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# Assessing the Impacts of Four Land Use Types on the Water Quality of Wetlands in Japan

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## Abstract

This study examined how changes in the composition of land use can affect wetland water quality. Twenty-four wetlands located in Hiroshima prefecture in the western part of Japan were selected for this purpose. The water quality parameters that were explored include: pH, electrical conductivity, turbidity, dissolved oxygen, total dissolved solid, temperature and different forms of nitrogen. These important indicators of the water quality in the study area were measured from December 2005 to December 2006. The composition of land uses was determined for the catchments of the wetlands. They were then categorized into three classes, including non-disturbed, moderately-disturbed and highly-disturbed wetlands, based on the extent of urban area (as the most disruptive land use type within the catchment of the wetlands). The relationship between land use types and water quality parameters for the wetlands was statistically examined. The findings indicated that there were significant positive relationships between the proportion (%) of urban areas within catchments of the wetlands and EC ($r = 0.67$, $p < 0.01$), TDS ($r = 0.69$, $p < 0.01$), TN ($r = 0.92$, $p < 0.01$), DON ($r = 0.6$, $p < 0.01$), $NH_4^+$($r = 0.47$, $p < 0.05$), $NO_2^-$ ($r = 0.50$, $p < 0.05$), while negative relationships were observed between the proportion (%) of forest area in these wetlands and EC ($r = –0.62$, $p < 0.01$), TDS ($r = –0.68$, $p < 0.01$), TN ($r = –0.68$, $p < 0.01$), DON ($r = -0.43$, $p < 0.05$), and $NH_4^+$ ($r = –0.55$, $p < 0.01$). Analysis of the variance also revealed significant differences within the wetland groups in terms of the annual mean of electrical conductivity, total dissolved solids, total nitrogen, nitrite, dissolved inorganic nitrogen and dissolved organic nitrogen in the study area. Moreover, the study also indicated that the forest area plays a significant role in withholding nutrient loads from the wetlands, and hence, it can act as a sink for surface/subsurface nutrient inputs flowing into such water bodies from the watersheds.

**Keywords** Wetland – Land use – Water quality – Catchment

## 1 Introduction

Aquatic ecosystems in general, and wetlands in particular, have been used by humans over the centuries to the extent that not that many have remained today in their natural condition, as a result of pollution loads, among other reasons (Ngoye and Machiwa *2004*)**.** The water quality of water resources is generally linked with land use within a catchment since a catchment can influence the quality and quantity of runoff during and after a rainfall event (Richards and Host *1994*). Hence, a lot of water pollution problems are caused by changes in the composition of land use within a catchment as human activities increase (Gikas et al. *2006*; Amiri and Nakane *2009*; Boskidis et al. *2011*). Among different aquatic ecosystems, wetlands not only play a key role in water quality improvement of other water bodies such as rivers at the catchment-scale, but they are also themselves influenced by changes in the composition of land uses due to human activities within their catchments (Plameri and Treppel *2002*; Papastergiadou et al. *2008*). Removal of nutrient loads from running waters is an important role of wetlands in watersheds (Jones et al. *2001*; Jordan et al. *2003*; Akratos et al. *2006*; Plameri and Treppel *2002*); these types of water bodies are often called the "kidney" of a watershed (Brooks et al. *2003*). The nutrient removal ability of wetlands is based on trapping sediment, removing nutrients, storing and releasing inorganic nutrients and transforming them into organic forms (DeBusk *1999*). Due to their significant role in the improvement of the environmental quality of catchments, significant attention has being given to the conservation and restoration of natural wetlands, as well as the construction of artificial ones in catchments (Tsihrintzis et al. *1995*; Lee et al. *2005*).

Due to the degradation in wetland ecosystems resulting from changes in the composition of land use of their catchments (Papastergiadou et al. *2008*), significant research has been conducted (*e.g.*;Schueler *1994*; Arnold and Gibbons *1996*; Tsihrintzis and Hamid *1997*; May et al. *1997*; Brabec et al. *2002*; Clapcott et al. *2011*) to explore the relationship between urban area (impervious cover) and non-point source pollution, in addition to documenting the adverse impacts on water quality. Although pollution is not generated by impervious covers, many of the physical and biological impacts affecting the quality of water resources, such as streams and wetlands, originate due to the impervious cover-induced hydrological changes (May et al. *1997*). The imperviousness of the urban areas increases their hydrological activity, and even small rainfalls are capable of washing the accumulated pollutants into water bodies (Basnyat et al. *1999*). Regarding the assessment of water quality conditions of streams, several authors (*e.g.*, Schueler *1994*; Arnold and Gibbons *1996*; May et al. *1997*; Helms et al. *2009*; Gregory and Calhoun *2007*) have suggested a linkage between the impervious cover and in-stream water quality degradation. Out of the afore-cited works, Schueler (*1994*) has described the impacts of an increase in the impervious cover on in-stream water quality as occurring in three ascending levels namely, less than 10 % (non-disturbed streams), 10 ~ 25 % (semi-disturbed streams) and more than 25 % (disturbed streams) within the catchment of a river. Moreover. Amiri et al. (*2012*) has determined a threshold for two disturbing land uses, namely, agricultural area (60 %) and grassland area (10 %) within the catchment of the rivers in relation to total phosphorus, so that when exceeding those thresholds, the in-stream total phosphorus concentration increases significantly. May et al. (*1997*) suggest a 10 % impervious cover threshold that can be applied to wetland communities. Moreover, an inverse correlation between wetland habitat quality and increasing impervious cover has also been reported by Clapcott et al. (*2011*), which is consistent with the 10 % impervious cover threshold as suggested by May et al. (*1997*). Although there have been some studies in which the relationship between a change in the composition of land uses and variations in the water quality parameters of wetlands has been examined (*e.g.*; Tsihrintzis et al. *1996*, *1997*; Crosbie and Chow-Fraser *1999*; Daley and McDowell *2002*; Houlahan and Findley *2004*; Haidary and Nakane *2008*), few studies have applied an ordination approach to specify how changes in land use types in the catchment of the wetlands affects their water quality. Accordingly, the objective of this study is firstly to assess whether different types of human activities expressed through four land uses (i.e., urban, forest, agricultural and grassland) within the catchment of the wetland can affect water quality parameters, and secondly whether a threshold of water quality degradation might be determined considering changes in the percentage of urban area as a disturbing land use type in the catchment of the wetlands of interest.

## 2 Materials and Methods

### 2.1 Study Area

The study area of this research project is located in Higashi-Hiroshima in Japan, which is placed within 132° 36′ 23″ ~ 132° 51′ 19″ E and 34° 15′ 19″ ~ 34° 34′ 58″ N, with an area of 635 km$^2$ (Fig. 1). Annual rainfall is, on average, 160 mm/month, with a maximum monthly value of 304 mm and a minimum monthly value of 19 mm, which were recorded in July and October in the study area, respectively. Although mean annual temperature is 14.1 °C, the monthly mean varies from 2.3 °C (in January) to 26 °C (in August) (Japan Meteorological Agency). Based on spatial analysis, which was conducted using geological formation, soil and land-use maps of the wetlands, it was found that granite and alluvial sand are the main geological formations in the study area. Dominant and sub-dominant soil types are residual regosols and brown forest soil, respectively. Figure 1 depicts the geographical distribution of the 24 wetlands explored in this study. It is noteworthy that all the wetlands have similar geographic features, except the physical features of the watersheds. Twenty-four out of 1,100 wetlands, as reported by Shimoda (*1993*), were chosen as our study sites using a topographical map (1:50,000) (Japan Geographical Survey Institute). Table 1 indicates geometric features, the proportion (%) of land use types and that of soil types and geological formations for the catchment of each wetland.

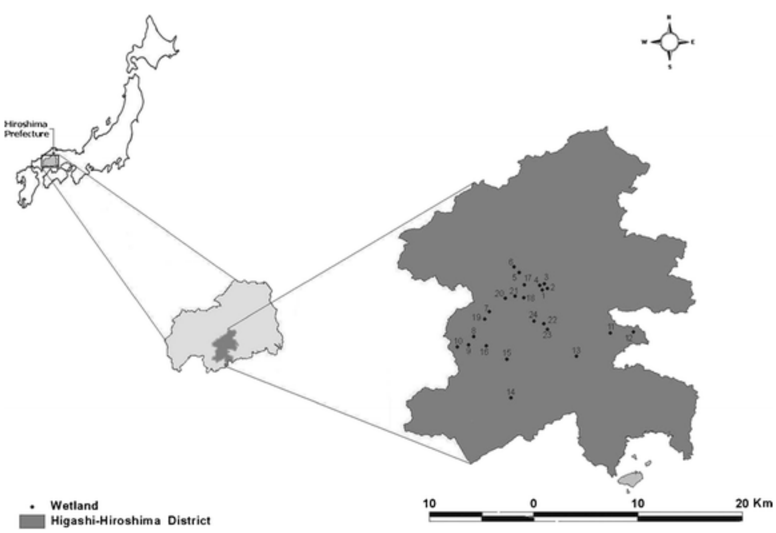


**Fig. 1**

Geographical position of the study area

**Table 1**

Geometric features, soil type and geological formations of the catchment of the wetlands in study area

| Wetland no. | Geometrics of the catchments | | | | | Proportion (%) of soil type | | | | | Proportion (%) of geol formation | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Catchment area (hac) | Average catchment slope (%) | Strahler order | Drainage density (m/hac) | Average main channel slope (%) | Gray lowland soil | Residual regosol | Brown forest soil | Brown forest soil (dry) | Regosol | Diluvial sand | Granite |
| 1 | 39.43 | 12.14 | 2 | 58.39 | 12.14 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 82.63 | 0.00 |
| 2 | 9.43 | 18.12 | 2 | 65.18 | 18.12 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 3 | 2.53 | 10.21 | 1 | 17.01 | 10.21 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 4 | 14.74 | 4.48 | 2 | 47.76 | 4.48 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| 5 | 14.38 | 2.75 | 2 | 51.82 | 2.75 | 5.00 | 0.00 | 7.34 | 91.00 | 0.00 | 82.89 | 17.11 |
| 6 | 27.45 | 17.33 | 2 | 42.51 | 17.33 | 0.00 | 0.00 | 100 | 0.00 | 0.00 | 0.00 | 0.00 |
| 7 | 17.07 | 7.54 | 2 | 51.45 | 7.54 | 0.00 | 5.40 | 94.60 | 15.02 | 0.00 | 0.00 | 84.73 |
| 8 | 330.87 | 14.76 | 4 | 49.56 | 14.76 | 0.00 | 1.00 | 0.00 | 0.00 | 0.00 | 16.72 | 83.28 |
| 9 | 4.5 | 1.92 | 1 | 20.32 | 1.92 | 0.00 | 0.00 | 0.00 | 0.00 | 1.32 | 0.00 | 9.72 |
| 10 | 42.98 | 11.78 | 2 | 46.58 | 11.78 | 0.00 | 0.00 | 0.00 | 0.00 | 17.72 | 0.00 | 0.06 |
| 11 | 4.14 | 15.75 | 1 | 26.69 | 15.75 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.26 | 97.92 |
| 12 | 2.55 | 20.83 | 1 | 44.46 | 20.83 | 0.00 | 0.00 | 0.00 | 0.00 | 100 | 57.63 | 6.46 |
| 13 | 4.52 | 7.89 | 1 | 26.32 | 7.89 | 0.00 | 0.00 | 89.17 | 0.00 | 0.00 | 100 | 94.11 |
| 14 | 150.78 | 6.35 | 3 | 44.54 | 6.35 | 50.43 | 0.00 | 26.15 | 23.42 | 0.00 | 19.63 | 80.37 |
| 15 | 3.09 | 4.19 | 1 | 35.05 | 4.19 | 0.00 | 79.35 | 0.00 | 7.92 | 0.00 | 0.00 | 61.91 |
| 16 | 34.97 | 3.18 | 2 | 53.86 | 3.18 | 0.00 | 1.66 | 0.00 | 48.32 | 0.00 | 45.83 | 54.17 |
| 17 | 44.08 | 1.66 | 3 | 47.64 | 1.66 | 27.28 | 72.72 | 0.00 | 0.00 | 0.00 | 1.39 | 98.61 |
| 18 | 2.78 | 10.39 | 1 | 39.64 | 10.39 | 0.00 | 100 | 0.00 | 24.63 | 0.00 | 72.83 | 27.17 |
| 19 | 21.30 | 9.41 | 2 | 30.94 | 7.53 | 0.00 | 8.62 | 91.38 | 0.00 | 0.00 | 0.00 | 27.08 |
| 20 | 14.53 | 11.74 | 2 | 28.87 | 11.74 | 1.02 | 18.38 | 80.60 | 0.00 | 0.00 | 25.26 | 74.74 |
| 21 | 9.40 | 5.28 | 1 | 41.06 | 5.28 | 0.00 | 72.77 | 27.73 | 0.00 | 0.00 | 28.68 | 100.00 |
| 22 | 5.08 | 3.08 | 1 | 29.59 | 3.08 | 18.83 | 81.17 | 0.00 | 0.00 | 0.00 | 47.84 | 52.16 |
| 23 | 26.23 | 10.28 | 2 | 46.29 | 10.28 | 100 | 0.00 | 0.00 | 0.00 | 0.00 | 2.08 | 17.37 |
| 24 | 45.35 | 8.62 | 3 | 46.04 | 8.62 | 100 | 0.00 | 0.00 | 0.00 | 0.00 | 100 | 0.00 |

## 2.2 Materials and Methods

Hydrological modeling is an efficient tool to explore the relationship between entities in a given catchment (*e.g.*: Boskidis et al. *2011*); statistical approaches are also helpful when one has an appropriate amount of data and no adequate knowledge regarding the entities of interest (Grant et al. *1997*). In this study, a statistical approach was applied. According to this approach, environmental impacts of human activities are statistically assessed by partitioning the ecosystem of interest into homogeneous environmental impact units considering the extent of the human disturbances, which may occur across the ecosystem of interest

(Liddle *1975*). These homogenous environmental impact units are then called the disturbed, semi-disturbed, and non-disturbed environmental impact units, respectively. These ecological impact units can also statistically be considered as treatments, and environmental impacts of the human disturbances can statistically be analyzed within and between these spatial treatments.

A variety of water quality parameters in the study area wetlands, namely pH, electrical conductivity (EC mS/m), turbidity, dissolved oxygen (DO mg/L), temperature (C), and total dissolved solids (TDS mg/L) were measured by a portable water quality monitoring device (HORIBA, Model U-21 XD) in the inflows of each wetland throughout the course of four seasons in 2006 (Table 2). In flowing surface were simultaneously sampled and immediately placed in a cooler in order to transport to the laboratory for determining different forms of nitrogen including nitrate ($NO_3^-$), nitrite ($NO_2^-$) (Ion Chromatography Method) and ammonium ($NH_4^+$) (APHA *1995*), dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON) (Semi-Micro Kjeldahl Method), and total nitrogen (TN) (4500-Norg C).

**Table 2**

Annual mean of water quality parameters of the wetlands

| Wetlands | Water quality parameters | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | pH | Ec (mS/m) | Turbidity (mg/L) | DO (mg/L) | TDS (mg/L) | T (C) | TN (mg/L) | $NO_3^-$ (mg/L) | DON (mg/L) | $NH_4^+$ (mg/L) | $NO_2^-$ (mg/L) | DIN (mg/L) |
| 1 | 6.27 | 12.33 | 15.00 | 6.60 | 0.08 | 18.44 | 0.75 | 0.03 | 0.68 | 0.007 | 0.006 | 0.06 |
| 2 | 6.50 | 4.00 | 12.75 | 6.95 | 0.03 | 18.20 | 0.55 | 0.03 | 0.52 | 0.003 | 0.005 | 0.03 |
| 3 | 6.20 | 11.75 | 9.75 | 5.98 | 0.08 | 16.63 | 0.93 | 0.02 | 0.89 | 0.006 | 0.003 | 0.05 |
| 4 | 6.28 | 14.00 | 23.00 | 7.43 | 0.10 | 18.60 | 0.60 | 0.06 | 0.51 | 0.016 | 0.007 | 0.08 |
| 5 | 6.63 | 15.75 | 9.75 | 5.65 | 0.14 | 17.95 | 1.06 | 0.15 | 0.66 | 0.232 | 0.021 | 0.40 |
| 6 | 6.48 | 7.00 | 7.25 | 5.10 | 0.05 | 19.15 | 0.60 | 0.02 | 0.57 | 0.004 | 0.004 | 0.03 |
| 7 | 6.85 | 5.00 | 21.5 | 9.35 | 0.03 | 17.40 | 0.49 | 0.02 | 0.47 | 0.000 | 0.004 | 0.02 |
| 8 | 6.75 | 6.75 | 7.00 | 4.45 | 0.05 | 18.73 | 0.47 | 0.03 | 0.43 | 0.005 | 0.002 | 0.04 |
| 9 | 6.53 | 4.50 | 3.00 | 7.38 | 0.03 | 18.80 | 0.50 | 0.04 | 0.46 | 0.001 | 0.002 | 0.04 |
| 10 | 6.40 | 10.50 | 8.25 | 7.65 | 0.05 | 17.93 | 0.49 | 0.21 | 0.27 | 0.007 | 0.007 | 0.22 |
| 11 | 6.58 | 9.50 | 20.50 | 6.93 | 0.06 | 18.38 | 0.64 | 0.15 | 0.40 | 0.082 | 0.005 | 0.24 |
| 12 | 6.88 | 8.50 | 14.25 | 7.90 | 0.07 | 16.43 | 0.42 | 0.08 | 0.34 | 0.005 | 0.003 | 0.08 |
| 13 | 6.35 | 7.75 | 11.75 | 4.73 | 0.05 | 18.35 | 0.63 | 0.05 | 0.56 | 0.018 | 0.003 | 0.07 |
| 14 | 6.40 | 9.00 | 18.00 | 6.78 | 0.06 | 19.90 | 0.89 | 0.03 | 0.77 | 0.009 | 0.015 | – |
| 15 | 6.57 | 8.00 | 16.00 | 6.53 | 0.05 | 19.48 | 0.66 | 0.03 | 0.41 | 0.004 | 0.004 | 0.04 |
| 16 | 6.64 | 16.50 | 22.25 | 7.35 | 0.11 | 19.28 | 1.28 | 0.25 | 0.90 | 0.099 | 0.019 | 0.37 |
| 17 | 6.30 | 13.00 | 31.75 | 6.50 | 0.09 | 17.80 | 0.87 | 0.08 | 0.70 | 0.06 | 0.027 | 0.17 |
| 18 | 6.19 | 8.00 | 8.75 | 6.10 | 0.05 | 18.80 | 0.51 | 0.03 | 0.48 | 0.007 | 0.000 | 0.02 |
| 19 | 6.44 | 6.50 | 14.50 | 6.13 | 0.04 | 19.65 | 0.42 | 0.04 | 0.37 | 0.006 | 0.002 | 0.05 |
| 20 | 6.44 | 7.00 | 16.00 | 6.80 | 0.05 | 18.73 | 0.46 | 0.05 | 0.40 | 0.007 | 0.002 | 0.06 |
| 21 | 6.38 | 8.25 | 23.25 | 6.95 | 0.06 | 19.05 | 0.78 | 0.02 | 0.75 | 0.006 | 0.001 | 0.03 |
| 22 | 6.07 | 4.75 | 12.20 | 10.22 | 0.03 | 13.13 | 0.42 | 0.07 | 0.32 | 0.025 | 0.005 | 0.10 |
| 23 | 6.63 | 5.33 | 52.30 | 7.84 | 0.04 | 16.50 | 0.63 | 0.06 | 0.48 | 0.069 | 0.023 | 0.15 |
| 24 | 6.51 | 15.25 | 18.75 | 5.34 | 0.10 | 16.26 | 2.26 | 0.65 | 0.36 | 1.164 | 0.086 | 1.90 |

All water quality parameter data was tested for normality using the Shapiro–Wilk test with a p-value of less than 0.05 (Table 3). The Spearman rank correlation test was then applied to determine if any of the water quality variables were associated with changes in percentage of land use, those of soil types and geological formations along with geometric features namely, catchment area (hac), average catchment slope (%), average main channel slope (%), drainage

**Table 3**

Result of normality test for water quality parameters

| Parameter | | Wilk–Shapiro test | |
|---|---|---|---|
| | | Statistic | Significance |
| Water quality | pH | 0.99 | 0.98[a] |
| | EC (mS/m) | 0.95 | 0.09[a] |
| | Turbidity (mg/L) | 0.83 | 0.00 |
| | DO (mg/L) | 0.96 | 0.47[a] |
| | TDS (mg/L) | 0.89 | 0.01 |
| | T (C) | 0.83 | 0.00 |
| | TN (mg/L) | 0.67 | 0.00 |
| | $NO_3^-$ (mg/L) | 0.56 | 0.00 |
| | DON (mg/L) | 0.93 | 0.09[a] |
| | $NH_4^+$ (mg/L) | 0.34 | 0.00 |
| | $NO_2^-$ (mg/L) | 0.52 | 0.00 |
| | DIN (mg/L) | 0.42 | 0.00 |
| Geometrics | Elevation (m) | 0.94 | 0.12[a] |
| | Catchment area (hec) | 0.48 | 0.00 |
| | Strahler order | 0.82 | 0.00 |
| | Drainage density (m/hec) | 0.97 | 0.59[a] |
| | Main channel slope | 0.94 | 0.16[a] |
| | Catchment slope | 0.94 | 0.57[a] |
| Land covers | Urban | 0.72 | 0.00 |
| | Forest | 0.88 | 0.10[a] |
| | Agriculture | 0.89 | 0.10[a] |
| | Grassland | 0.77 | 0.00 |
| Soils | Grey lowland soil | 0.49 | 0.00 |
| | Residual regosol | 0.59 | 0.00 |
| | Brown forest soil | 0.61 | 0.00 |
| | Regosol | 0.49 | 0.00 |
| Geology | Diluvial sand | 0.26 | 0.00 |
| | Granite | 0.78 | 0.00 |
| | Rhyolite | 0.84 | 0.00 |

[a]Correlation is significant at the 0.01 level (2-tailed)

density (m/hac) and Strahler order within catchment of the wetlands (Table 4).

**Table 4**

Spearman correlation coefficient test between water quality parameters and proportion (%) of land use, catchment geometrics, and soil and geological features in the catchment of the wetlands

| Water quality parameter | Land uses | | | | Catchment geometrics | | | Soil and geology | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Urban | Forest | Agriculture | Grassland | Catchment area | Strahler order | Drainage density | Grey lowland soil | Regosol | Diluvial sand | Rhyolite |
| Ec (mS/m) | 0.67[a] | –0.62[a] | 0.40[b] | 0.06 | – | – | – | – | – | – | – |
| DO (mg/L) | –0.42[b] | 0.34 | –0.17 | –0.06 | – | – | – | – | 0.42 | – | – |
| TDS (mg/L) | 0.69[a] | –0.68[a] | 0.45[b] | 0.15 | – | – | – | – | – | 0.41[b] | |
| TN (mg/L) | 0.92[a] | –0.68[a] | 0.44[b] | –0.05 | – | – | – | – | – | – | 0.46[b] |
| DON (mg/L) | 0.65[a] | –0.43[b] | 0.20 | 0.04 | – | – | – | – | –0.44[b] | – | –0.49[b] |
| $NH_4^+$ (mg/L) | 0.47[b] | –0.55[a] | 0.56[a] | 0.03 | – | – | – | 0.60[a] | – | 0.58[a] | –0.47[b] |
| $NO_2^-$ (mg/L) | 0.50[b] | –0.39 | 0.31 | –0.09 | 0.54[a] | 0.52[a] | 0.53[a] | 0.59[a] | – | – | – |

[a] Correlation is significant at the 0.01 level (2-tailed)

[b] Correlation is significant at the 0.05 level (2-tailed)

Considering the landscape degradation within the study wetlands, they were classified into three groups in order to assess the impact of landscape degradation on the water quality of the wetlands (Fig. 2). The classification of the wetlands was conducted considering the percentage of urban areas within the catchment of each wetland. The Kruskal–Wallis test was then applied to specify whether there were significant differences among the three wetland groups in terms of the annual mean of water quality parameters (Table 5).

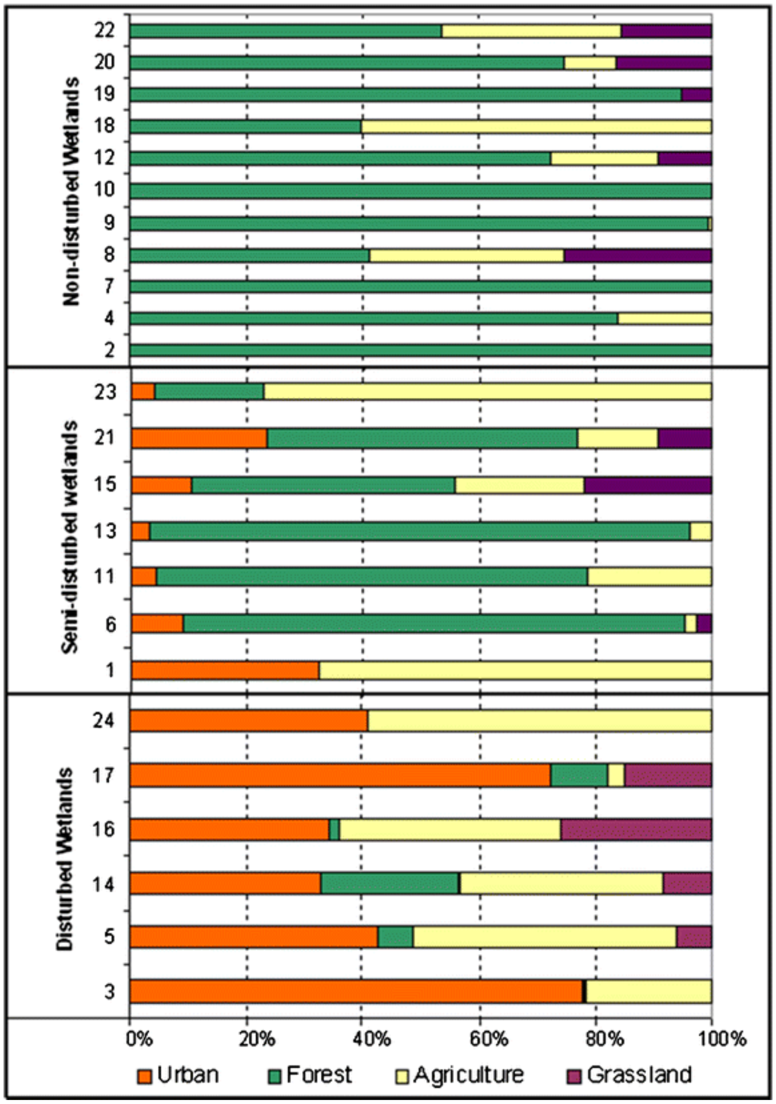


**Fig. 2**

Proportion (%) of land use in the catchment of the wetlands along with their classification into three classes: non-disturbed, semi-disturbed and disturbed wetlands

**Table 5**

Result of Kruskal–Wallis test of water quality parameters within three wetland groups

| Water quality parameter | Statistics | |
|---|---|---|
| | Adjusted H | *P* value |
| pH | 0.01 | 0.99 |
| EC (mS/m) | 10.71 | 0.00* |
| Turbidity (mg/L) | 2.83 | 0.24 |
| DO (mg/L) | 3.94 | 0.14 |
| TDS (mg/L) | 11.19 | 0.00* |
| TN (mg/L) | 19.90 | 0.00* |
| $NO_3^-$ (mg/L) | 2.49 | 0.29 |
| DON (mg/L) | 9.30 | 0.01* |
| $NH_4^+$ (mg/L) | 7.24 | 0.03** |
| $NO_2^-$ (mg/L) | 7.87 | 0.02** |
| DIN (mg/L) | 6.23 | 0.04** |

*Significant at $p < 0.01$

**Significant at $p < 0.05$

The watershed boundary of the wetlands was defined by applying 30 m DEM by the Spatial Analyst extension in ArcMap 9.3. The land use map (Amiri and Nakane *2006*) was then overlaid on the catchment boundary of the wetlands to calculate the real extent of each land use type within the catchments. The result of this calculation was then divided by the area of the catchments to determine the percentage of the catchment covered by each type of land use by Geographical Information System (ArcMap 9.3). The same procedure was also applied to calculate the proportion (%) of soil type and geological formations for each of the wetlands.

## 3 Results and Discussion

The relationship between the annual mean value of the water quality parameters of the wetlands and the proportion of land use types within the watersheds of the wetlands was examined using the Spearman correlation coefficient test ($p < 0.05$). The findings revealed that there were significant positive associations between the proportion (%) of urban areas within the watersheds of the wetlands and the EC value ($r = 0.67, p < 0.01$), TDS ($r = 0.69, p < 0.01$), TN ($r = 0.92, p < 0.01$), DON ($r = 0.6, p < 0.01$), $NH_4^+$ ($r = 0.47, p < 0.05$), $NO_2^-$ ($r = 0.50, p < 0.05$), while negative associations were observed between DO ($r = -0.42, p < 0.05$), and % of urban area in the watershed of the wetlands. This could be the result of an increase in nutrient concentrations. Accordingly, if the proportion (%) of the urban area increases within the catchment of the wetlands, the annual mean of the water quality parameters, except DO, increased in the wetlands. This is related to landscape degradation resulting from transformation of forest areas into urban areas, and to soil losses in urban areas due to improper storm water management, which can cause degradation of water quality due to soil erosion and sediment transportation (Kaste et al. *1997*; Castillo et al. *2000*). This can be expected in study areas that have experienced rapid development in the urban sector. Urban development influences wetlands through vegetation cover clear cutting, drainage, and cut and fill practices. Moreover, following urban development, significant increases can be expected in the volume of surface waters entering wetlands due to the building up of the impervious cover in the wetlands (Water and River Commission *2001*). Mouri et al. (*2011*) cited a significant relationship between the concentration of TN and the area of either urban or agricultural land. Ye et al. (*2009*) and Norton and Fisher (*2000*) also reported a negative relationship between forest areas and concentration of TN. Haidary and Nakane (*2008*) noted that depending on which type of land use is dominant within the catchment of the wetlands, the dominant type of nitrogen might be different in the out-flowing water from the wetlands. They have reported that $NH_4^+$ was the dominant form of the nitrogen in the wetlands whose catchment is covered by a high percentage of urban area, and ($NO_3^- + NO_2^-$) were sub-dominant in the wetlands whose catchments were covered by a high percentage of agricultural area.

In contrast, the percentage of forest areas in these wetlands has a significant negative relationship with EC ($r = -0.62, p < 0.01$), TDS ($r = -0.68, p < 0.01$), TN ($r = -0.68, p < 0.01$), DON ($r = -0.43, p < 0.05$), and $NH_4^+$ ($r = -0.55, p < 0.01$). These relationships suggest that forest areas play a controlling role in regulating the water quality of wetlands. Hence, if the proportion of forest areas increases within the wetlands, annual mean values of EC, TDS, TN, DON, and $NH_4^+$ would significantly decrease in the wetland water. It has been well documented that there is a positive relationship between watershed land use practices and soil erosion. In particular, the loss of forest cover is associated with increased soil erosion (Cooke and Prepas *1998*; Arnold and Gibbons *1996*) and diminished water quality (Houlahan and Findley *2004*). The findings of this study are in agreement with those of Amiri and Nakane (*2009*), who found an inverse relationship between water quality parameters such as pH, SS, *E.coli*, TN and TP and the forest area of the catchments of interest. Additionally, the existence of a high percentage of forest area within the catchment of the wetland can affect the role they play as a nutrient sink. Accordingly, Haidary and Nakane (*2008*) found that those wetlands, whose catchments were covered by a high percentage of forest area, played a sink function for nitrogen in out-flowing water from the wetlands.

Positive relationships were observed between the annual mean of EC ($r = 0.40, p < 0.05$) and TDS ($r = 0.45, p < 0.05$), TN ($r = 0.44, p < 0.05$), $NH_4^+$ ($r = 0.56, p < 0.01$) and the proportion of agricultural areas in the watershed of the wetlands. These relationships could originate from the application of chemical fertilizers and soil erosion from farmlands; the fertilizers and eroded soils are carried out by agricultural runoff into the wetland water (EPA *2002*). Ukita and Nakanishi (*1999*) reported that the potential nitrogen loss rates for agricultural areas (5.5–52.5 kg/ha/year) are 1.38 times that of natural vegetation (0.9–38.0 kg/ha/year). However, no significant relationship was observed in this study between the proportion (%) of grassland areas in the watershed of the wetlands and changes in water quality parameters and nitrogen concentrations. Our results are also confirmed by those of Crosbie and Chow-Fraser (*1999*) and Haidary and Nakane (*2008*), who found that the wetlands whose catchments were covered by agricultural area tended to be more turbid and nutrient rich compared to those that were located in forest catchments. Amiri and Nakane (*2009*) reported a direct relationship between the change in the agricultural area of a given catchment and that of suspended solid and total nitrogen.

Three catchment geometrics including catchment area of the wetland ($r = 0.54, p < 0.01$), Strahler order ($r = 0.52, p < 0.01$) and drainage density ($r = 0.53, p < 0.01$), out of five geometric features of the catchments, indicated positive significant relationship with $NO_2^-$. Accordingly, if these geometrics increase, the NO2$^-$ concentration will increase in the wetland. Regarding the Strahler order, a direct significant relationship was observed between this geometric feature of the catchment and the $NO_2^-$ concentration in the wetland. It is possible that an increase in the Strahler order is associated with a decline in the general slope of the catchment. In turn, this could occur due to an increase in the agricultural activities in a given catchment.

The result of the Spearman test revealed that a proportion (%) of grey lowland soil has a significant relationship with $NH_4^+$ ($r = 0.60, p < 0.01$), that of regosol with DO ($r = 0.42, p < 0.05$) and DON ($r = -0.44, p < 0.05$); and that of diluvial sand with TDS ($r = 0.41, p < 0.05$) and $NH_4^+$ ($r = 0.58, p < 0.01$). Moreover, rhyolite has a positive significant relationship with TN concentration ($r = 0.46, p < 0.05$) and an indirect relationship with DON ($r = 0.49, p < 0.05$) and $NH_4^+$ ($r = 0.47, p < 0.05$) in the wetland.

The Kruskal–Wallis test (Table 5) results indicate significant differences in the annual mean of TDS, TN, DON, $NH_4^+$, $NO_2^-$ and DIN within the three wetland groups with $p < 0.05$. On the other hand, the water quality parameters, namely pH, EC, turbidity, $NO_3^-$ and DO did not reveal significant differences in the annual mean between wetland groups with $p < 0.05$. The analysis of variance for different forms of nitrogen in this study suggest that all forms of nitrogen including nitrite dissolved inorganic nitrogen, dissolved organic nitrogen, and total nitrogen, except nitrate and ammonium, reveal significant differences among wetland groups.

## 4 Conclusion

Studying the relationships between the proportion of land use types and water quality parameters of wetlands in the 24 wetland sites of this study indicate that the concentration of TDS, TN, DIN, DON, $NO_2^-$, and EC have decreased along with increase in proportion of forest areas within catchment of the wetland.. Hence, such the forest component of the landscape within the catchment of the wetlands can be used to regulate the water quality of these water bodies. The findings of this study have also revealed that there were direct relationships between the proportion (%) of urban areas within the catchment of the wetlands and the annual mean of nutrients such as TN, DON, $NO_2^-$, TDS and EC in the wetland waters. Moreover, an increase in the proportion of agricultural areas increases the annual mean of $NO_2^-$ and TN in the wetland sites. Hence, urban and agricultural areas can be considered as water quality 'disturbing components' of the landscape. In comparison with the impact of urban area on the water quality, in case of ammonium, of the wetlands, the impact of agricultural areas can be expected to be more significant than that of urban areas. This can be related to the application of chemical fertilizers within agricultural areas. Hence, land use and water resources planning should consider controling the extent of agricultural and urban areas in wetlands to improve the environmental quality of the wetlands. The results of this study also indicate that degradation of the landscape within wetlands can cause the deterioration of water quality in such water bodies.

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