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Estimation of the Water Quality Amelioration Value of Wetlands

A Case Study of the Western Cape, South Africa

Jane Turpie, Elizabeth Day, Vere Ross-Gillespie, and Anton Louw

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Estimation of the Water Quality Amelioration Value of Wetlands: A Study of the Western Cape, South Africa

Jane Turpie, Elizabeth Day, Vere Ross-Gillespie, and Anton Louw

Abstract

Wetlands are commonly understood to have the capacity to reduce the loads of excess nutrients, pathogens, sediments, and other contaminants generated by various activities in their catchment areas. However, quantifying these "services" is difficult and most research in this field has concentrated on artificial treatment wetlands. Understanding the value of their water treatment characteristics, as well as the other services they provide, is increasingly recognized as essential to achieving a balance between conservation and activities that degrade or replace wetlands.

The aim of this study is to estimate the water treatment capacity of wetlands on a landscape scale in the South Western Cape of South Africa and estimate the economic value of the service performed. We collected samples at the outflow points of 100 subcatchment areas and measured the loads of nitrogen, dissolved phosphorus, and suspended solids, which were analyzed with respect to detailed spatial data on land cover and wetlands area. Wetlands play a significant role in the reduction of nitrates, nitrites, and ammonium, but not dissolved phosphorus or suspended solids. Estimated removal rates range from 307 to 9,505 kg N per ha^{-1} year^{-1}, with an average of 1,594 ± 1,375 kg N per ha^{-1} year^{-1}. Data from a number of water treatment works suggest that the cost of removal of ammonium nitrogen is in the order of ZAR 26 per kilogram. Applied to the wetlands in the study area—assuming wetlands do play a role in total phosphorus removal—this suggests that the average value of the water treatment service provided by wetlands in the study area is about ZAR 14,350 ± 12,385 ha^{-1} year^{-1}. These values are high enough to compete with the alternative land uses that threaten their existence. The results suggest that wetlands should be given considerably more attention in land-use planning and regulation.

Key Words: ecosystem services, wetlands, economic valuation, water treatment

JEL Classification: Q57

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Introduction

Wetlands are among the most threatened habitats globally, and it is estimated that since 1900 more than half of the world's wetlands have been destroyed and lost to other land uses (Barbier 1993). Indeed, despite various forms of international and national legislation ratifying their protection (Bergstrom and Stoll 1993), wetlands continue to be affected by human activities, including channelization, drainage, crop production, effluent disposal, and water abstraction, including in South Africa (Walmsley 1991; Barbier et al. 1997; Turner et al. 2000; Bowers 1983).

A major factor contributing to this international trend of destroying wetlands is the fact that their value is poorly understood. In addition to providing habitat to rare or endangered plants and animals, wetlands host a range of "ecosystem services," which endow surrounding and downstream communities with direct and indirect benefits (Barbier et al. 1997). These include services or goods, such as reeds and fish; regulating services (e.g., the attenuation of floods; treatment of water quality by the sequestration or uptake of pollutants, including nutrients and heavy metals; and effective trapping of suspended sediments); and cultural services, such as opportunities for recreation, scientific research, and spiritual fulfillment (ibid.). The economic benefits and services provided by wetland ecosystems such as these are frequently overlooked by governments, developers, private industry, and other land users (Emerton 1998), resulting at times in distorted decisions. Estimation of the economic value of wetlands is thus seen as a

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potentially significant means of correcting these distortions and achieving a better balance between conservation and activities that degrade or replace wetlands.

The global value of wetlands and their associated ecosystem services has been estimated in the past as US$ 14 trillion annually (Costanza et al. 1997). However, the estimation of wetlands values on a local scale requires a more accurate understanding of their capacity to deliver services and the demand for those services. While it is relatively straightforward to quantify harvests of natural resources from wetlands and the value of tourism with survey-based valuation techniques, valuing the regulating services of wetlands is particularly challenging because it requires in-depth understanding of complex, interconnected biophysical processes. The biophysical functioning of different types of wetlands in different ecoregions varies dramatically and may be difficult to measure and conceptualize. Past valuation studies have been hampered by a lack of information, particularly regarding the ability of wetlands to ameliorate the quality of water passing into systems downstream.

The main water-quality constituents that wetlands influence include the loading and/or concentrations of phosphorus and nitrogen nutrients, ammonia, and various heavy metals, as well as suspended solids and their load of sorbed compounds. As streamflows (flow of water in streams, rivers, etc., from land to ocean) enter wetlands, they slow down, with the result that suspended sediments settle out of the water column. Because many pollutants (e.g., metals and organic chemicals) attach strongly to suspended matter, this process is also important for reducing these materials in downstream systems.

While uptake by plants and epiphytes, and sorption to soil surfaces, are primary processes that change phosphorus concentrations in wetlands water in the short term, plants and their epiphyton release up to 75% of this phosphorus back into the water column, and long-term storage (accumulation) relies primarily on sediment and peat accumulation (Kadlec and Knight 1996; Cooke et al. 2005). Removal of heavy metals occurs by short-term uptake into plant structures, but longer-term storage is achieved in sedimentation.

Wetlands are also effective in processing nitrates (Cooke et al. 2005). They have the capacity to remove various pathogens from water passing through. Although this is true of many wetlands when pathogens (e.g., coliform bacteria) are present in high loads, it should be noted that wetlands themselves include active populations of many bacteria. Wetlands with large populations of birds or other wildlife may well contribute more fecal bacteria to through-flowing water than they remove (Kadlec and Knight 1996).

Seasonality is also important. Wetlands are thought to be better at removing total suspended solids, phosphorus, and ammonia during high flow periods (when sediment loads entering the wetland increase), but they can also remove nitrates during low flow periods (Johnston et al. 1990). During extreme flow events, the sediments and nutrients that have accumulated in wetlands may be flushed out, temporarily elevating downstream loads. This may have a smaller effect on the downstream environment than if they were released during lower flow periods. However, where downstream systems have areas of permanent sediment entrapment (e.g., basins and lakes), this same net loading may occur with a large, dilute load of sediments or a smaller but continual supply of sediment.

A number of studies have researched the function of wetlands in the treatment of waste water (e.g., Peltier et al. 2003; Thullen et al. 2005; Batty et al. 2005), but most have been carried out in artificial or dedicated treatment wetlands, and few have used a landscape-scale approach. In treatment wetlands, absolute removal rates of nutrients, such as nitrogen (N) and phosphorus (P) are often proportional to the concentration of inflowing water, and the proportion of N and P removed tends to increase as water detention time increases (Jordan et al. 2003). In such wetlands, inflowing water quality, loading rates, and detention time are usually known variables, along with outflowing quality and loading, which makes quantification of internal wetlands services possible to a relatively high level of accuracy. Comparatively little research has looked at quantifying the water treatment capacity of natural wetlands (Verhoeven et al. 2006), and it suggests that it is critical to take landscape-level processes into account.

Because of the common perception that wetlands act as pollution filters in a catchment area, some authors have likened wetlands to a point-source equivalent in a landscape dominated by nonpoint-source pollution. However, uptake of pollutants does not only occur within aquatic ecosystems, but also during the drainage process, as surface and sometimes groundwater flows pass through various environments en route to streams and rivers. In Florida, it was estimated that 9.3% of total nitrogen inputs of a catchment reached surface water, and 19.6% reached the groundwater, with the contribution varying for different types of inputs (Young et al. 2008). The balance was attributed to the assimilation capacity of the soil. Measurement on a landscape scale allows the assessment of the integrated effect of wetlands on downstream water quality, as well as the effect on suspended solids, which cannot be easily measured on an individual wetland scale.

Water-quality amelioration functions of wetlands benefit both the ecology and human users in downstream systems. For example, preventing contamination of downstream areas may protect fisheries from harmful pollutants or reduce the impact on human health, for example, associated with extensive growth of algae or aquatic macrophytes in response to nutrient loading. Reduced sediment loads may reduce the frequency of dredging (and thus the cost) needed to prolong the lifespan of downstream impoundment. Once such services have been quantified, they can be valued using a damage cost avoided or a replacement cost approach (Pearce and Turner 1990; James 1991; Barbier 1993; Emerton et al. 1999).

The aim of this study is to estimate the water treatment capacity of wetlands in the Fynbos Biome of the Western Cape, South Africa, using a novel landscape-scale approach, and to estimate its economic value. The study focuses on the removal of nitrogen, phosphorus, and suspended solids only.

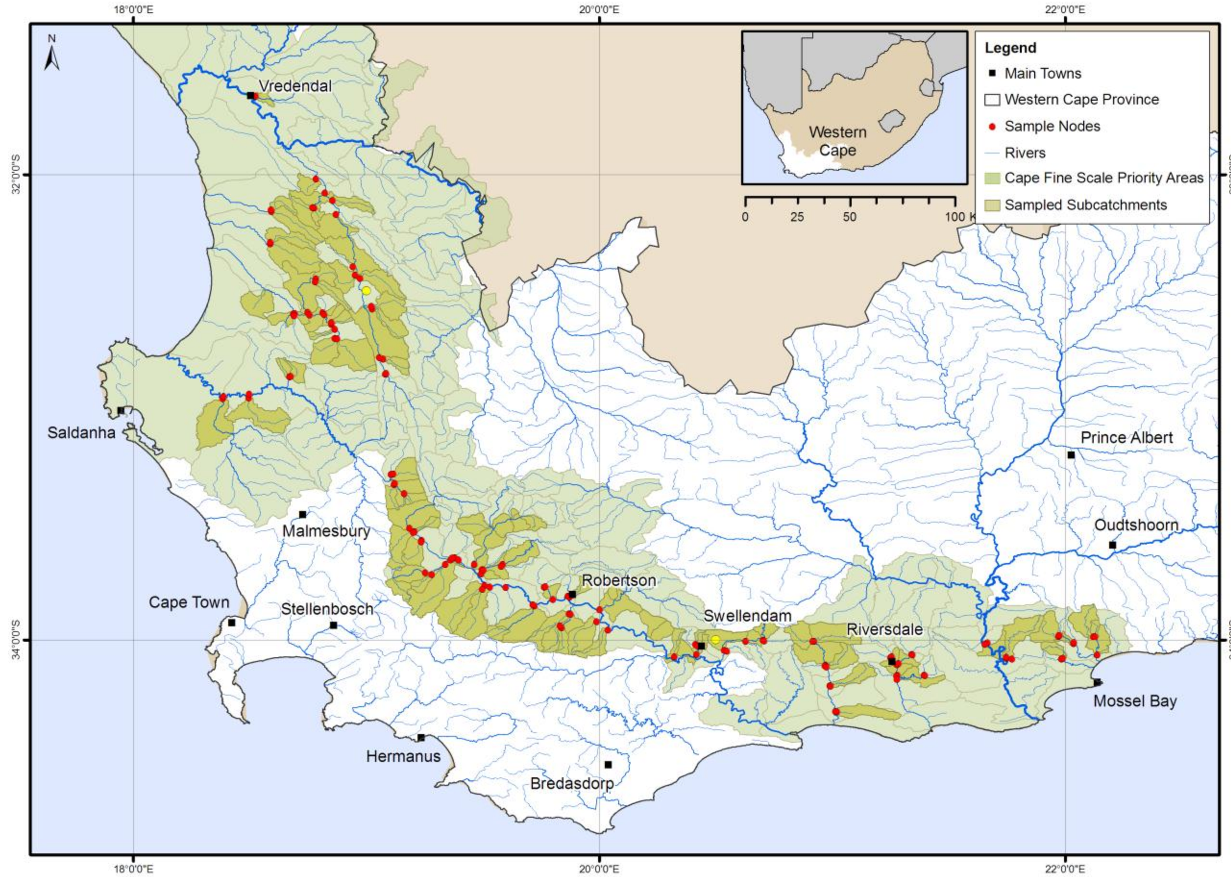
1. Methods

We valued the water treatment capacity of wetlands using a replacement cost approach, which entailed quantifying the removal of pollutants by the wetlands in the study area and estimating the equivalent cost of performing this service with man-made water treatment plants. Because of the difficulties of measuring flows through individual wetlands, we took a landscape approach to estimate the service performed by wetlands, where water quality at catchment outflow points was related to the prevalence of wetlands, as well as other land uses, using multivariate statistical analysis.

1.1 Study Area

We chose to study the Fynbos Biome within the Western Cape Province, South Africa, because accurate and recent fine-scale spatial data on land cover was available, collected as part of the CAPE Fine Scale Planning project (Snaddon et al. 2008). Samples were collected from the outflow points of 100 subcatchments (figure 1), which collectively covered an area of 797,000 hectares. Of these, 75% were fed only by the immediate subcatchment and the remainder was at the outflow points of subcatchments fed by other subcatchments. In the latter case, we assumed that the influence of land cover in the distal subcatchments would be negligible, compared with land cover in the immediate subcatchment, and only land cover in the immediate subcatchment was taken into account.

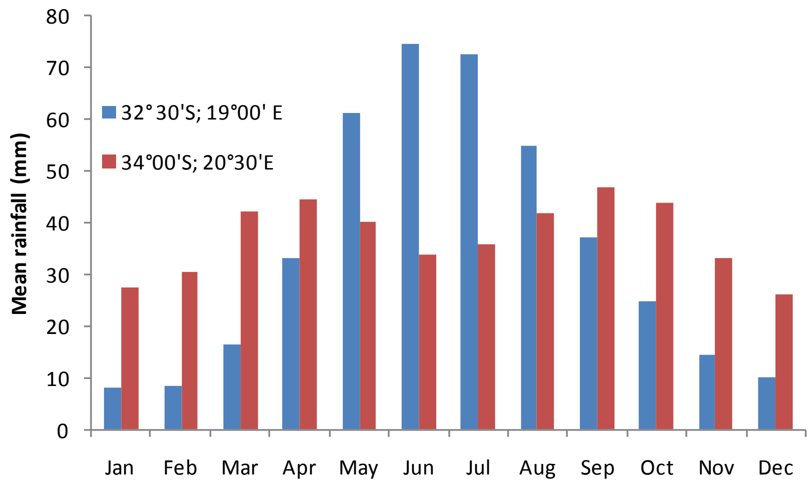
Figure 1. Map of Study Area



The map of the study area shows the location of the sampling points (red dots) and their subcatchment areas (dark green shading), within the area of the Western Cape Province that has been mapped at a fine scale (light green shading). The two points from which rainfall data were taken for figure 2 are marked as yellow dots.
Source: Subcatchment data from CSIR, Stellenbosch.

The study area falls in the winter rainfall (June–September) area of South Africa. Toward the east of the study area, rainfall distribution becomes more bimodal (figure 2). Most of the smaller tributaries within these areas flow as seasonal rather than perennial systems. All of the sampled nodes fall within the seasonal rather than the perennial portion of the catchment areas.

Figure 2. Average Monthly Rainfall from North and East Points in the Study Area



Note: Also see Figure 1.
Source: Zucchini et al. (2002).

The majority of wetlands in the study area are channeled and unchanneled valley-bottom wetlands. Hillslope and valley-head seeps are also well represented, although these two categories were potentially underrepresented as a consequence of poor visibility in the aerial photography that informed much of the Fine Scale Planning Wetland Layer (Job et al. 2008).

The natural vegetation of the Fynbos Biome (and the sampled landscape) is dominated by low shrublands associated with fynbos1 and renosterveld2 vegetation types, although much of it is degraded. Most renosterveld, which occurs on richer soils, has been converted to croplands, and the natural grazing capacity of the remaining fynbos areas is relatively low. Winter wheat (dryland) is the primary agricultural crop, plus significant areas of irrigated orchards and vineyards. Livestock operations tend to be intensive. Urban settlements are concentrated at the coast, and settlements within the sampled area tend to be small. (Detailed land-use data from the CAPE Fine Scale Planning project were grouped together for this study into 13 land-use categories. See figure 3).

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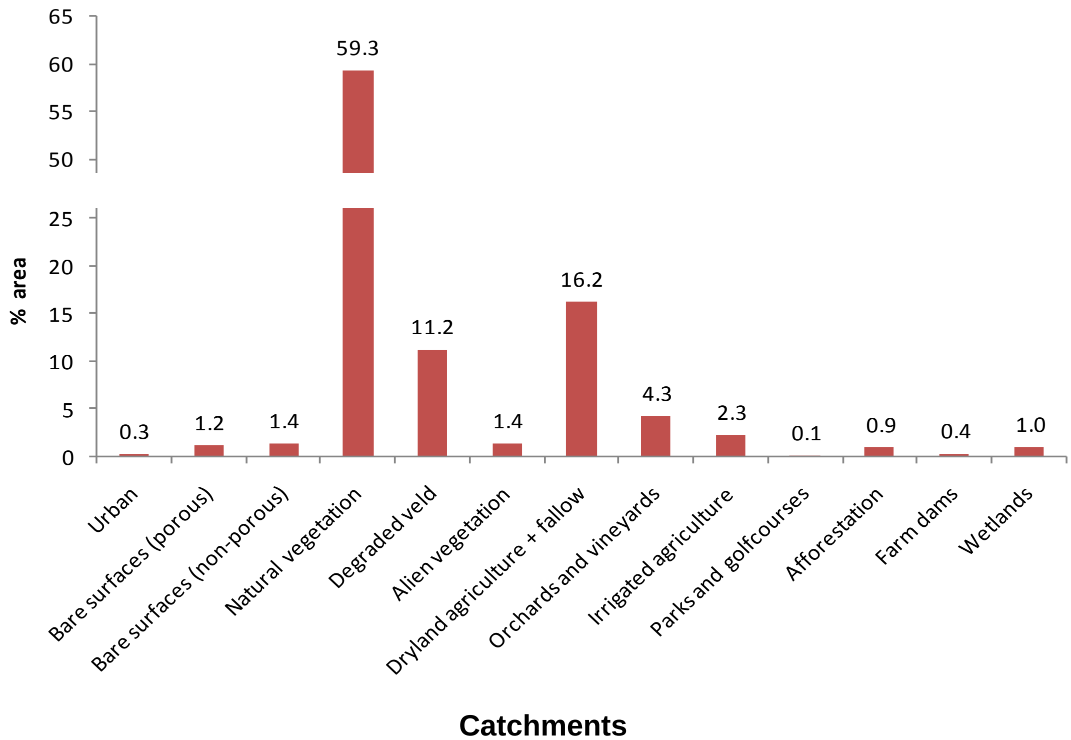
[1] Vegetation unique to the Cape Floral Kingdom made up chiefly of proteaceae, restios, and Ericaceae.

[2] A vegetation type of the Fynbos Biome characterized by small, tough, grey leaves, and predominated by the Daisy family (Asteraceae).

Figure 3. Overall Percentage of Different Land Cover Categories in the Sampled



Source: Authors' calculations based on CAPE Fine Scale Planning data.

1.2 Timing of the Study

Data used in this study were collected between late August and mid-September 2008. The field measurements were taken in the last quarter of the rainy season, when the heavier rains had passed and we assumed the uniform wet-season flushing of the catchments had occurred. This timing helped us avoid inadvertent sampling of certain wetlands during the first major rainfall events (with higher concentrations of nutrients and contaminants), and sampling others after more extensive periods of catchment flow, when water quality may have lower concentrations of nutrients and other dissolved contaminants. From this perspective, estimates of loading based on our research are likely to underrepresent contributions made by wetlands to catchment water quality because the wetlands were sampled at a period usually associated with the most dilute runoff conditions and before application of fertilizers for summer crops.

1.3 Field Data Collection and Analysis

Positions of the outflow "nodes" from each subcatchment were identified using geographic information system (GIS) data and located in the field with a hand-held global positioning system (GPS). Once located, a sampling site was chosen up to a maximum of 200 meters upstream of the nodal point, where flow could be calculated with relative accuracy (i.e., ideal sites had a simple cross-sectional profile). Photographs were taken of each site, a cross-sectional profile of the river and surrounding area was sketched, and depth and flow were measured within the channel along a transect at 0.5-meter intervals. In situ measurements were taken of pH levels, dissolved oxygen concentration, oxygen saturation, temperature, and electrical conductivity. Suspended solid concentrations were calculated by filtering known volumes of water on site through pre-weighed filter papers, which were subsequently dried and weighed, and burned to ash in a muffle furnace (where the material burned is isolated from the fuel and products of combustion) at 450ºC (842ºF), so we could calculate both organic and inorganic suspended sediment components. Water samples of 50 mls were collected at each site, frozen, and later analyzed at the University of Cape Town for concentrations of the following variables: ($NO_3 + NO_2$)-N, ammonium nitrogen (NH_4-N), and orthophosphate (PO_4-P). We calculated downstream loading with instantaneous flow data. The concentrations of selected water quality variables as follows:

Loading (mg/s) = concentration (mg/l) x flow (l/s).

It was hypothesized that nutrient and sediment loads in the water flowing from each subcatchment would be a function of the relative area of different land cover types, as follows:

Loading (mg/s) = $f(A_w, A_1, A_2 \ldots A_n)$,

where A_w = percentage area of wetlands and A_1–A_n are the percentage area of other land-cover types 1…n. The sign of each influential land cover type depends on whether it represents a land cover that is associated with nutrient or sediment input (e.g., irrigated fields) or removal (e.g., as hypothesized here, wetlands).

Since all variables were continuous, linear stepwise multiple regression analysis was used (Statistica 8®). For each nitrite + nitrate nitrogen, ammonium nitrogen, orthophosphate, and total suspended solids, the instantaneous load at the sampling point (quantity per unit time) was regressed against the percentage area of grouped land-cover categories of the catchment apart from bare surfaces. Data were not transformed in any way.

In those cases where wetlands had a significant impact on load, the equation was used to predict what the load would be for each subcatchment if the percentage area of wetlands was changed to zero. The difference between measured value and the latter value was the amount removed by the wetlands. This amount, expressed as a quantity per second, was converted to an absolute amount removed per year by estimating the total time of flow. In the absence of time series data, it was conservatively estimated that this level of service would only be performed during the main rainfall months, and that the elevated loads—expected at the onset of the rainy season when catchments are "flushed"—would largely go untreated due to the high flows during these flushing events. The estimated amount removed annually was then divided by the actual area of wetlands to determine the average rate of removal per hectare of wetland per year in each subcatchment.

1.4 Valuation

The water treatment function was valued using the replacement cost method, based on the cost of treatment in water treatment plants. The data, collected from 24 water treatment plants, included the total amount of water treated, the concentration of N and P before and after treatment, and the capital and operating costs of the plants. Multiple regression analysis was used to estimate the marginal cost of treatment per unit mass of N and P. It was assumed that any treatment service provided by the wetlands was fully demanded, in that it was always beneficial to downstream users, as opposed to a situation where there are few or no users downstream. This is reasonable, given the scarcity of water in general in South Africa due to low rainfall, and the scarcity of clean water in particular due to government failure to provide adequate treatment services (Turton 2008).

2. Removal of Nutrients and Sediments by Wetlands

Both irrigated lands and wetlands studied significantly influenced both nitrite + nitrate nitrogen and ammonium nitrogen loads, with irrigated lands having a positive influence and wetlands having a negative influence (tables 1 and 2). In the case of ammonium nitrogen, degraded veld (rangeland) was also found to have a positive influence (table 2). Although highly significant, the regressions had a poor fit, which suggests that not all important factors were taken into account.

Table 1. Regression Summary for Load ($(NO_3 + NO_2)$-N mg/s

	Beta	Std. error of beta	B	Std. error of B	t(92)	p-level
Intercept			**334.8203**	126.9991	2.63640	0.009835
% Irrigated lands	0.218609	0.099158	**18.4488**	8.3681	2.20465	0.029972
% Wetlands	-0.213352	0.099158	**-43.7603**	20.3381	-2.15164	0.034045

Note: n = 93, F(2,92) = 4.9106, Adjusted $R^2 = 0.077$, $p < 0.001$

Table 2. Regression Summary for Load NH_4-N mg/s

	Beta	Std. error of beta	B	Std. error of B	t(91)	p-level
Intercept			**74.95**	42.59	1.76	<0.10
% Degraded veld	0.562359	0.097672	**9.52**	1.65	5.76	<0.001
% Wetlands	-0.293469	0.096943	**-22.13**	7.31	-3.03	<0.01
% Dryland agriculture	-0.153267	0.090831	**-1.89**	1.12	-1.69	<0.10

Notes: $R^2 = 0.27407242$; adjusted $R^2 = 0.250$; F(3,91) = 11.452; $p < 0.001$.

The above results yield the following equations:

$$N\ (NO_3 + NO_2)\ (mg.s^{-1}) = 334.82 + 18.45{*}\%\ I - 43.76{*}\%\ W \text{, and} \tag{1}$$

$$N\ (NH_4)\ (mg.s^{-1}) = 74.95 + 9.52{*}\%\ DV - 22.13{*}\%\ W - 1.89 * \%\ DA \text{,} \tag{2}$$

where $N\ (NO_3 + NO_2)$ is the load of N leaving a particular subcatchment, % *I* is the percentage area of irrigated lands (including orchards, vineyards, pastures, parks, and golf courses) in the subcatchment, % *W* is the percentage area of wetlands in the subcatchment, % *DV* is the percentage area of degraded veld in the subcatchment, and % *DA* is the percentage area of dryland agriculture in the subcatchment.

PO_4-P loading was not significantly correlated with any form of land cover. The results for total suspended solids (TSS) suggest that sediment loads were driven predominantly by the presence of dryland agriculture, which have a positive impact on sediment loads, while wetlands did not have a significant impact on downstream sediment loads (table 3).

Table 3. Regression Summary for TSS Load g/h

	Beta	Std. error of beta	B	Std. error of B	t(96)	p-level
Intercept			-0.028551	0.335998	-0.084974	0.932459
% Dryland agriculture	0.271353	0.098233	0.033448	0.012109	2.762353	0.006878

Notes: Adjusted $R^2 = 0.064$; $F(1,96) = 7.6306$; $p < 0.05$. TSS = total suspended solids.

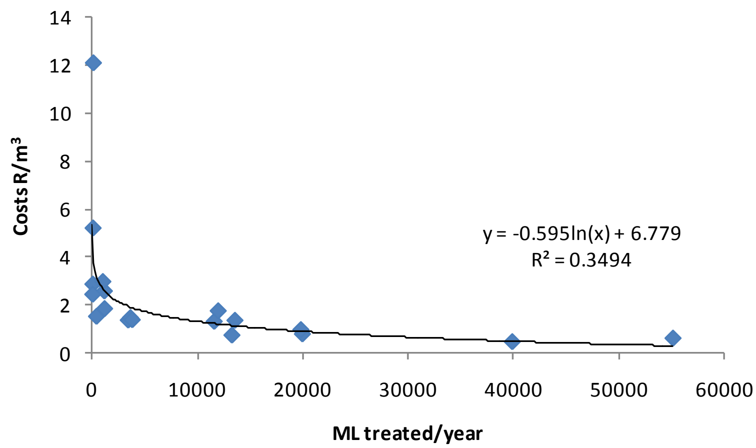
Equations 1 and 2 were used to estimate the removal of N per unit area per year in each catchment by applying the equation with and without the wetland area. Estimated removal rates ranged from 307 kg to 9,505 kg N per ha^{-1} $year^{-1}$, with an average of 1,594 ± 1,375 kg N per ha^{-1} $year^{-1}$.

3. Valuation of the Water Treatment Service

Costs of water treatment vary according to the quantity treated due to economies of scale (figure 4). The weighted average treatment cost of the 19 plants for which detailed data were available was R0.83.m^{-3}. About R0.63 m^{-3} of this is the annual operating cost and the balance is depreciation and maintenance of capital. Current construction costs were estimated to be in the range of ZAR 7 million (USD 0.9 million)[3] per mega liter (Ml) for a large works and up to ZAR 10 million (USD 1.37 million) for smaller works.

[3] ZAR = South African rand; ZAR 7.5 = US$ 1.

Figure 4. Variation of Total Costs of Water Treatment Plants per Unit and the Quantity of Effluent Treated Annually (n = 19 Treatment Works)



Note: Total costs = capital depreciation, maintenance, and operating costs.
Source: Authors' calculations.

The amounts of TSS, N, and P removed were highly correlated (table 4). Thus, it was not possible to perform multiple regression analysis to isolate marginal costs of removal of any one constituent.

Table 4. Correlation Matrix between the Average Daily Cost of Treatment (Including Cost of Capital) and Removal of TSS, NH_3-N, and PO_4-P

	Cost ZAR/day	TSS kg	Kg N	Kg P
Cost ZAR/day	1.00			
TSS kg	0.76	1.00		
Kg N	0.74	0.97	1.00	
Kg P	0.75	0.97	0.96	1.00

Notes: All correlations are highly significant (P<0.001).
TSS = total suspended solids; NH3-N = ammonium nitrogen; PO4-P = orthophosphate; Kg N = kilogram of nitrogen; kg P = kilogram of phosphorus.
ZAR 7.5 = US$ 1

In other words, while treatment works are designed primarily with the removal of P in mind (and thus are driven by the average cost per kg of P removed), if N was the targeted nutrient, the costs of treatment would not differ significantly from the average cost per kg of N removed that is achieved while P is being targeted. Thus, the value of treatment by wetlands can theoretically be determined as follows:

$$\text{Value (ZAR/y)} = \text{Max (kg TSS removed x } C_{TSS}\text{, kg N removed x } C_N\text{, kg P removed x } C_P\text{)},$$

where:

$$C_i = \text{total cost of treatment / total kg of substance } i \text{ removed} .$$

The rates of removal of different substances from water treatment works in the Western Cape suggested that an average of at least 33 mg of N is removed per liter of effluent, which translates to 0.033 kg per cubic meter. Based on the above, the average cost of treatment was about ZAR 26 (US$ 3.47) per kg of N removed (from total ammonium). The analysis was limited by lack of data on the rate of removal of total P, with ortho PO4-P only accounting for 67% of influent total P. If removal of total P was also correlated with cost of treatment and removal rates of the other elements, then it could be assumed that the cost of treatment in terms of P removal was in the order of ZAR 71 ($9.47) per kg of P. Similar estimates could not be made for total N due to lack of data on influent concentrations of NO3-N. These values are not additive and are merely calculated for application in the above equations (table 5).

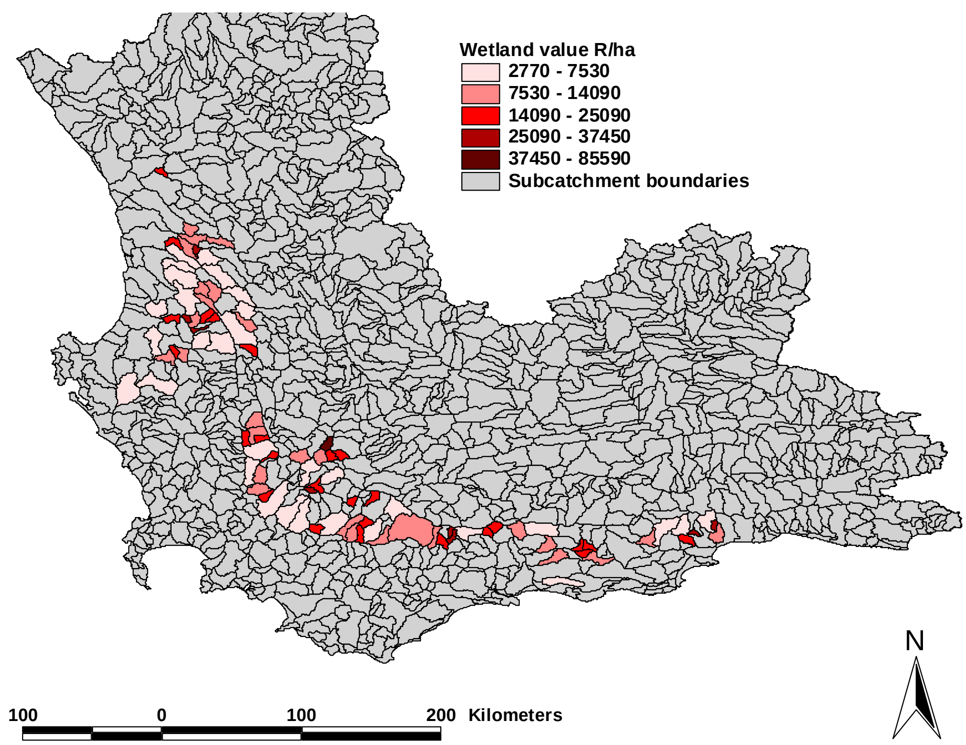
Table 5. Estimated Average Removal Rates in Water Treatment Works

	TSS	NO_3	NH_3	Ortho P	Total P
	mg/l	*mg N/l*	*mg N/l*	*mg P/l*	*mg P/l*
Sample size (treatment works)	20	20	24	24	19
Average influent concentration	475.5	No data	37.7	8.2	14.4
Average effluent concentration	24.8	3.4	4.7	3.3	No data
Difference (mg)	450.8		33.0	5.0	
Removal rate (kg/m^{-3})	0.451		0.033	0.005	0.009*
Average cost per substance (ZAR/kg) (not additive among substances)	ZAR 2.17		ZAR 26.16		ZAR 71.15

* Assumes similar rates of removal as for ortho P.

On this basis, using only the removal of ammonium nitrogen to avoid double-counting, and assuming that removal of total P is correlated to that of N, the value of wetlands in the different subcatchments was estimated to have an average value of ZAR 14,350 ± 12,385 (US$ 1,913 ± 1,651) ha^{-1} per y^{-1}, and the total value of wetlands in the study area was estimated to be ZAR 328 million ($ 43.7 million). There was no spatial pattern in the average value of wetlands in different subcatchments, but higher values tended to be associated with smaller subcatchments (figure 5).

Figure 5. Variation in the Average Value of Wetlands in the Sampled Subcatchments.



Source: Authors' calculations.

4. Discussion

The results of this study suggest that both wetlands area and land use do play a role in determining water quality, as expected, although the regression models were weak, implying that not all factors had been considered. The results suggested that irrigated lands (including orchards, vineyards, pastures, parks, and golf courses) and dryland agriculture increase the concentrations of nitrogen (in ammonium, nitrates, and nitrites), probably due to the application of fertilizers in these areas, while wetlands have the opposite effect.

4.1 Factors Influencing Water Quality on a Landscape Scale

There were no significant correlations between land cover and dissolved phosphorus concentrations. This was probably due to the fact that much of the total phosphorous load is bound to particles (DWAF 1996). Nevertheless, we would expect effects on total phosphorus to be similar to the effects on TSS. Dryland agriculture had a weakly positive influence on TSS, but wetlands were not found to play a role. The influence of dryland agriculture probably relates to the high potential for erosion in these disturbed areas and the prevalence of drainage channels across them, which convey water to downstream drainage systems, potentially bypassing remnant wetland areas which might have had an ameliorating impact. Data on the condition of the wetlands would potentially have shed some light on this aspect: channelization of valley-bottom wetlands is a common impact to this type of wetlands and likely to dramatically affect the efficacy of wetlands functions, such as sediment trapping and associated phosphorus removal (see Ellery et al. 2009).

Surprisingly, farm dams did not have a significant influence on any of the water quality parameters considered. Indeed, trapping TSS in farm dams has been hypothesized as a primary mechanism for reducing downstream phosphorus loading in agricultural areas. The lack of correlation may be linked to the timing of the water quality study in late winter, when small farm dams were likely to be full and only have capacity for reduced rates of sediment retention.

The analysis was limited in that it did not take wetlands types and conditions into account (due to a lack of data) or other factors that might have an influence on water quality, including antecedent runoff events, natural hydrology, and geology.

Wetlands types and conditions are both are likely to play an important role in determining the efficacy with which different wetlands are able to ameliorate water quality. Job et al. (2008) outlined the likely roles of different wetlands on a catchment scale, noting that only those wetlands directly linked to surface and/or groundwater flows through a catchment area are likely to exert a measurable impact on water quality. Hence, channeled and unchanneled valley bottom wetlands (in terms of wetlands types specified by SANBI 2009), river channels, and hillslope and valley-head wetlands would be the main wetlands types expected to play a role in catchment-level impacts on water quality. On the other hand, depressional wetlands and flats, which are not linked to directional flow, are unlikely to affect water quality at this scale.

Kotze et al. (2008) postulated probable levels of delivery of ecosystem services from different types of wetlands. In terms of sediment trapping and phosphate and nitrate removal, valley bottom wetlands (which dominate the wetlands of the study area) were attributed low-to-moderate levels of function (with unchanneled valley-bottom wetlands accruing a higher rating in terms of sediment trapping than channeled valley-bottom systems), hillslope seeps (which are also well represented) were attributed higher levels of ecosystem service in terms of nitrate removal (but were considered to play no role in phosphorus removal), and floodplain wetlands (rare, if not absent, in the study area) were assumed to play an important role in phosphorus removal.

Ideally, the approach should have included multiple site visits, carried out over a full annual flow cycle. The timing of the study in late winter attempted to standardize the effects of antecedent rainfall effects to a degree by allowing for data collection at a time when dry season accumulations of nutrients and other pollutants in the catchments were likely to have been flushed out of the catchment by rainfall during the early and peak wet season. Although the impact of abstraction on the efficacy of wetland function was not specifically measured, it was accounted for indirectly in terms of the presence of farm dams. Accurate present-day mean annual runoff data were not available for the mapped subcatchment areas. However, real-time data measured at each sampling “node” allowed at least comparison of estimated loading between catchments.

Geology was not considered to be a variable and might indeed play a role in the discrimination of water quality characteristics between different potions of the study area. Since the study area comprised three broad vegetation zones (mapped for the Fine Scale Planning study by Helme [2008]), it is likely that these botanical zones respond at least in part to changes in geology. Future work on this project should include testing for differences in water quality between catchments in different botanical zones. Budget and logistical constraints in the present study limited the extent to which such question could be explored, given the limited number of sites sampled across the three broad botanical areas included in the Fine Scale Planning Study.

4.2 Capacity of Wetlands for Water Quality Treatment

Estimates of the rates of removal of nitrogen by wetlands in the study area were higher than expected and fell within the broad ranges for nitrogen removal observed in artificial wetlands, 300–9,000 kg per ha^{-1} $year^{-1}$ (Verhoeven et al. 2006).

The high levels of variability in removal rates estimated in this study are likely to reflect differences in land use, but to some extent may also reflect the unassessed variability in wetlands types and conditions, as well as regional variation in precipitation, evaporation, and vegetation. Variability in wetlands characteristics may be associated with greater or lesser efficacy in terms of facilitating pollutant sequestration (Kadlec and Knight 1996), especially differences in the degree to which flows are spread through wetlands and affect aquatic contact with microbial communities. Wetlands conditions, as noted earlier, is also likely to be a primary determinant of wetland function on a landscape level, and this in turn is likely to be tied into land use and the existence of an ecological buffer area or "setback" between wetlands and their surrounding land use. These uncertainties highlight the importance of providing a measure of wetlands conditions and types, if greater levels of confidence are to be attached to modeled valuations of wetlands ecosystem services.

Kadlec and Knight (1996) also found seasonal differences in nitrogen uptake by aquatic macrophytes, with uptake in temperate climates at a maximum during spring and summer, and die back (associated with the release of nitrogen nutrients back into wetland soils and waters) often occurring in autumn and early winter. Seasonal variation in nitrogen uptake was not investigated in the present study, but should be considered in efforts to fine-tune models for wetland loading rates.

4.3 Valuation

We used a replacement cost technique in the valuation of the wetlands service. However, it is difficult to isolate the cost of removal of different water quality variables in the treatment process. The main costs entailed in design and management of water treatment works are usually associated with the reduction of phosphorus and total ammonia concentrations to levels that concur with licensing requirements. These in turn are often dictated by ecological concerns, where phosphorus is often a limiting nutrient in natural inland aquatic ecosystems.

Elevated phosphorus concentrations are usually associated with increased productivity, such as increases in algal and/or cyanobacterial blooms, and increased invasion by (often alien) aquatic macrophytes. Management of ammonia concentrations is also accorded high priority. Although the total nitrogen loading associated with ammonia is often lower than that associated with nitrates and nitrites, the un-ionized form of ammonium nitrogen (ammonia-NH_3) is highly toxic to many aquatic organisms, even at very low concentrations (DWAF 1996). It might be argued that a plant designed purely for the removal of sediment and nitrogen might be less costly, and the values applied in this study could thus be an overestimate. Furthermore, most of the waste-water treatment works analyzed in this study are currently operating over-capacity, with the result that they might not have been as efficient as they were designed to be. These are areas that deserve further study. Nevertheless, conservative assumptions were applied, and we are confident that the estimates are in the right order of magnitude.

Because of the economy of scale and the lack of data on the removal of nitrogen in the form of nitrates and nitrites, it was not possible to derive the marginal cost of removal of different substances. The assumption that the average cost of treatment can be attributed to nitrogen might produce an overestimate of value. This is particularly the case if the assumption that total phosphorus removal is correlated with nitrogen removal is relaxed. If wetlands do not remove phosphorus, then water treatment would still be necessary and the wetlands would not perform a cost-saving service. Thus, the value estimates in this study must be viewed with caution and taken only to be potential values of the service.

4.4 Scale of Study

This study allows wetlands to be valued at the subcatchment level—assigning an average value per hectare to all wetlands in a particular subcatchment. In reality, the service performed will vary among the wetlands, depending on their position in the landscape, their type, and their condition. Nevertheless, understanding value at this scale may be useful in prioritizing conservation and restoration action or in analyzing broad-scale conservation trade-offs. In order to estimate value on a more local scale, it would be necessary to assess the relative value of different types of wetlands, the influence of their position in the landscape, and the influence of their condition (Kotze et al. 2008; Turpie et al. 2009).

5. Conclusion

Although in need of further investigation and refinement, the findings of this study suggest that wetlands should be given considerably more attention in land-use planning and regulation. If current trends are allowed to persist, then in-stream water quality problems already being experienced over much of South Africa will be exacerbated. Given the potentially high value of wetlands, particularly in stressed catchments, efforts should be made to regulate their protection and, where possible, to incentivize this. Indeed, the results of this study suggest that the services provided by wetlands could be sufficiently valuable to warrant the introduction of a payments-for-ecosystem-services mechanism, in which downstream users contribute to the protection of catchment area wetlands.

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