



The effect of water turnover time on decomposition of wild rice (*Zizania latifolia*) and nutrient dynamics in an artificial wetland system

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Abstract

The effect of different turnover time of water on the decomposition of emergent macrophyte litter (*Zizania latifolia* Griseb.) was investigated using a microcosm experiment. Microcosm treatment represented different turnover time of water; 1, 2, 4 and 8 weeks. The litterbags from each treatment were retrieved every 2 weeks until the 8th week and the water simultaneously sampled with the litterbag. The dry weight and the content of major cations in the litter, and the content of available N, P, and major cations in the water were analyzed. Dry weight loss after 8 weeks indicated the lower decay rates under the condition of short turnover time of water. Major cations from the litter and the water showed that the leached amounts of K and Mg from the litter were highest in the 2nd week and dramatically decreased from the 4th week. The dynamics of available nitrogen and phosphorus in the water showed that as the water turnover time was getting longer, the amounts of available nitrogen and phosphorus remained higher. These results suggest that wetlands with longer turnover time of water could maintain the increased nitrogen and phosphorus and no outflow of the nutrients could cause eutrophication problem.

Key words: closed system, decay rate, eutrophication, hydrological regime

INTRODUCTION

Wetlands dominated by emergent macrophytes are among the most productive ecosystems (Wetzel and Howe 1999, Gessner 2000). The emergent macrophytes serve as nutrient sink by fixing nutrients on plant during the growth period, whereas the dead biomasses do as nutrient source by releasing nutrients on surrounding during the decomposition period (Carpenter 1980). The nutrients released into the environment are reabsorbed by other organisms such as microorganisms and plants, indicating that the decomposition of plant litter regulates the nutrient cycle (Hobbie and Vitousek 2000).

The decomposition of plant litter in wetlands can be

affected by the various environmental conditions. The environmental variables such as water temperature and oxygen level affect directly the microbial activity, and the litter qualities such as C:N ratio, initial nutrient levels, and lignin content also have a significant influence on the availability of decomposed materials (Swift et al. 1979, Aber and Melillo 1980, Hodson et al. 1984, Kim 2001). In addition, the increase of nutrient availability including the increased concentration of dissolved nitrogen or phosphate can correlate positively with litter decay (Surbekropp and Chauvet 1995, Rosemond et al. 2002, Kim 2005).

<http://dx.doi.org/10.5141/ecoenv.2014.002>



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Received 10 October 2013, Accepted 03 February 2014

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Among the factors that affect litter decomposition, the water flow regimes can be a key driver in wetland ecosystem (Molles et al. 1995, Glazebrook and Robertson 1999, Andersen et al. 2003). The water flow regimes can affect nutrient circulation by determining the physical and chemical characteristics of wetland (Robertson et al. 2001, Tockner et al. 2000, Heffernan and Sponseller 2004). Hydrological factors that can cause the change of nutrient cycle include frequency, timing and duration of flooding, and water turn-over time.

Langhans and Tockner (2006) reported that the duration of inundation was the crucial variable of decay rather than the frequency. Study in experimental microcosms has shown that dry litter decomposed slower than flooded litter but decay rate of litter subjected to different duration of inundation were similar (Day 1983). The field study by Neckles and Neill (1994) showed similar result to the study in microcosms. Floodplain inundation changes riparian vegetation and species diversity (Tiegs et al. 2005). The change can affect leaf litter quality and quantity which affect further litter decomposition (Kim and Kim 2009).

In particular, water turnover time in wetlands is one of important determinants of the chemical characteristics and, thus closely associated with the retention and circulation of nutrients in wetland ecosystems. However, most of previous studies on how the hydrology affects nutrient cycling have focused on frequency and duration of inundation, and there is a paucity of study with the effects of water turnover time on the decomposition of macrophytes and nutrient cycling (Day 1983, Shim et al. 1992, Neckles and Neill 1994, Langhans and Tockner 2006).

Therefore, the present study aims to investigate how the difference of turnover time affects the decomposition of emergent macrophytes and nutrient dynamics between the litter and surroundings in artificial condition. Based on the results, the further purpose is to suggest ways of managing the hydrological regimes in order to promote nutrient retention in wetland ecosystems.

MATERIALS AND METHODS

Microcosm experiment: preparation for leaching experiment

The plant species used for being provided as litter, wild rice *Z. latifolia* Griseb. was one of typical emergent macrophytes in freshwater wetlands. The plant mostly spread along the littoral zone in East Asia (Okuda 1978). The

plant was collected from a single population in Gyeongan Stream at Gyeonggi province, Korea in July 2012.

Only the leaf blades were prepared for the litter. The leaf blades were cut into 10 cm size and air-dried at 60°C for more than 72 hours. The litterbag was 20 × 20 cm nylon bag with 1 mm mesh. Each litterbag enclosed about 7 g of litter. Four litterbags were placed into each pot microcosm (27 × 20 × 8 cm) with 3 L of distilled water and submerged under water which is the default state, because wetland ecosystem was based on the experiment. A total of 68 bags were prepared, 4 bags were used for determining initial conditions and 64 bags were used for four treatments. Four treatments representing different water turnover time, exchange of water every 1 (treatment A), 2 (treatment B), and 4 (treatment C) weeks and non-exchange of water (treatment D) until 8th week, were applied with 4 replicates for each treatment. The microcosms were transferred into growth chambers with no light for limiting the growth of green algae and the temperature of 35°C for enhancing microbial activities (O'Connell 1990, Kirschbaum 1995). All the microcosms were remained in aerobic conditions.

Litterbag retrieval and chemical analysis

The 4 litterbags from each treatment were retrieved every 2 weeks after the litterbag installation until 8th week (Table 1). The water for the nutrient analysis was simultaneously sampled with the litterbags.

The plant leaves in the sampled litterbags were weighed after drying at 60°C for more than 72 hours. Each dried sample was ground in a planetary ball mill (Pulverisette 23; Fritsch, Idar-Oberstein, Germany) and was sieved through 0.42 mm mesh of the US Standard No. 40 sieve.

The organic matter content was determined by measuring weight loss on ignition at 450°C for 4 hours in a muffle furnace (Dean 1974, Boyle 2004). To determine the concentrations of K, Ca, and Mg, 0.3 g of each sample powder was mixed with 5 mL of nitric acid and 1 mL of

Table 1. Water turnover time and litterbag retrieval of the experimental microcosms. 'W' in cells indicates the exchange of water and shaded cells indicate timing of litterbag retrieval

Treatment	Exchange of water	Weeks after installation							
		1	2	3	4	5	6	7	8
A	Every 1 week	W	W	W	W	W	W	W	W
B	Every 2 week		W		W		W		W
C	Every 4 week				W				W
D	Non-exchange								W

peroxide and acid-digested with a MARS Xpress microwave (CEM, Matthews, NC, USA). An atomic absorption spectrophotometer (AA240FS; Varian, Palo Alto, CA, USA) was used to quantify the K, Ca, and Mg in these digested solutions.

The remaining weight of the litter was expressed as a percentage of the initial sample weight. Decomposition rate, k , was derived from a single negative exponential model,

$$-k = \ln(X/X_0)/t$$

where X_0 is the initial dry weight and X is the dry weight at time t , t in days (Olson 1963).

Sampled water was filtered through glass fiber filters to remove particulate organic matter. We analyzed content of available N ($\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$), available P ($\text{PO}_4\text{-P}$), and cation such as K, Ca, and Mg in the water. $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, and $\text{PO}_4\text{-P}$ were analyzed by the hydrazine method, indophenol method, and ascorbic acid method, respectively (Kim et al. 2004). The contents of K, Ca, and Mg were measured by using an atomic absorption spectrophotometer (AA240FS; Varian).

SPSS ver. 20.0 software (SPSS, Inc., Chicago, IL, USA) was used for a one-way analysis of variance (ANOVA) and Scheffe's post hoc test at a 5% significance level for analyses.

RESULTS AND DISCUSSION

Weight loss

The dry weights of *Z. latifolia* litter significantly changed during the experimental period (Fig. 1a). In all treatments, the dry weights decreased by around 45% of their original weights in the first 2 weeks. In the last 8th week, means and standard errors (SE) of the remaining dry weight percentage in the treatment A (1 week of water turnover time), B (2 weeks), C (4 weeks), and D (non-exchange) were $47.5 \pm 0.1\%$, $38.4 \pm 0.2\%$, $32.7 \pm 0.1\%$, and $31.5 \pm 0.1\%$, respectively. The decay rate (k) of litters in the treatments A, B, C, and D were 1.33×10^{-3} , 1.71×10^{-3} , 2.00×10^{-3} , and $2.07 \times 10^{-3} \text{ day}^{-1}$, respectively. These results showed that as the water exchange cycle is shorter, the decomposition can be getting slower.

The percentage of organic matter content increased in all treatments from 92.11 ± 0.28 to the range of 93.78 ± 0.24 to 94.62 ± 0.36 during the first 2 weeks (Fig. 1b). This result indicates that leaching of the inorganic matter is likely much faster than decay of the organic mat-

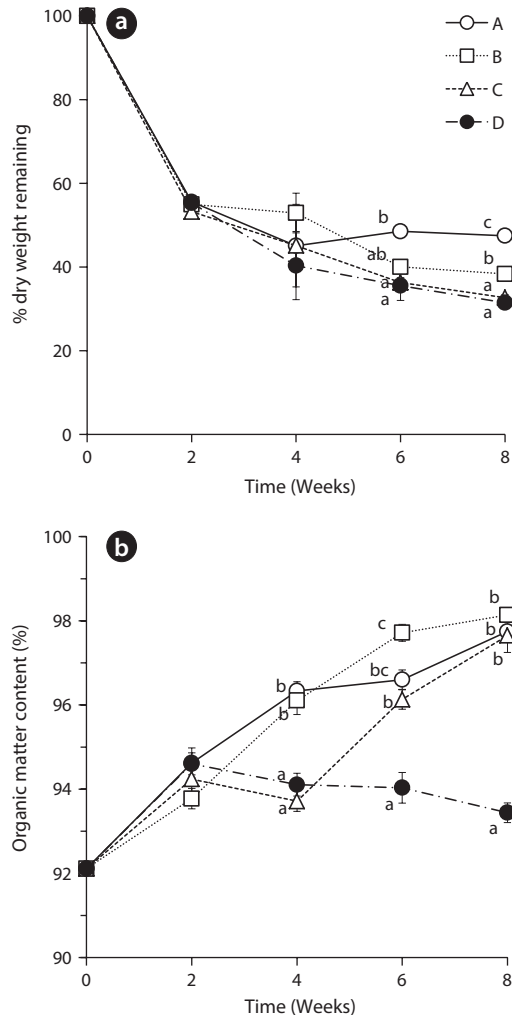


Fig. 1. Changes of dry weight (a) and organic matter content (b) in *Z. latifolia* of each treatment. Treatments, A–D, are described in Table 1. Lower case letters on the graphs mean significant different at the 5% level based on Scheffe's test. Vertical bars indicate \pm SE ($n = 4$).

ter in the beginning of the experiment (Kim and Chang 1989). During the next two weeks, the organic matter content increased in treatments A and B with the exchange of water at 2nd week but decreased in the treatment C and D without water exchange at that time. These results showed the statistically significant difference at 4th week. Also, after 2nd week, the treatment D with no water exchange showed a gradually decreasing trend of organic matter content. This result suggests that non-exchange treatment had more rapid decay of organic matters than other treatments. The slow decay of leaves in the treatments with water exchange may be due to the removal of micro-organisms and dissolved nutrients which were involved in decomposition of organic matter (Howard and Howard 1974).

Changes of nutrients

The leached rates of major cations in the leaves of *Z. latifolia* are given in Fig. 2. K in all treatments decreased sharply in the first 2 weeks (Fig. 2a). After 2 weeks, the remaining percentages of K in the treatment A, B, C, and D were 0.87 ± 0.12 , 4.31 ± 0.13 , 4.93 ± 0.26 , and 4.43 ± 0.08 , respectively. At the 8th week, the treatment D with no water exchange had the significantly higher value in the remaining percentage of K than other treatments with water exchange.

The remaining percentage of Ca decreased more steadily than that of K during the experiment period (Fig. 2b). However, the remaining percentage of Ca at the 8th week was higher than that of K. After 8 weeks, the percentages of Ca in the treatment A, B, C, and D were 17.42 ± 1.86 , 17.19 ± 3.20 , 17.25 ± 2.54 , and 26.61 ± 2.86 , respectively. These results support that K is known more labile nutrient and Ca is known more tightly held by some species (Davis and van der Valk 1978, Mun et al. 2001).

The change of remaining percentage of Mg showed a similar tendency with K (Fig. 2c). Mg in all treatments decreased sharply in 2 weeks and thus, most of Mg in *Z. latifolia* litter leached out during the period. After 8 weeks, the remaining percentages of Mg were significantly different among the treatments. The values of means and standard errors (SE) in A, B, C, and D were 2.36 ± 0.16 , 3.52 ± 0.24 , 7.46 ± 0.96 , and $14.03 \pm 1.58\%$, respectively. The results indicate that as the water was exchanged more frequently, Mg in the leaves was more leached out. Davis and van der Valk (1978) suggested that Mg and K had the similar trend of the leaching-out coincident with our results, which showed that the leaching rates of cations were in the order of $K > Mg > Ca$.

Total amount of nutrients including $NH_4\text{-N}$, $NO_3\text{-N}$, $PO_4\text{-P}$, K, Ca, and Mg in the water of the microcosms was estimated to determine the leached amount from the litter to the surrounding water (Figs. 3–5). Treatment A was shown at an interval of one week because of the weekly sampling during the water exchange, while the results for the other treatments were shown every two weeks.

The $NH_4\text{-N}$ content in water was highest in the 2nd week in all treatments except B (Fig. 3a). After the water exchange at the 4th week, it was still high in the treatment B. The treatment C and D showed a decreasing trend of $NH_4\text{-N}$ content even during the period without water exchange. This might be due to the volatilization of $NH_4\text{-N}$ to NH_3 gas at the water surface (Kirchmann and Witter 1989) or oxidation to $NO_3\text{-N}$. $NO_3\text{-N}$ content in water also was highest in the 2nd week in all treatments but less than

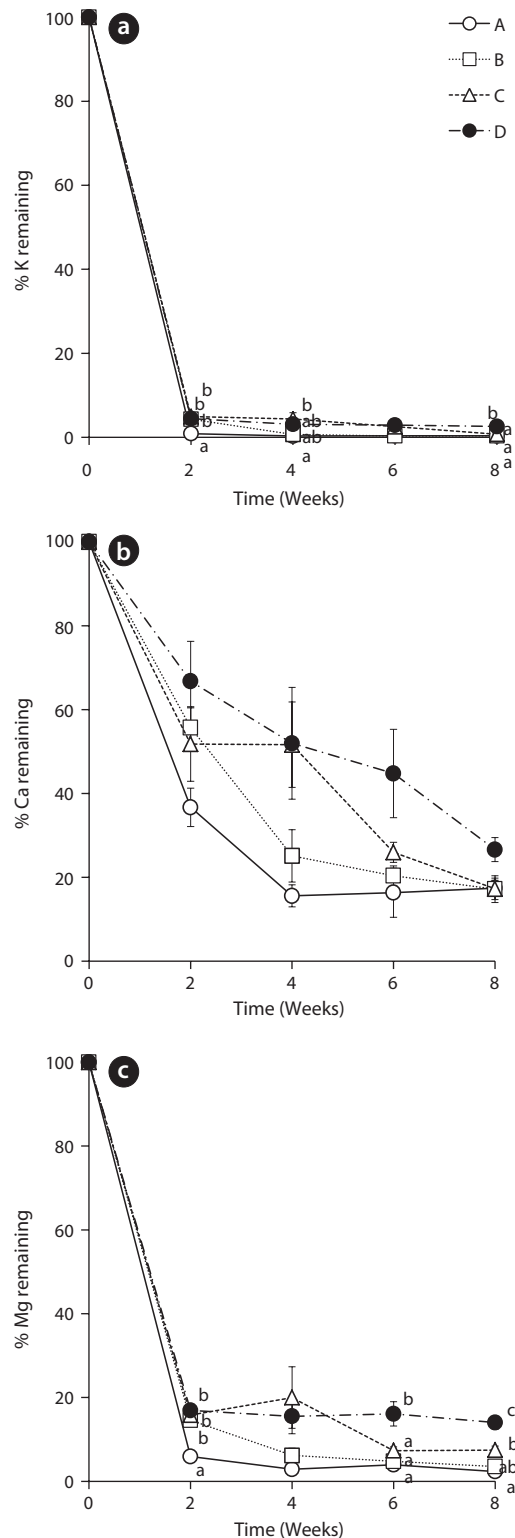


Fig. 2. Changes of percentage of remaining (a) K, (b) Ca, (c) Mg in *Z. latifolia* from each treatment. Treatments, A–D, are described in Table 1. Lower case letters on the graphs mean significant different at the 5% level based on Scheffe's test. Vertical bars indicate \pm SE (n = 4).

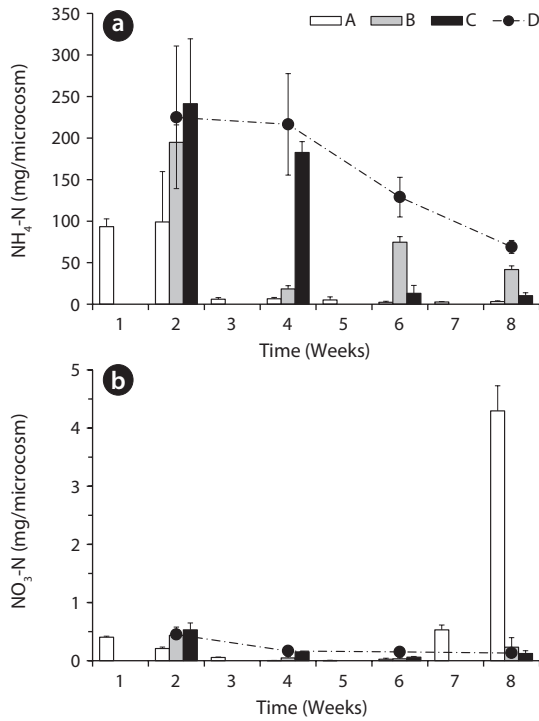


Fig. 3. Changes of available nitrogen contents in the water. (a) Changes of NH₄-N content in the water. (b) Changes of NO₃-N content in the water. Treatments, A–D, are described in Table 1. The line shows the flow of data result from non-exchange water (treatment D), the line do not draw if water exchanged between the time (treatment A–C). Vertical bars indicate ± SE (n = 4).

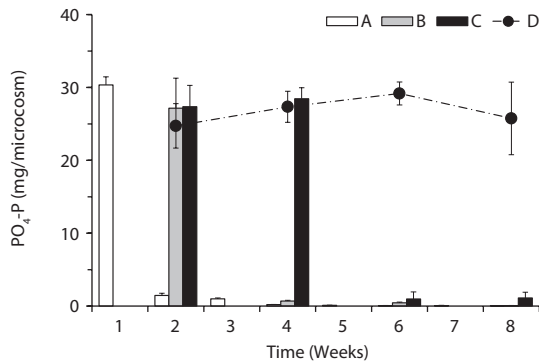


Fig. 4. Changes of PO₄-P content in the water. Treatments, A–D, are described in Table 1. The line shows the flow of data result from non-exchange water (treatment D), the line do not draw if water exchanged between the time (treatment A–C). Vertical bars indicate ± SE (n = 4).

1 mg/box (Fig. 3b). These results indicate that most of nitrogen in the litter was leached out as the form of NH₄-N rather than NO₃-N. After the 6th week, NO₃-N content in water sharply increased to 4.30 ± 0.43 mg/box in the treatment A. This might be due to the transform of NH₄-N to NO₃-N, which can be often triggered by the addition of

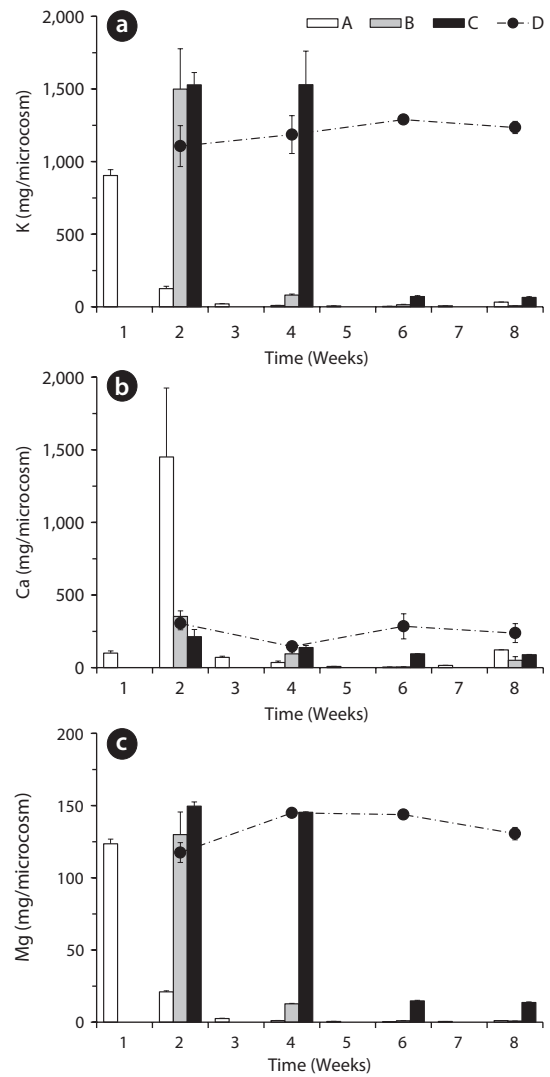


Fig. 5. Changes of major cation contents in the water. (a) Changes of K content. (b) Changes of Ca content. (c) Changes of Mg content. Treatments, A–D, are described in Table 1. The line shows the flow of data result from non-exchange water (treatment D), the line do not draw if water exchanged between the time (treatment A–C). Vertical bars indicate ± SE (n = 4).

dissolved oxygen in the exchanged water (Kirchmann and Witter 1989).

PO₄-P content in water showed relatively steady pattern in all treatments compared with available nitrogen (Fig. 4). PO₄-P content in the treatment D ranged from 24.73 ± 3.05 to 29.17 ± 1.57 mg/box during the experimental period, while the treatment A showed 30.36 ± 1.10 mg/box during the first 1 week. These results suggest that the most PO₄-P from the litter was maintained in the microcosm system unless the water containing PO₄-P would be exchanged out.

Based on the change of available nitrogen and phos-

phorus in water during the experimental period, the water turnover time can provide an additional clue to the difference of the decay rates among the treatments in Fig. 1a. As the water turnover time is getting longer, the amount of available nitrogen and phosphorus are maintained higher and this can lead to the increase of microbial activity (Suberkropp and Chauvet 1995, Rosemond et al. 2002).

The changing tendency of major cation contents in water except Ca at the 2nd week in the treatment A was similar to that of PO₄-P content in water (Fig. 5). These results indicate that a large amount of cations could be leached from the litter at the early period and remain in the water as measurable ion-forms in the treatment with the slow exchange of the water because of less outflow and transform of the cations. In particular, K showed relatively very high content during the first 2 weeks and this result supports the fast decrease of K in the litter (Fig. 2a) and K has high leached rate at the beginning of the litter decay.

The present study showed that the litter decomposition is decreasing with the shorter turnover time of water in the closed system. In addition, most of the nutrients in the litter were leached to water during the first 2 weeks and the leached nutrients were removed from the microcosm by the water exchange representing the turnover time of water in the field. The microcosm without the turnover of water has a consistent influx of nutrients, and the increased nitrogen and phosphorus and no outflow of the nutrients can cause the eutrophication problem. Therefore, our study suggests that artificial wetlands with very slow water turnover time may have higher potential of eutrophication and the fast water turnover time has to be applied in order to decrease the decay rate of litters and to remove the accumulative nutrients for preventing the eutrophication of water.

ACKNOWLEDGMENTS

This work was supported by the National Research Foundation of Korea (NRF) grant funded by the Korea government (NRF-2012R1A1A2001007 and No. 2011-0028564) and by the Center for Aquatic Ecosystem Restoration (CAER) of the Eco-STAR project from the Ministry of Environment (MOE), Republic of Korea (EW33-08-12).

LITERATURE CITED

Aber JD, Melillo JM. 1980. Litter decomposition: measuring relative contributions of organic matter and nitrogen to

- forest soils. *Can J Bot* 58: 416-421.
- Andersen DC, Nelson SM, Binkley D. 2003. Flood flows, leaf breakdown, and plant-available nitrogen on a dryland river floodplain. *Wetlands* 23: 180-189.
- Boyle J. 2004. A comparison of two methods for estimating the organic matter content of sediments. *J Paleolimnol* 31: 125-127.
- Carpenter SR. 1980. Enrichment of Lake Wingra, Wisconsin, by submersed macrophyte decay. *Ecology* 61: 1145-1155.
- Davis CB, van der Valk AG. 1978. The decomposition of standing and fallen litter of *Typha glauca* and *Scirpus fluviatilis*. *Can J Bot* 56: 662-675.
- Day FP Jr. 1983. Effects of flooding on leaf litter decomposition in microcosms. *Oecologia* 56: 180-184.
- Dean WE Jr. 1974. Determination of carbonates and organic matter in calcareous sediments and sedimentary rocks by loss on ignition: comparison with other methods. *J Sediment Petrol* 44: 242-248.
- Gessner MO. 2000. Breakdown and nutrient dynamics of submerged *Phragmites* shoots in the littoral zone of a temperate hardwater lake. *Aquat Bot* 66: 9-20.
- Glazebrook HS, Robertson AI. 1999. The effect of flooding and flood timing on leaf litter breakdown rates and nutrient dynamics in a river red gum (*Eucalyptus camaldulensis*) forest. *Aust J Ecol* 24: 625-635.
- Heffernan JB, Sponseller RA. 2004. Nutrient mobilization and processing in Sonoran desert riparian soils following artificial re-wetting. *Biogeochemistry* 70: 117-134.
- Hobbie SE, Vitousek PM. 2000. Nutrient limitation of decomposition in Hawaiian forests. *Ecology* 81: 1867-1877.
- Hodson RE, Christian RR, Maccubbin AE. 1984. Lignocellulose and lignin in the salt marsh grass *Spartina alterniflora*: initial concentrations and short-term, post-depositional changes in detrital matter. *Mar Biol* 81: 1-7.
- Howard PJA, Howard DM. 1974. Microbial decomposition of tree and shrub leaf litter, 1: Weight loss and chemical composition of decomposing litter. *Oikos* 25: 341-352.
- Kim JG. 2001. Decomposition of *Carex* and *Nuphar* plants in a subalpine marsh. *J Plant Biol* 44: 73-80.
- Kim JG. 2005. Nutrient dynamics in montane wetlands, emphasizing the relationship between cellulose decomposition and water chemistry. *J Korean Wetl Soc* 7: 33-42.
- Kim JG, Chang NK. 1989. Litter production and decomposition in the *Pinus rigida* plantation in Mt. Kwan-ak. *Korean J Ecol* 12: 9-20. (in Korean with English abstract)
- Kim JG, Park JH, Choi BJ, Shim JH, Kwon GJ, Lee BA, Lee YW, Ju EJ. 2004. *Method in Ecology*. Bomundang, Seoul. (in Korean)
- Kim S, Kim JG. 2009. *Humulus japonicus* accelerates the de-

- composition of *Miscanthus sacchariflorus* and *Phragmites australis* in a floodplain. *J Plant Biol* 52: 466-474.
- Kirchmann H, Witter E. 1989. Ammonia volatilization during aerobic and anaerobic manure decomposition. *Plant Soil* 115: 35-41.
- Kirschbaum MUF. 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biol Biochem* 27: 753-760.
- Langhans SD, Tockner K. 2006. The role of timing, duration, and frequency of inundation in controlling leaf litter decomposition in a river-floodplain ecosystem (Tagliamento, northeastern Italy). *Oecologia* 147: 501-509.
- Molles MC Jr, Crawford CS, Ellis LM. 1995. Effects of an experimental flood on litter dynamics in the middle Rio Grande riparian ecosystem. *Reg Riv Res Manage* 11: 275-281.
- Mun HT, Namgung J, Kim JH. 2001. Decay rate and changes of nutrients during the decomposition of *Zizania latifolia*. *Korean J Ecol* 24: 81-85.
- Neckles HA, Neill C. 1994. Hydrologic control of litter decomposition in seasonally flooded prairie marshes. *Hydrobiologia* 286: 155-165.
- O'Connell AM. 1990. Microbial decomposition (respiration) of litter in eucalypt forests of south-western Australia: an empirical model based on laboratory incubations. *Soil Biol Biochem* 22: 153-160.
- Okuda S. 1978. Plant sociological studies on the vegetation of the alluvial plains in the Kanto district, central Japan. *Bull Inst Environ Sci Tech Yokohama Natl Univ* 4: 43-112.
- Olson JS. 1963. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* 44: 322-331.
- Robertson AI, Bacon P, Heagney G. 2001. The responses of floodplain primary production to flood frequency and timing. *J Appl Ecol* 38: 126-136.
- Rosemond, AD, Pringle CM, Ramirez A, Paul MJ, Meyer JL. 2002. Landscape variation in phosphorus concentration and effects on detritus-based tropical streams. *Limnol Oceanogr* 47: 278-289.
- Shim KC, Lee SK, Lee JE, Chang NK. 1992. Study of initial decomposition of the litters of *Platanus orientalis* in stream ecosystem. *Korean J Ecol* 15: 387-394. (in Korean)
- Suberkropp K, Chauvet E. 1995. Regulation of leaf breakdown by fungi in streams: influences of water chemistry. *Ecology* 76: 1433-1445.
- Swift MJ, Heal OW, Anderson JM. 1979. Decomposition in Terrestrial Ecosystems. *Studies in Ecology*, Vol. 5. University of California Press, Berkeley, CA.
- Tiegs SD, O'Leary JF, Pohl MM, Munill CL. 2005. Flood disturbance and riparian species diversity on the Colorado River Delta. *Biodivers Conserv* 14: 1175-1194.
- Tockner K, Malard F, Ward JV. 2000. An extension of the flood pulse concept. *Hydrol Process* 14: 2861-2883.
- Wetzel RG, Howe MJ. 1999. High production in a herbaceous perennial plant achieved by continuous growth and synchronized population dynamics. *Aquat Bot* 64: 111-129.