being used or have the potential to supply water to Kano City. Although these releases were included in the hydrological simulations by Barbier and Thompson (1998), the economic analysis was unable to calculate the benefits to Kano City of these water supplies. However, the hydrological analysis shows that the proposed regulated water release from Tiga Dam to reduce downstream floodplain losses would not affect the ability of Tiga Dam to supply water to Kano. Although the potential exists for Challawa Gorge to supply additional water to Kano, it is unclear how much water could be used for this purpose. The resulting economic benefits are unlikely to be large enough to compensate for the substantial floodplain losses incurred by the gorge and the additional upstream developments in the Hadejia Valley. Currently, there are no plans for Kafin Zaki dam to be used to supply water to Kano.

In addition, Barbier and Thompson (1999) were unable to calculate other important floodplain benefits, such as the role of the wetlands in supporting pastoral grazing and in recharging groundwater both within the floodplain and in surrounding areas.

For example, one of the concerns is that disruptions to flood extent will affect the annual recharge of the underlying aquifers, which will in turn impact the welfare of local populations dependent on this groundwater for drinking water and other household uses (Thompson and Hollis 1995, Thompson and Goes 1997). In a separate study, Acharya and Barbier (2002) estimate the value placed on groundwater, either purchased or collected from village wells, by households in the floodplain region. Approximately 108 000 households in the region depend on groundwater that is recharged by the wetlands. Three villages in the Madachi region of the Hadejia-Jama'are floodplain and one village in the Sugum region were chosen for the economic valuation study, based on the hydrological evidence that the villages in these areas rely on groundwater recharged mainly by wetlands (Thompson and Goes 1997). The flooding in Madachi is caused by the floodwaters of the Hadejia River. The Sugum region is located in the eastern part of the wetlands and is influenced by the flooding of the Jama'are River.

The results of the analysis by Acharya and Barbier (2002) suggest that the value of the recharge function is US$13 209 per day for the floodplain. The average income-equivalent loss to households for a 1 metre drop in groundwater levels is approximately US$0.12 per household per day. This average suggests a daily loss of approximately 0.23% of monthly income for households that purchase their water, 0.4% of monthly income for households that collect their water and 0.14% of monthly income for households that do both.

If upstream water diversion is causing less flooding and standing water downstream, then the resulting reduction in groundwater recharge could have important implications for dry season irrigated agricultural production downstream (Thompson and Hollis 1995, Thompson and Goes 1997). Acharya and Barbier (2000) also conducted an economic analysis of the impact of a decline in groundwater levels on dry season vegetable and wheat irrigated agricultural production in the floodplain region. They surveyed a sample of 37 farms in the Madachi area, out of a total 309 dry season farmers on 6600 ha of cropland irrigated through tubewell abstraction from shallow aquifers. Wheat, tomato, onions, spring onions, sweet potatoes and pepper are the main cash crops grown by the farmers, although okra and eggplant are more minor crops grown principally for home consumption. On average, irrigated dry season agriculture in the Madachi area is worth US$412.5 per ha, with a total estimated annual value of US$2.72 million over the entire 6600 ha.

Acharya and Barbier (2000) estimate that a fall in groundwater levels from 6 to 7 m depth led to additional costs of pumping water and less use of water inputs. The result is losses of US$32.5 per vegetable farmer, approximately 7.65% of yearly income, and US$331 for vegetable and wheat farmers, or around 77% of annual income. The total loss associated with the 1 m change in groundwater was estimated to be US$62 249 for all 6600 ha of dryland farming in the Madachi area. As shallow aquifers could irrigate 19 000 ha within the floodplain region, a total loss of US$1.18 million was estimated for the entire wetlands.

To summarize, the northern Nigerian case illustrates how valuing the downstream benefits of various floodplain services is critical to the decision as to whether or not to divert water upstream. The diversion of water upstream for irrigated agriculture and, more recently, urban water supply, is drastically affecting the extent of flood inundation downstream. The economic losses due to diminishing floodplain landscape are highly significant, and include impacts on flood-recession agriculture, fishing and fuel wood, and on groundwater recharge of domestic water supplies and irrigation for agriculture. Although it is too late to stop some of the upstream water projects and dams, halting planned developments and introducing regulated water releases and flooding regimes for existing dams may be the only realistic hope of minimizing floodplain losses and conflicts in the rapidly degrading wetlands.

CASE STUDY: MANGROVE LAND USE, THAILAND

In Thailand, aquaculture expansion has been associated with mangrove wetlands destruction. Since 1961 Thailand has lost from 1500 to 2000 km^2 of coastal mangroves, or about 50–60% of the original area (Wilkie and Fortuna 2003). Over 1975–1996, 50–65% of Thailand's mangroves was lost to shrimp farm conversion alone (Aksornkoae and Tokrisna 2004).

Mangrove deforestation in Thailand has focused attention on the two principal services provided by mangrove ecosystems: their role as nursery and breeding habitats for off-shore fisheries, and their role as natural "storm barriers" to periodic coastal storm events, such as wind storms, tsunamis, storm surges and typhoons. In addition, many coastal communities exploit mangroves directly for a variety of products, such as fuel wood, timber, raw materials, honey and resins, and crabs and shellfish. Various studies have suggested that these benefits of mangroves are significant in Thailand (Sathirathai and Barbier 2001, Barbier 2003, 2007).

Therefore, valuation of the ecosystem services provided by mangroves is important for two land-use policy decisions in Thailand. First, although declining in recent years, conversion of mangroves to shrimp farm ponds and other commercial coastal developments continues to be a major threat to Thailand's remaining mangrove areas. Second, since the December 2004 Tsunami Disaster, there is now considerable interest in rehabilitating and restoring mangrove ecosystems as "natural barriers" to future coastal storm events. Thus valuing the goods and services of mangrove ecosystems can help to address two important policy questions: Do the net economic returns to shrimp farming justify further mangrove conversion to this economic activity? and Is it worth investing in mangrove replanting and ecosystem rehabilitation in abandoned shrimp farm areas?

To illustrate how the improved and more accurate valuation of ecosystems can help inform these two policy decisions, Table 3 compares the per hectare net returns to shrimp farming, the costs of mangrove rehabilitation, and the value of mangrove services. All land uses are implemented from 1996 to 2004, and are valued in 1996 US$ per hectare (ha).

Several analyses have demonstrated that the overall commercial profitability of shrimp aquaculture in Thailand provides a substantial incentive for private landowners to invest in such operations (Tokrisna 1998, Sathirathai and Barbier 2001, Barbier 2003). However, many of the conventional inputs used in shrimp pond operations are subsidized below border-equivalent prices, thus artificially increasing the private returns to shrimp farming. In Table 3 the net economic returns to shrimp farming, which are calculated once the estimated subsidies are removed, are based on non-declining yields over a five-year period of investment (Sathirathai and Barbier 2001). After this period, there tend to be problems of drastic yield decline and disease; shrimp farmers then usually abandon their ponds and find a new location. In Table 3 the annual economic returns to shrimp aquaculture are estimated to be US$322 per ha, and, when discounted over the five-year period at a rate of 10–15%, yield a net present value of US$1078–1220 per ha.

There is also the problem of the highly degraded state of abandoned shrimp ponds after the five-year period of their productive life. Across Thailand, those areas with abandoned shrimp ponds degenerate rapidly into wasteland, since the soil becomes very acidic, compacted and too poor in quality to be used for any other productive use, such as agriculture. Rehabilitation of the abandoned shrimp farm sites requires treatment and detoxification of the soil, replanting mangrove forests and maintenance and protection of mangrove seedlings for several years. As shown in Table 3, these restoration costs are considerable: US$8812–9318 per ha in net present value terms. This reflects the fact that converting mangroves to establish shrimp farms is almost an "irreversible" land use, and, without considerable additional investment in restoration, these areas do not regenerate into mangrove forests. As the restoration costs exceed the net economic returns per ha, the decision should have been to prevent the shrimp aquaculture operation from occurring in the first place.

Unfortunately, past land-use policy in Thailand has ignored the opportunity costs of shrimp farming in terms of foregone mangrove services, and, as a result, extensive coastal areas have been deforested of mangroves. Many short-lived shrimp farms in these areas have also long since fallen unproductive and are now abandoned. Thus, an important issue today is whether it is worth restoring mangroves in these abandoned areas. If the foregone benefits of the ecological services of mangroves are not large, then mangrove restoration may not be a reasonable option. Therefore, Table 3 indicates the value of three of these benefits: the net income from local mangrove forest products, habitat-fishery linkages and storm protection.

Sathirathai and Barbier (2001) estimate the value to local communities of using mangrove resources in terms of the net income generated from the forests by means of various wood and non-wood products. If the extracted products were sold, market prices were used to calculate the net income generated (gross income

Table 3 Comparison of land use values per ha, Thailand, 1996–2004 (US$).

Land use	Net present value per ha (10–15% discount rate)
Shrimp farming	
Net economic returns[a]	1078–1220
Mangrove ecosystem rehabilitation	
Total cost[b]	8812–9318
Ecosystem goods & services	
Net income from collected forest products[c]	484–584
Habitat-fishery linkage[d]	708–987
Storm protection service[e]	8966–10 821
Total	10 158–12 392

[a]Based on annual net average economic returns US$322 per ha for five years from Sathirathai and Barbier (2001), updated to 1996 US$.

[b]Based on costs of rehabilitating abandoned shrimp farm site, replanting mangrove forests and maintaining and protecting mangrove seedlings. From Sathirathai and Barbier (2001), updated to 1996 US$.

[c]Based on annual average value of US$101 per ha over 1996–2004 from Sathirathai and Barbier (2001), updated to 1996 US$.

[d]Based on a dynamic analysis of mangrove-fishery linkages over 1996–2004 from and assuming the estimated Thailand deforestation rate of 3.44 km^2 $year^{-1}$ (see Barbier 2007).

[e]Based on marginal value per ha of expected damage function approach of Barbier (2007).

minus the cost of extraction). If the products were used only for subsistence, the gross income was estimated based on surrogate prices, i.e. the market prices of the closest substitute. Based on surveys of local villagers in Surat Thani Province, the major products collected by the households were various fishery products, honey, and wood for fishing gear and fuel wood. As shown in Table 3, the net annual income from these products is US$101 per ha, or a net present value of US$484–584 per ha.

The coastal habitat-fishery of mangroves in Thailand may also be modelled through incorporating the change in wetland area within a multi-period harvesting model of the fishery (Barbier 2007). The key to this approach is to model a coastal wetland that serves as a breeding and nursery habitat for fisheries as affecting the growth function of the fish stock. As a result, the value of a change in this habitat-support function is determined in terms of the impact of any change in mangrove area on the returns earned from the fishery over many harvesting seasons. As Table 3 indicates, the net present value of this service ranges from US$708 to 987 per ha.

The value of the coastal protection service of mangroves in Table 3 is derived by employing the expected damage function (EDF) valuation methodology for estimating the expected damage costs avoided through increased provision of the storm protection service of coastal wetlands (Barbier 2007). By applying this EDF approach, Table 3 estimates the benefits from the storm protection service of mangroves in Thailand to be US$1879 per ha, or US$8966–10 821 per ha in net present value terms.

Table 3 indicates that the net present value of all three mangrove ecosystem benefits ranges from US$10 158 to 12 392 per ha. These ecosystem service values clearly exceed the net economic returns to shrimp farming. In fact, the net income to local coastal communities from collected forest products and the value of habitat-fishery linkages total US$1192–1571 per ha, which is greater than the net economic returns to shrimp farming. However, the value of the storm protection is critical to the decision as to whether or not to replant and rehabilitate mangrove ecosystems in abandoned pond areas. As shown in Table 3, the storm-protection benefit makes mangrove restoration an economically feasible land-use option, as the net present value of all three mangrove benefits exceeds even the high costs of restoration

To summarize, this case study has shown the importance of valuing the ecological services in wetland conversion and restoration decisions, as outlined in Fig. 1. The irreversible conversion of mangroves for aquaculture results in the loss of ecological services that generate significantly large economic benefits. This loss of benefits should be taken into account in land-use decisions that lead to the widespread conversion of mangroves, but typically are ignored in private sector calculations. The high restoration costs also reflect the fact that "reversing" mangrove conversion is difficult, and should not always be considered *ex post*. Instead, before the decision to allow shrimp farming to take place, the restoration costs should be considered as part of the decision as to whether or not it is worthwhile to irreversibly convert mangroves.

FINAL REMARKS

Viewing wetlands as natural assets is an important way of communicating to policymakers the economic importance of these valuable ecosystems. As they provide a flow of beneficial goods and services over time, wetland ecosystems should be considered no different from any other form of wealth in an economy. Policymakers are then confronted with a clear choice of deciding whether or not to conserve this form of natural wealth or to convert it to another form of wealth. As illustrated by Fig. 1, if the hydrology-related services of wetlands are related to their landscape extent, then characterizing wetland ecosystems as natural assets and analysing the trade-off between conservation and conversion are straightforward decisions.

The case study examples of floodplain loss in Nigeria and mangrove land use in Thailand further illustrate the importance of this approach. In both cases, the key to managing the wetland asset is in assessing its hydrology-related goods and services. In the case of Nigeria, the trade-off between developing and conserving the floodplain is determined by comparing the gains from upstream water diversion to the downstream flooding benefits from farming, fishery and forestry, as well as groundwater recharge. In the case of Thailand, the trade-off is the gains from mangrove conversion to shrimp farms compared to the mangrove benefits of locally harvested products, habitat-fishery linkages and storm protection.

As these case studies illustrate, to estimate wetland benefits requires understanding the underlying hydrological and ecological relationships. Unfortunately, as Table 1 indicates, for many key hydrology-related wetland services, much work needs to done in improving our knowledge of how the structure and functions of wetland ecosystems generate these services and how to value their contribution to human welfare. Although the number of wetland valuation studies has increased in recent years, very few or no valuation studies exist for some important wetland ecosystem services.

However, the concept of treating wetlands as natural assets is gaining acceptance, as reflected in the increasing number of studies that are evaluating different conservation *versus* development scenarios. Various examples include: balancing agricultural conversion with riverine wetland conservation in South Africa (Jogo and Hassan 2010); managing environmental change in the Norfolk and Suffolk Broads of the UK (Turner *et al*. 2004); comparing flood control regimes to natural floodplain production in Bangladesh (Islam and Braden 2006); examining rural land-use changes and floodplain management scenarios in the UK (Posthumus *et al*. 2010); valuing changes in ecosystem services from various wetland management regimes in Greece (Birol *et al*. 2006); evaluating preferences for alternative restoration options for the Greater Everglades ecosystem in the USA (Milon and Scrogin 2006); valuing ecosystem services from wetlands restoration in the Mississippi Valley, USA (Jenkins *et al*. 2010); and assessing different mangrove management options in Malaysia (Othman *et al*. 2004).

As policymakers continue to consider conservation, development and restoration wetland options, characterizing wetlands as natural assets and estimating their multiple hydrology-related goods and services will serve as a valuable analytical approach for assessing these options. Increasingly, a hydrology-based landscape approach is being used to assess the cumulative effects of wetland loss and degradation, characterizing wetland boundaries and identifying restoration or mitigation opportunities (Bedford 1996, 1999, Gwin *et al*. 1999, Mitsch and Gosselink 2000, NRC 1995, Simenstad *et al*. 2006). As emphasized in this paper, such an approach is consistent with the view of wetlands as natural assets that generate multiple ecosystem services. What is urgently needed is more inter-disciplinary research collaboration on how the hydrological processes underlying wetlands generate these services and how to value their contribution to human welfare.

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