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**Using SWAT model to predict water flow, sediment and
nutrients loads in Shibetsu river watershed, eastern
Hokkaido, Japan**

**(SWAT モデルによる北海道東部の標津川流域における水、土砂、
養分流出予測)**

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PH.D. Dissertation

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1. General Introduction

Water use trends in the world indicate that many countries will soon face a crisis in both water quantity and water quality unless current policies and practices are amended (Gleick, 1998). Agricultural operations including animal and crop farming are important contributors to the depletion of water resource quality and quantity. For example, in the United States, agriculture represents approximately 40% of total U.S. water demand, and irrigation is the largest consumptive water use (Covalla et al., 2001). Agricultural operations have been identified by the Environmental Protection Agency as the leading source of water quality impairment to rivers and lakes (Covalla et al., 2001). In China, the water resources are also affected by both severe water quantity shortages and severe water quality pollution due to agriculture management (Jiang, 2009; Liang et al., 2004). A growing population and rapid economic development as well as lax environmental oversight have increased water demand and pollution. In Japan, high contaminant levels of pesticides have been monitored in the irrigation canals of paddy field (Li and Morioka, 2000). Also, high concentration of nitrate (NO_3^- -N) has been monitored in streams of the dairy farming area (Woli et al., 2004). Hojito (1998) also indicated that river water became significantly polluted by N while flowing through dairy farming areas in Japan.

Nonpoint source pollution (NPS) is the leading remaining cause of water quality problems. NPS pollution occurs when rainfall, snowmelt, or irrigation runs over land or through the ground, picks up pollutants, and deposits them into rivers, lakes, and coastal waters or introduces them into ground water. Imagine the path taken by a drop of rain from the time it hits the ground to when it reaches a river, ground water, or the ocean. Any pollutant the water picks up on its journey can become part of the NPS problem. NPS pollution also includes adverse changes to the vegetation, shape, and flow of streams and other aquatic systems. Common NPS pollutants include sediment, nutrients pesticides, pathogens (bacteria and viruses), salts, oil, grease, toxic chemicals, and heavy metals (Liu et al., 2005; Basnyat et al., 2000). The most common NPS pollutants are sediment and nutrient which wash into water bodies from agricultural

land (Nikolaidis et al., 1998). Beach closures, destroyed habitat, unsafe drinking water, fish kills, and many other severe environmental and human health problems result from NPS pollutants. The pollutants also ruin the beauty of healthy, clean water habitats. Restoring and protecting the areas damaged by NPS pollutants usually demand high costs (Carpentier et al., 1998).

The watershed modeling approaches have the capacity to analyze the quantity and quality of water resources, and identify existing and potential watershed stressors or pollutants from NPS (Muttiah and Wurbs, 2002). A set of process-based simulation approach has been used for understanding NPS pollutants transportation processes at watershed scale over a long period time. Agricultural Non-Point Source model (AGNPS) (Young et al., 1987); the Environmental Management Support System (EMSS) (Vertessey et al., 2001; Watson et al., 2001); the Hydrologic Simulation Program, Fortran (HSPF) (Johanson et al., 1980); the Integrated Water Quantity and Quality Model (IQQM) (DLWC, 1995); the Simulator for Water Resources in Rural Basins (SWRRB) (USEPA, 1994) and the Soil and Water Assessment Tool (SWAT) (Neitsch et al., 2005) are among them (Polyakov et al., 2007). One such model that is recommended by US Environmental Protection Agency (EPA) is the SWAT model. The SWAT model has the ability to adequately simulate hydrology and the sediment, nutrient and pesticide losses under different management conditions in agricultural watersheds. Major components of the SWAT model include hydrology, weather, erosion, soil temperature, crop growth, nutrients, pesticides and agricultural management practices (Neitsch et al., 2005).

Hokkaido is known as Japan's primary dairy farming area. There are large amount of fertilizer and manure application in this area. About 93% of livestock excreta are used for manure (Hokkaido Government, 1996). The applied fertilizer and manure include large amount of nitrogen, phosphorous and potassium. These nutrients might become NPS pollutants and cause water quality problems. In the previous studies conducted in Shibetsu river watershed (SRW) in eastern Hokkaido, Hayakawa et al. (2009) and Woli et al. (2002; 2004) reported that NPS have caused water quality problems in the SRW,

NO_3^- -N concentrations in the stream water were significantly correlated with the proportion of upland fields in the drainage basins in SRW. Jiang et al. (2010) reported that sediment was also a major NPS pollutant in SRW, and particulate organic nitrogen concentrations in stream water were significantly correlated with the sediment concentration. Therefore, the sediment associated particulate organic nitrogen and particulate organic phosphorous are NPS pollutants. The water quality of SRW could also be impacted by other nutrient pollutants from crop lands. Similar with NO_3^- -N, potassium is also easily soluble in water, therefore, the NO_3^- -N pollution in SRW indicates that potassium might also loss to streams and influence water quality. Permissible limit of potassium concentration of 12 mg L^{-1} for drinking water was recommended by WHO (1984) (Taouil et al., 2013). The increased transport of nutrients from the basin could continuously contribute to groundwater and surface water pollutions (Rabalais et al., 1996). A better understanding of the type, extent and sources of pollutant loadings, and the effects of agriculture management practices to mitigate water quality problems is needed in the SRW. Thus, it is imperative to conduct research that can contribute towards understanding and mitigating the contaminant loads of sediment, sediment bound particulate nutrients and water soluble nutrients from the watersheds. For this purpose, the Soil and Water Assessment Tool (SWAT) model developed by USDA-ARS was selected to study the SRW watershed. Evaluation of the hydrological performance needs to be performed prior to attempting any water quality model simulations. Stream flow was generated from snowmelt in spring and from rainfall in summer and autumn in SRW. The contributions of this study will be:

Understand the snowmelt and rainfall seasons hydrology in Shibetsu river watershed using the SWAT model.

Understand the sediment and associated particulate nutrients loads in Shibetsu river watershed using the SWAT model.

Development and application of SWAT model for understanding potassium load in Shibetsu river watershed.

This dissertation is divided into seven chapters. The current chapter (Chapter 1) is a

general introduction to the study. The Chapter 2 provides a background and a review of the relevant literature on research problems. Chapter 3 describes a general introduction of Shibetsu river watershed, introduction of monitoring data of sediment and nutrient export, and introduction, application and development of SWAT model to the monitoring data. The SWAT model simulation of hydrology is dealt with in Chapter 4. The Chapter 5 reports on the SWAT model simulation of sediment, particulate organic nitrogen (PON) and particulate organic phosphorous (POP) exports. Chapter 6 mentions modifying SWAT model for simulation of potassium load. Finally, a general discussion on the results and the connection of each chapter of this study is presented in Chapter 7.

2. Literature review

2.1 Sediment and nutrients transport and nonpoint source pollution

Pollution of water resource is a matter of global concern at present (Gleick, 1993). As nations make efforts to correct abuses to their water resources, there is a need to determine the causes of water quality degradation and to quantify pollution contributions from different sources as well as to recommend control measures (Zimmerman et al., 2008). Traditionally, urban and industrial polluters are identified as point sources, while agricultural polluters are identified as nonpoint sources. Non-point source pollutants, irrespective of source, are transported overland and through the soil by the rainwater and the melting snow. These pollutants ultimately find their way into groundwater, wetlands, rivers and lakes and finally to oceans in the form of sediment and chemical loads carried by rivers. Despite several decades of research on management practices in agriculture to minimize water pollution, agriculture production remains a major diffuse source of water pollutants and water quality degradation throughout the world (Shortle et al., 2012). Nutrient loading owing to intensive agricultural practices around the world (Figure 2.1) typically occurs through diffuse or ‘non-point’ sources, which are hard to tackle technologically (Verhoeven et al., 2006). Quantification and control of non-point source (NPS) pollution remain relatively difficult tasks as compared to those of “point” sources of pollution. The concern for agricultural diffuse pollution sources in integrated water quality management has therefore grown recently throughout the World. Such sources are more critical in developing countries.

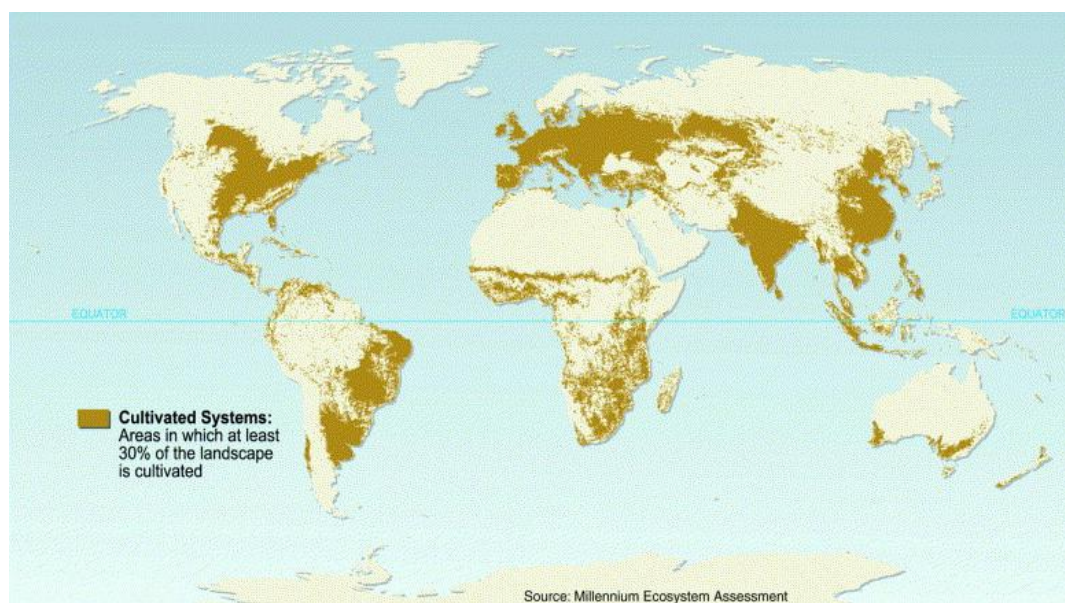


Figure 2.1 Cultivated areas of the world. Brown regions indicate areas in which at least 30% of the landscape is cultivated (Verhoeven et al., 2006)

Surface and groundwater quality degradation due to agricultural practices can be categorized as follows: a) increased erosion and soil loss due to agricultural practices; b) chemical pollution by fertilizer and pesticides and c) pollution from animal operations. Several studies have pointed out the significance of water pollution commonly produced by agricultural operations and pollutants such as sediment, nitrogen and phosphorus, and restate the fact that achieving the desired standards of water quality cannot be done through controlling the point source only (Ribaudó, 2009). Emphasizing the contributing role of agriculture to water pollution, Ribaudó et al. (2008) noted that the complete elimination of nitrogen point sources across the United States would reduce the total nitrogen emissions by only 10%. Nitrogen pollutant from NPS was the main water pollution source due to that 71% of the US crop land is located in watersheds (Ribaudó, 2009). Some studies suggest that over 70% of sediment load in rivers comes from agricultural land in the UK (Rickson, 2014). Potassium leaching from soil to groundwater can affect groundwater quality and suitability for consumption, irrigation and industrial activities. Exchangeable potassium is a mobile ion in soils and consequently significant amounts can be lost with water movement. Potassium losses in the clay soil were more than twice those in the sand soil

because of the development of preferential flow in the clay soil (Alfaro et al. 2004). Potassium can also easily leach from sandy soils than from clayey soils because sandy soils do not contain enough clay to hold the potassium (Malavolta 1985; Dassannayake 1990). As a consequence, potassium concentration might increase to above the permissible limits (12 mg L^{-1}) for drinking water recommended by WHO (1984) (Taouil et al., 2013).

FAO (2011) report presented the world nitrogen, phosphate and potassium fertilizer medium-term supply and demand projections for the period 2011-2015. Consumption of fertilizer nutrients increased significantly in 2010 and was expected to grow in a stabilized way during the following years. World demand for total fertilizer nutrients was estimated to grow at 2.0 percent per year from 2011 to 2015 (Figure 2.2) The demand for nitrogen, phosphate, and potassium was forecast to grow annually by 1.7, 1.9, and 3.1 percent, respectively, during the period. Global total nutrient production rose significantly in 2010 keeping pace with world consumption. The world potential nitrogen balance as a percentage of global total demand was expected to remain between 3 to 5 percent between 2011 and 2013 and was likely to increase to 7 percent in 2014 and 10 percent in 2015. The potential phosphate balance was likely to grow from 3 percent in 2011 to 6 per cent by 2015. The potential potassium balance as a percentage of global total demand was expected to rise from 24 percent in 2011 to a high level of 44 percent in 2015.

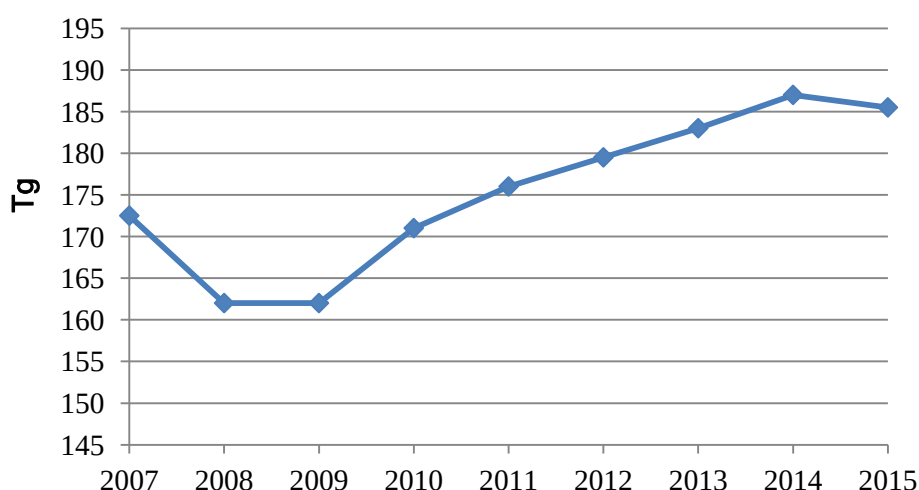


Figure 2.2 World demand for total fertilizer nutrients ($\text{N}+\text{P}_2\text{O}_5+\text{K}_2\text{O}$) from 2011 to 2015

Sediment control measures that have been assessed include cover crops, buffer strips,

no or low till, and the use of polymers (Rickson, 2014). Methods for controlling of nitrogen and potassium cycling include agricultural input (commercial fertilizer and animal sources) reduction, improved irrigation management, and cover cropping in winter months (Trachtenberg and Ogg, 1994). Since most phosphorus from agricultural lands is transported with sediment, sediment control has been the major focus for phosphorus control. Furthermore, because phosphorus binds to the soil, phosphorus has not been a groundwater concern. Thus, the majority of research on phosphorus management has focused on the control of sediments (Parker, 2000). Over time, the industrialized countries have shifted their attention from water pollution created by point sources towards the water pollution created by agricultural runoff (Olmstead, 2010). In the past, environmental studies were often designed to determine the nutrient concentrations in runoff sediments in order to quantify the outflows of the ecosystem's nutrient budget for preserving agricultural productivity and diminishing non-point pollution (Arhonditsis et al., 2000). Recently, with the increase uses of nitrogen, phosphate and potassium fertilizer, environmental consequences of agricultural practices need to be further studied for agriculture and environment sustainability based on a well understanding of water flow, sediment and nutrient transport mechanisms.

2.2 Watershed water flow, sediment and nutrients transport mechanism

The watershed scale hydrologic balance component encompasses all of the key processes that occur in the hydrologic cycle. Initially, incoming precipitation can be intercepted by plant canopies, and then precipitation, snowmelt water, and/or irrigation input are partitioned between surface runoff and infiltration. Infiltrated water can be stored in the soil profile, percolate vertically to groundwater, be lost via evapotranspiration, or routed laterally in subsurface or tile drainage flow. Together with surface runoff, return flow from groundwater and lateral subsurface flow enter into stream channels (Sklash and Farvolden, 1979; Arnold et al., 1993; Sophocleous, 2002).

The process of erosion can be described in three stages: detachment, transport and

deposition. Detachment of sediment from the soil surface was originally considered to be exclusively the result of raindrop impact (Hudson, 1975), although the importance of overland flow as an erosive agent has now been recognised. Rainfall detachment is caused by the locally intense shear stresses generated at the soil surface by raindrop impact (Loch and Silburn, 1996). Likewise, overland flow causes a shear stress to the soil surface which, if it exceeds the cohesive strength of the soil, termed the critical shear stress, results in sediment detachment. In different situations, the major processes leading to sediment detachment will differ. There are four main types of erosion processes: sheet, rill, gully and in-stream erosion. Sheet erosion refers to the uniform detachment and removal of soil, or sediment particles from the soil surface by overland flow or raindrop impact evenly distributed across a slope (Hairsine and Rose, 1992a). Rill erosion is generally small erosion features which occurs when water moving over the soil surface flows along preferential pathways forming an easily recognisable channel (Hairsine and Rose, 1992b; Rose, 1993; Loch and Silburn, 1996). Gully erosion describes channels of concentrated flow that are too deep to be obliterated by cultivation (Rose, 1993; Loch and Silburn, 1996). Gully development is considered to be controlled by thresholds, as with rills, although these thresholds have been related to slope and catchment area rather than flow erosivities (Loch and Silburn, 1996). In-stream erosion involves the direct removal of sediment from stream banks (lateral erosion) or the stream bed. These erosion types do not necessarily occur in isolation from one another. They are influenced by the landscape factors as well as rainfall characteristics (Loch and Silburn, 1996). In a catchment scale modelling exercise, most erosion models tend to predict erosion for one of these erosion types, or at most a couple, this raises the possibility that in certain areas of the catchment the processes considered by the model being used are not truly representative of the processes actually occurring in the catchment. Water pollution by sediment has two major dimensions i.e. physical dimension (top soil loss and land degradation) and chemical dimension (a primary carrier of adsorbed chemicals, which are transported into the aquatic system). Control of agricultural pollution should usually begin with preventive

measures to control erosion and sediment runoff.

The nutrient transport component uses the same division of a watershed as the hydrology and sediment components described above, in which the watershed is divided into representative overland and channel flow elements. Similar to the hydrologic and sediment simulations, nutrient transport processes are simulated in each of these elements, in a unit width for overlands, and in the entire cross section for channels. Chemicals at the soil surface can be transferred to runoff in solution form through mixing of rainwater with soil solution, dissolution of chemicals present in the solid form, desorption of chemicals adsorbed to the soil, or adsorption of chemicals to eroded soil or sediment (Baily et al., 1974). Contamination of runoff occurs only when water ponding and initiation of runoff occur. Prior to initiation of runoff, chemicals are mixed with infiltrating rainwater and migrate downward into the deeper soil (Ingram and Woolhiser, 1980). When water ponding occurs and runoff begins, part of the rainwater becomes runoff and another part infiltrates soil and finally recharges groundwater and results in groundwater contamination (Valiela et al., 1999). Pollution of stream water happens when the nutrient in surface runoff, eroded sediment, and groundwater enters into streams (Seitzinger et al., 2005).

2.3 Methodologies of water flow, sediment and nutrients transport

2.3.1 Field studies

Field study of watershed hydrology concerns with questions of water flow paths and components of the water balance include precipitation, streamflow, soil moisture, evaporation and transpiration, water storage and infiltration. Field studies of watershed sediment and nutrients transport concern with questions of source, transport paths and critical contribution areas.

To investigate the watershed scale sediment and nutrients transport, field studies on watershed monitoring network were conducted to collect hydrologic, sediment and nutrient data for selected watersheds for finding scientific questions and improving the scientific understanding. The field studies including short or long-term hydrological and water quality monitoring are helpful in understanding sediment and nutrients

transport mechanisms (e.g. Spalding and Exner, 1993; Schulz, 2004; Jiang et al., 2011). In addition, to understand the water flow paths, nutrients sources and pathways during baseflow and storm flow periods, valuable insight has been gained by using isotopes ($\delta^{15}\text{N-NO}_3^-$ and $\delta^{18}\text{O-NO}_3^-$) or natural chemical tracer (e.g. Si, Ca, K, Na, and Cl) to the urban, agricultural or forest watersheds (Kendall, 1998; Kendall et al., 2007; McGuire and McDonnell et al., 2007; Burns et al., 2009).

2.3.2 Watershed-scale hydrological and non-point source pollution models

A wide range of models exists for use in simulating sediment transport and associated pollutant transport, including empirical (or statistical) models, conceptual models and physics based models. Empirical and statistical models are based primarily on the analysis of observations and seek to characterise response from these data (Wheater et al., 1993). The computational and data requirements for such models are usually less than for physics-based models, often being capable of being supported by coarse measurements. Conceptual models usually incorporate the underlying transfer mechanisms of runoff, sediment and nutrients generation in their structure, representing flow paths in the catchment as a series of storages, each requiring some characterisation of its dynamic behaviour (Sorooshian, 1991). Traditionally, conceptual models lump representative processes over the scale at which outputs are simulated (Wheater et al., 1993). Recently developed conceptual models have provided outputs in a spatially distributed manner. Beck (1987) noted that conceptual models play an intermediary role between empirical and physics-based models. Whilst conceptual models tend to be aggregated, they still reflect the hypotheses about the processes governing system behaviour. However, empirical models make no inferences as to the processes at work, instead relying on observed or stochastic relationships between the causal variables and modelled output. Therefore, the distributed manner of conceptual models make themselves distinguishes from empirical models.

The Soil Conservation Service (SCS) curve number runoff equation is basically an empirical model which came into common use in the 1950s (SCS, 1972). It is the product of more than 20 years of studies involving rainfall runoff relationships from

small rural watersheds. The model was developed to provide a consistent basis for estimating the amounts of runoff under varying land use and soil types (USDA SCS, 1972; King et al., 1999). No other rainfall-runoff model has been used as successfully or as often on ungauged rangeland catchments as the curve number method. In the curve number method, the curve number varies non-linearly with the soil moisture content. The curve number drops as the soil approaches to wilting point and increases to 100 as the soil approaches to saturation. A major limitation of the curve number method is that rainfall intensity and duration are not considered, only total rainfall volume. Five different options are provided for estimating potential evaporation: Hargreaves (Hargreaves and Samani, 1985), Penman (1948), Priestley-Taylor (Priestley and Taylor, 1972), Penman-Monteith (Monteith, 1965), and Baier-Robertson (Baier and Robertson, 1965). The Priestly-Taylor method requires solar radiation and air temperature as input, while the Hargreaves and Baier-Robertson methods require only air temperature. The Baier-Robertson method was developed in Canada and can provide more accurate potential evaporation estimates for colder climate conditions. Evaporation from soils and plants are calculated separately, as described by Ritchie (1972). The Penman and Penman-Monteith methods among the five methods are the most data intensive, requiring solar radiation, air temperature, wind speed, and relative humidity as input.

The Universal Soil Loss Equation (USLE) is a soil erosion model used widely within the United States and worldwide (Wischmeier and Smith, 1978). Since the model was developed in the 1970s by the USDA, the model has undergone much research and a number of modifications. These include the Onstad-Foster (AOF) modification of the USLE (Onstad and Foster, 1975); the Modified USLE (MUSLE) method (Williams, 1975); the Revised USLE (RUSLE) method (Renard et al., 1997); and RUSLE2 (USDA Agricultural Research Service, 2005).

Catchment water balance budgets are a valuable tool to assess hydrology components, including precipitation, streamflow, soil moisture, evaporation and transpiration, water storage and infiltration (Jiang et al., 2011). Catchment nutrient

budgets are a valuable tool to assess relative magnitude of nutrient inputs (fertilizer, atmospheric deposition, plant fixation, and import as food and feed) and predict nutrient losses (export as food and feed, riverine and atmosphere losses) (e.g., Howarth et al., 1996; Jordan and Weller, 1996; Boyer et al., 2002; David and Gentry, 2000; Hayakawa et al., 2009; Hatano et al., 2005). However, such studies cannot quantify detailed transfer processes of runoff, sediment and nutrients because the budgets are usually at annual time-step. Also, such studies contain considerable uncertainty and potential bias, because estimates of nutrient sources, losses, and the net nutrient input balance can vary widely depending on the definition of the system boundary, the assumptions and approximations used to estimate nutrient flows, and on the quantity and quality of available data over space and time (Boyer et al., 2002; David and Gentry, 2000; Howarth et al., 1996).

Commonly used watershed scale hydrologic, sediment and nutrients transport conceptual models include Agricultural Non-Point Source model (AGNPS) (Young et al., 1987); the Environmental Management Support System (EMSS) (Vertessey et al., 2001; Watson et al., 2001); the Hydrologic Simulation Program, Fortran (HSPF) (Johanson et al., 1980); the Integrated Water Quantity and Quality Model (IQQM) (DLWC, 1995); the Simulator for Water Resources in Rural Basins (SWRRB) (USEPA, 1994) and the Soil and Water Assessment Tool (SWAT) (Neitsch et al., 2005). These conceptual models are likely to mix with empirical models. While the rainfall-runoff component of a water quality model is conceptual, empirical relationships are used to model hydrology or erosion. For example, SCS curve number method has been incorporated into many catchment scale erosion and sediment transport modelling applications, e.g. AGNPS and SWAT. Although developed for application to small hillslopes, the USLE and its derivatives have been incorporated into many catchment scale erosion and sediment transport modelling applications, e.g. AGNPS, LASCAM and SWAT. Empirical models are generally based on the assumption of stationarity; that is, it is assumed that underlying conditions are assumed to be unchanged for the duration of the study period. Empirical models also tend not to be event-responsive,

ignoring the processes of rainfall-runoff in the catchment being modelled. This assumption limits the potential applicability of such models for predicting the effects of catchment change. Among these models, SWAT model has experienced continuous evolution and been used worldwide for analyzing a wide variety of environmental problems (Gassman et al., 2007).

2.4 SWAT model

2.4.1 Model development

The SWAT model is developed by USDA Agricultural Research Service (ARS) and the continuous model evolution spans a period of roughly 30 years. Early origins of SWAT can be traced to previously developed USDA-ARS models (Figure 2.3) including the modules simulating runoff, chemicals, and erosion from Agricultural Management Systems (CREAMS) model (Knisel, 1980), the Groundwater Loading Effects on Agricultural Management Systems (GLEAMS) model (Leonard et al., 1987), and the Erosion Productivity Impact Calculator (EPIC) model (Williams, 1990). The current SWAT model is a direct descendant of the Simulator for Water Resources in Rural Basins (SWRRB) model (Arnold and Williams, 1987), which was designed to simulate management impacts on water and sediment movement for ungauged rural basins across the U.S.

Development of SWRRB began in the early 1980s with modification of the daily rainfall hydrology model from CREAMS. A major enhancement was the expansion of surface runoff and other computations for up to ten subbasins, as opposed to a single field, to predict basin water yield. Other enhancements included an improved peak runoff rate method, calculation of transmission losses, and the addition of several new components: groundwater return flow (Arnold and Allen, 1993), reservoir storage, the EPIC crop growth submodel, a weather generator, and sediment transport. Further modifications of SWRRB in the late 1980s included the incorporation of the GLEAMS pesticide fate component, optional USDA-SCS technology for estimating peak runoff rates, and newly developed sediment yield equations. These modifications extended the model's capability to deal with a wide variety of watershed water quality

management problems.

In order to support an assessment of the downstream impact of water management, flows output from multiple SWRRB runs were routed through channels and reservoirs in Routing Outputs to Outlet model (ROTO) (Arnold et al., 1995). This methodology overcame the SWRRB limitation of allowing only ten subbasins; however, the input and output of multiple SWRRB files was cumbersome and required considerable computer storage. To overcome the awkwardness of this arrangement, SWRRB and ROTO were merged into the single SWAT model (Figure 2.3). Key enhancements for previous versions of the SWAT model also included the incorporation of in-stream kinetic routines from the QUAL2E model (Brown and Barnwell, 1987), as shown in Figure 2.3. Documentation for some previous versions of the model is available at the SWAT web site (<http://swat.tamu.edu/>). Detailed theoretical documentation and a user's manual for the latest version of the model (SWAT2009) are given by Neitsch et al. (2011). The current version of the model is briefly described here to provide an overview of the model structure.

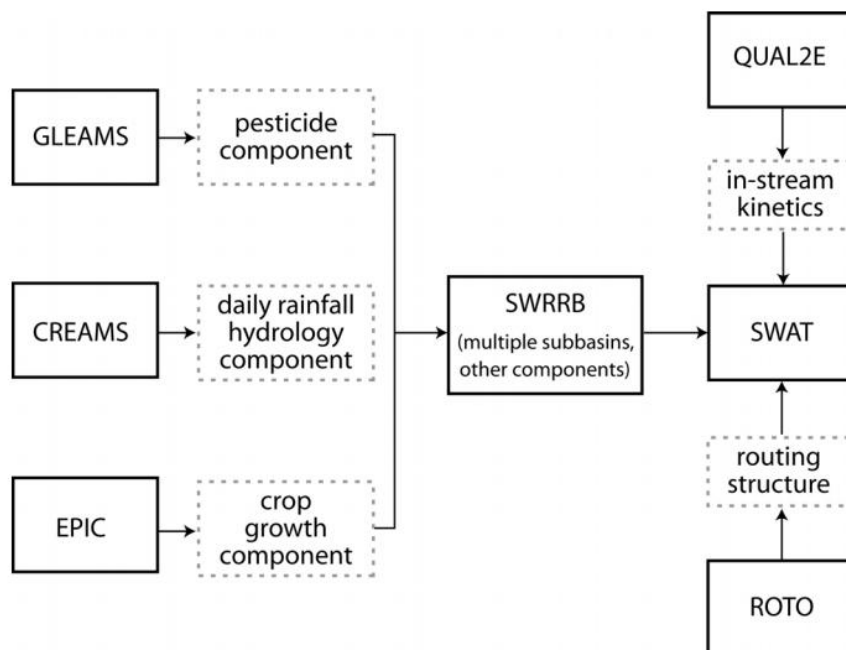


Figure 2.3 Schematic of SWAT developmental history

2.4.2 Model applications

Hydrology

Simulation of the hydrologic balance is foundational for all SWAT watershed applications and is usually described in some form regardless of the focus of the analysis. For example, Arnold and Allen (1993) used measured data from three Illinois watersheds to successfully validate surface runoff, groundwater flow, groundwater evaporation, evaporation in the soil profile, groundwater recharge, and groundwater height parameters.

Streamflow

The majority of SWAT applications reported some types of graphical and/or statistical hydrologic calibration, especially for streamflow, and many of the studies also report validation results. A wide range of statistics has been used to evaluate SWAT hydrologic predictions. The SWAT hydrologic subcomponents have been refined and validated at a variety of scales. Santhi et al. (2001, 2006) performed extensive streamflow validations for two Texas watersheds that cover over 4,000 km². Arnold et al. (1999) evaluated streamflow and sediment yield data in the Texas Gulf basin with drainage areas ranging from 2,253 to 304,260 km². Streamflow data from approximately 1,000 stream monitoring gauges from 1960 to 1989 were used to calibrate and validate the model. Predicted average monthly streamflows for three major river basins (20,593 to 108,788 km²) were 5% higher than measured flows, with standard deviations between measured and predicted within 2%. Annual runoff and evapotranspiration were validated across the entire continental U.S. as part of the Hydrologic Unit Model for the U.S. (HUMUS) modeling system. Rosenthal et al. (1995) linked GIS to SWAT and simulated ten years of monthly streamflow without calibration for a watershed in the Lower Colorado River basin of Texas (8,927 km²). SWAT underestimated the extreme events but produced overall accurate streamflows. Bingner (1996) simulated runoff for ten years for a watershed in northern Mississippi (0.05-21.3 km²). The SWAT model produced reasonable results in the simulation of runoff on a daily and annual basis from multiple subbasins, with the exception of a

wooded subbasin. Rosenthal and Hoffman (1999) successfully used SWAT and a spatial database to simulate flows, sediment, and nutrient loadings on a 9,000 km² watershed in central Texas to locate potential water quality monitoring sites. SWAT was also successfully validated for streamflow for the Mill Creek watershed (5,000 km²) in Texas for 1965-1968 and 1968-1975 (Srinivasan et al., 1998). Monthly streamflow rates were well predicted, but the model overestimated streamflows in a few years during the spring/summer months. The overestimation may be brought by variable rainfall events during those months.

Soil Water

Mapfumo et al. (2004) tested the model's ability to simulate soil water patterns in small watersheds under three grazing intensities in Alberta, Canada (1.52-2.26 ha). They observed that SWAT had a tendency to overpredict soil moisture content in dry soil conditions and to underpredict soil moisture content in wet soil conditions. Overall, the model was adequate in simulating soil water patterns for all three watersheds with a daily time step. Soil moisture was simulated with SWAT by Narasimhan and Srinivasan (2005) for six large river basins in Texas at a spatial resolution of 16 km² and a temporal resolution of one week. The simulated soil moisture was evaluated on the basis of vegetation response, by using 16 years of normalized difference vegetation index (NDVI) data derived from NOAA-AVHRR satellite data. The predicted soil moistures were well correlated with agriculture and pasture NDVI values. Narasimhan and Srinivasan (2005) describe further applications of a soil moisture deficit index and an evapotranspiration deficit index. Arnold et al. (2005) validated a crack flow model for SWAT in a watershed of 53.4 ha, which simulates soil moisture conditions with depth to account for flow conditions in dry weather. Simulated crack volumes were in agreement with seasonal trends, and the predicted daily surface runoff levels also were consistent with measured runoff data. Sun and Cornish (2005) simulated 30 years of bore data for a 437 km² watershed. They used SWAT to estimate recharge in the headwaters of the Liverpool Plains in New South Wales, Australia. These authors determined that SWAT could estimate recharge and incorporate land use and land

management at the watershed scale. A code modification was performed by Vazquez-Amable and Engel (2005) that allowed reporting of soil moisture for each soil layer. The soil moisture values were then converted into groundwater table levels based on the approach used in DRAINMOD (Skaggs, 1982). Model performance was evaluated in Muscatatuck River basin in southeast Indiana (2,952.21 km²), it was concluded that predictions of groundwater table levels would be useful to include in SWAT.

Snowmelt-Related Applications

Fontaine et al. (2002) modified the original SWAT snow accumulation and snowmelt routines by incorporating improved accounting of snowpack temperature and accumulation, snowmelt, and areal snow coverage, and an option to input precipitation and temperature as a function of elevation bands. These enhancements resulted in greatly improved streamflow estimates for the mountainous 5,000 km² upper Wind River basin in Wyoming. Abbaspour et al. (2007) calibrated several snow-related parameters and used four elevation bands in their SWAT simulation of the 1,700 km² Thur watershed in Switzerland that is characterized by a pre-alpine/alpine climate. They reported excellent SWAT discharge estimates. Other studies have reported mixed SWAT snowmelt simulation results, including three that reported poor results for watersheds (0.395 to 47.6 km²) in eastern Pennsylvania. Peterson and Hamlett (1998) found that SWAT was unable to account for unusually large snowmelt events, and Srinivasan et al. (2005) found that SWAT underpredicted winter streamflows; both studies used SWAT versions that predated the modifications performed by Fontaine et al. (2002). Srivastava et al. (2006) also found that SWAT did not adequately predict winter flows. Qi and Grunwald (2005) found that SWAT did not predict winter season precipitation runoff events well for the 3,240 km² Sandusky River watershed. Chanasyk et al. (2003) found that SWAT was not able to replicate snowmelt dominated runoff for three small grassland watersheds in Alberta (0.015-0.023 km²) that were managed with different grazing intensities. Wang and Melesse (2005) report that SWAT accurately simulated the monthly and annual (and seasonal) discharges for the

Wild Rice River watershed in Minnesota, in addition to the spring daily streamflows, which were predominantly from melted snow. Accurate snowmelt-dominated streamflow predictions were also found by Wang and Melesse (2005) for the Elm River in North Dakota (515.4 km²). Wu and Johnston (2007) found that the snow melt parameters used in SWAT are altered by drought conditions and that streamflow predictions for the 901 km² South Branch Ontonagon River in Michigan improved when calibration was based on a drought period (versus average climatic conditions), which more accurately reflected the drought conditions that characterized the validation period. Benaman et al. (2005) found that SWAT2000 reasonably replicated streamflows for the 1,200 km² Cannonsville Reservoir watershed in New York, but that the model underestimated snowmelt driven winter and spring streamflows. Improved simulation of cumulative winter streamflows and spring base flows were obtained by Tolson and Shoemaker (2007) for the same watershed by modifying SWAT2000 so that lateral subsurface flow could occur in frozen soils. Francos et al. (2001) also modified SWAT to obtain improved streamflow results for the Kerava River watershed (400 km²) in Finland by using a different snowmelt submodel that was based on degree days and that could account for variations in land use by subwatershed. Incorporating modifications such as those described in these two studies may improve the accuracy of snowmelt-related processes in future SWAT versions

Infiltration and Irrigation

Gosain et al. (2005) assessed SWAT's ability to simulate return flow after the introduction of canal irrigation in a basin in Andra Pradesh, India. SWAT provided the assistance water managers needed in planning and managing their water resources under various scenarios. Santhi et al. (2005) describe a new canal irrigation routine that was used in SWAT. Cumulative irrigation withdrawal was estimated for each district for each of three different conservation scenarios (relative to a reference scenario). The percentage of water that was saved was also calculated. Very few SWAT applications in the literature report the use of the Green-Ampt infiltration option. Di Luzio and Arnold (2004) report sub-hourly results for two different calibration methods using the

Green-Ampt method. King et al. (1999) found that the Green-Ampt option did not provide any significant advantage as compared to the curve number approach for uncalibrated SWAT simulations for the 21.3 km² Goodwin Creek watershed in Mississippi. Kannan et al. (2007) report that SWAT streamflow results were more accurate using the SCS curve number approach as compared to the Green-Ampt method for a small watershed in UK. However, they pointed out that several assumptions were not optimal for the Green-Ampt approach.

Applications Incorporating Wetlands, Reservoirs, and Other Impoundments

Arnold et al. (2001) simulated a wetland with SWAT that was proposed to be sited next to Walker Creek in the Fort Worth, Texas, area. They found that the wetland needed to be above 85% capacity for 60% of a 14-year simulation period, in order to continuously function over the entire study period. Conan et al. (2003) found that SWAT adequately simulated conversion of wetlands to dry land for the upper Guadiana River basin in Spain but was unable to represent all of the discharge details impacted by land use alterations. Wu and Johnston (2007) accounted for wetlands and lakes in their SWAT simulation of a Michigan watershed, which covered over 23% of the watershed. The impact of flood-retarding structures on streamflow for dry, average, and wet climatic conditions in Oklahoma was investigated with SWAT by Van Liew et al. (2003). The flood-retarding structures were found to reduce average annual streamflow by about 3% and to effectively reduce annual daily peak runoff events. Reductions of low streamflows were also predicted, especially during dry conditions. Mishra et al. (2007) report that SWAT accurately accounted for the impact of three checkdams on both daily and monthly streamflows for the 17 km² Banha watershed in northeast India. Hotchkiss et al. (2000) modified SWAT based on U.S. Army Corp of Engineers reservoir rules for major Missouri River reservoirs, which resulted in greatly improved simulation of reservoir dynamics over a 25-year period. Kang et al. (2006) incorporated a modified impoundment routine into SWAT, which allowed more accurate simulation of the impacts of rice paddy fields within a South Korean watershed.

Sediment

Several studies showed the robustness of SWAT in predicting sediment loads at different watershed scales. Saleh et al. (2000) conducted a comprehensive SWAT evaluation for the 932.5 km² upper North Bosque River watershed in north central Texas, and found that predicted monthly sediment losses matched measured data well but that SWAT daily output was poor. Srinivasan et al. (1998) concluded that SWAT predictions of monthly sediment accumulation were satisfactory for the 279 km² Mill Creek watershed, again located in north central Texas. Santhi et al. (2001) found that SWAT simulated monthly sediment loads matched measured sediment loads well for two Bosque River (4,277 km²) subwatersheds, except in March. SWAT sediment yield component has also been tested in several Midwest and northeast U.S. states. Chu et al. (2004) evaluated SWAT sediment prediction for the Warner Creek watershed located in the Piedmont physiographic region (3.46 km²) of Maryland. Evaluation results indicated strong agreement between yearly measured and SWAT simulated sediment load, but simulation of monthly sediment loading was poor. Tolson and Shoemaker (2007) modified the SWAT2000 sediment yield equation to account for both the effects of snow cover and snow runoff depth (the latter is not accounted for in the standard SWAT model) to overcome snowmelt-induced prediction problems identified by Benaman et al. (2005) for the Cannonsville Reservoir watershed (37-913 km²) in New York. They also reported improved monthly sediment loss predictions. Jha et al. (2007) found that the monthly and annual sediment loads predicted by SWAT were consistent with sediment loads measured for the Raccoon River watershed (8930 km²) in Iowa. Arabi et al. (2006) report satisfactory SWAT monthly sediment simulation results for two small watersheds (6.2-7.3 km²) in Indiana. White and Chaubey (2005) reported that SWAT monthly sediment predictions for the Beaver Reservoir watershed (362-1020 km²) in northeast Arkansas were satisfactory. Monthly sediment results are also reported by Cotter et al. (2003) for another Arkansas watershed (18.9 km²). Hanratty and Stefan (1998) calibrated SWAT using monthly sediment data measured in the Cottonwood River in Minnesota (3400 km²). In Wisconsin, Kirsch et al. (2002)

calibrated SWAT annual sediment predictions for two subwatersheds (190 km²) located in the Rock River basin, which lies within the glaciated portion of south central and eastern Wisconsin. Muleta and Nicklow (2005a) calibrated daily SWAT sediment yield with observed sediment yield data from the Big Creek watershed (23.9-86.5) in southern Illinois and concluded that daily sediment fit seems reasonable. However, validation was not conducted due to lack of data. SWAT sediment simulations have also been evaluated in Asia, Europe, and North Africa. Behera and Panda (2006) concluded that SWAT simulated daily sediment yield satisfactorily throughout the entire rainy season based on comparisons with daily observed data for an agricultural watershed (9.73 km²) located in eastern India. Kaur et al. (2004) concluded that SWAT predicted annual sediment yields reasonably well for a test watershed in Damodar-Barakar (9.58 km²), India, the second most seriously eroded area in the world. Tripathi et al. (2003) found that SWAT sediment predictions agreed closely with observed daily sediment yield for the same watershed. Mishra et al. (2007) found that SWAT accurately replicated the effects of three checkdams on daily and monthly sediment transport within the Banha watershed (17 km²) in northeast India. Hao et al. (2004) state that SWAT was the first physically based watershed model validated in China's Yellow River basin. They found that the predicted monthly sediment loading accurately matched loads measured for the 4,623 km² Lushi subwatershed. Cheng et al. (2006) successfully tested SWAT using monthly sediment data collected from the 7,241 km² Heihe River, another tributary of the Yellow River. Gikas et al. (2005) conducted an extensive evaluation of SWAT for the Vistonis Lagoon watershed (1349 km²), a mountainous agricultural watershed in northern Greece, and concluded that agreement between observed and SWAT predicted monthly sediment loads were acceptable. Bouraoui et al. (2005) evaluated SWAT for the Medjerda River basin (7,700 km²) in northern Tunisia and reported that the predicted monthly concentrations of suspended sediments were within an order of magnitude of corresponding measured values.

Nutrients

Several published studies from the U.S. showed the robustness of SWAT in predicting nutrient losses. SWAT was evaluated by comparing SWAT nitrogen prediction with measured nitrogen losses in the upper North Bosque River or Bosque River watersheds in Texas (Saleh et al., 2000; Santhi et al., 2001; and Di Luzio et al., 2002). They all concluded that SWAT reasonably predicted nitrogen loss, with most of the average monthly validation acceptable. Phosphorus losses were also satisfactorily simulated with SWAT in these four studies. Chu et al. (2004) applied SWAT to the Warner Creek watershed (3.46 km²) in Maryland and reported satisfactory annual but poor monthly nitrogen and phosphorus predictions. Hanratty and Stefan (1998) calibrated SWAT nitrogen predictions using measured monthly data collected for the Cottonwood River (3,400 km²), Minnesota, and concluded that if properly calibrated, SWAT is an appropriate model to use for simulating the effect of climate change on water quality; they also reported satisfactory SWAT phosphorus results. In Iowa, Chaplot et al. (2004) calibrated SWAT using nine years of data for the Walnut Creek watershed (51.3 km²) and concluded that SWAT gave accurate predictions of monthly nitrate load. Du et al. (2006) showed that the modified tile drainage functions in SWAT-M resulted in far superior nitrate loss predictions for Walnut Creek (51.3 km²), as compared to the previous approach used in SWAT2000. However, Jha et al. (2007) report accurate nitrate loss predictions for the Raccoon River watershed (8930 km²) in Iowa using SWAT2000. In Arkansas, Cotter et al. (2003) calibrated SWAT with measured monthly nitrate data for the Moores Creek watershed (18.9 km²) and reported that SWAT's response was similar to that of other published reports. Bracmort et al. (2006) and Arabi et al. (2006) found that SWAT could account for the effects of best management practices (BMPs) on phosphorus and nitrogen losses for two small watersheds (6.2-7.3 km²) in Indiana, with acceptable monthly validation. SWAT tended to underpredict both monthly mineral and total phosphorus yields for the months with high measured phosphorus losses, but overpredicted the phosphorus yields for months with low measured losses. Cerucci and Conrad (2003) calibrated

SWAT soluble phosphorus predictions using measured data obtained for the Townbrook watershed (36.8 km²) in New York. They reported good monthly performance if the measured data from February and March were excluded. Kirsch et al. (2002) reported that SWAT phosphorus loads were considerably higher than corresponding measured annual loads for the Rock River watershed (190 km²) in Wisconsin. Veith et al. (2005) found that SWAT-predicted losses were similar in magnitude to measured watershed exports of monthly dissolved and total phosphorus during a 7-month sampling period from a Pennsylvania watershed (0.395 km²).

SWAT nutrient predictions have also been evaluated in several other countries. In India, SWAT daily N and P predictions were tested using measured data within the Midnapore (9.73 km²) (Behera and Panda, 2006). They concluded that the SWAT model could be successfully used to satisfactorily simulate nutrient losses. SWAT-predicted monthly ammonia was close to the observed value for the Heihe River (7241 km²) study in China (Cheng et al., 2006). Studies conducted in Finland for the Vantaanjoki River (295-1682 km²) (Grizzetti et al. 2003; Bouraoui et al. 2004) watersheds reported that SWAT daily and monthly N and P simulations were generally satisfactory. Plus et al. (2006) evaluated SWAT from data on two rivers in the Thau Lagoon watershed (280 km²), which drains part of the French Mediterranean coast. The best correlations were found for daily nitrate loads, and the worst for daily ammonia loads. Gikas et al. (2005) evaluated SWAT in the Vistonis Lagoon watershed (1349 km²) in Greece and found that the monthly validation statistics generally indicated good model performance for nitrate and total P. SWAT biweekly nitrate and total phosphorus predictions were found to be excellent and good, respectively, by Abbaspour et al. (2007) for the 1700 km² Thur River basin in Switzerland. In addition to nitrogen and phosphorus, modifications were performed by Zhang et al. (2013) to SWAT2005 for six eddy covariance (EC) flux tower sites located across the U.S. Midwest to evaluate the performance of SWAT-C in simulating land-atmosphere C exchange. SWAT-C yielded acceptable estimate of land-atmosphere carbon exchange. For more SWAT model applications, please refer to Gassman et al. (2007).

3. Material and Methods

3.1 Study site

The Shibetsu river watershed covers 672 km² and is located in eastern Hokkaido (outlet as shown in Figure 3.1; 43.634N, 145.085E), Japan. This region has a hemi-boreal climate, characterized by warm summers and cold winters. Precipitation averages 1147 mm y⁻¹ and the annual mean temperature is 5 °C, with the lowest mean monthly temperature in February (-8.3 °C) and the highest mean monthly temperature in August (18 °C) (1978-2002 average, Japan Meteorological Agency 2007, <http://www.jma.go.jp>). There are five weather stations used for this study were all located outside the mountainous area. However, the mountainous area, with higher elevations, might have significantly different weather conditions from the other areas, and as 29% of the watershed. The snow period persists from early December to late April, with the annual maximum snowpack depth of 73 cm, and annual snowfall of 480 cm. Streamflow varies with seasons at the outlet of the Shibetsu river watershed. Peak streamflow occurs during the snowmelt season and again in August to November during rainfall events. The annual discharge is usually larger than annual precipitation (1980-2008, Japan Meteorological Agency 2007, <http://www.jma.go.jp/jma/indexe.html>), and this watershed was recognized as receiving external groundwater from other areas, as shown in Table 3.1. The average value from 1980 to 2008 was 504 mm year⁻¹ (1.38 mm day⁻¹, Table 3.1) calculated from annual water balance budget (Jiang et al. 2011).

Table 3.1 Annual water balance budget in the Shibetsu river watershed from 1980 to 2008 (Jiang et al., 2011).

Year	Discharge (mm)	Precipitation (mm)	Evapotranspiration (mm)	External groundwater (mm)
1980	774	890	548	432
1981	1369	1441	546	474
1982	928	933	565	560
1983	968	1010	527	485
1984	789	867	560	482
1985	954	1086	574	442
1986	905	919	527	514
1987	807	1060	542	289
1988	953	1036	511	428
1989	1209	1478	548	278
1990	1321	1513	555	363
1991	1112	1117	550	545
1992	1128	1317	525	337
1993	1236	1187	527	576
1994	1235	1183	601	653
1995	1065	1147	542	460
1996	1146	1106	529	569
1997	1117	1198	532	452
1998	1245	1391	515	369
1999	1066	1088	577	554
2000	- -	1263	528	- -
2001	- -	932	527	- -
2002	- -	1264	538	- -
2003	1258	1276	510	492
2004	967	931	579	615
2005	1103	975	531	659
2006	1642	1329	547	861
2007	1123	1038	614	699
2008	699	751	578	526
Mean	1082	1128	547	504
SD	210	196	26	131

The watershed consists of forest (53.7%), agriculture (40.8%), urban area (4.5%), and water (1.0%) (Table 3.2). Grassland and pasture occupy 95% of the agricultural land. The major vegetation types are *Phleum pretense* in agriculture land and Japanese larch-*Larix kaempferi* in the forest. And the remaining is cultivated with maize (*Zea mays* L.), sugar beet (*Saccharum officinarum*), potato (*Solanum tuberosum*), and Japanese radishes (*Raphanus sativus* Linn.). The major soil types of the SRW include Peat soils (3.26 %), Regosolic Kuroboku soils (13.94 %), Brown Forest soils (20.56 %), Kuroboku soils (46.08 %), Brown Lowland soils (9.10 %), Regosols (4.51 %) and Gray Lowland soils (2.55 %) (Cultivated Soil Classification committee, Japan 1995), which corresponds to Histosols, Vitric Andosols, Cambisols, Silandic Andosols, Haplic Fluvisols, Regosols, and Gleyic Fluvisols, respectively (IUSS, 2006) (Table 3.2). The principal characteristics of the soils have been presented by Jiang et al. (2011). All the soils in this watershed are volcanogeneuous soils. The topographical characteristic of the region has a mean slope of 4.28 °with a maximum value of 34 °. The Shibetsu watershed has a coupled land use and topography characteristics. The upstream area is characterized by steep topography and covered by forest, whereas downstream area possessed gentle sloping areas being used for agriculture. The human population was estimated at 24000 persons (35 people km⁻²) in 2003 concentrated in the middle part of the watershed. The Nakashibetsu town, covering most of the Shibetsu river watershed, has a large grassland area, and dairy farming is a main occupation. The livestock mainly includes beef cattle, dairy cattle and horses, and there is about 2.07 livestock unit ha⁻¹ (One livestock unit is equivalent to 1 dairy or beef cattle, or 1.3 horses). The Shibetsu river suffers from non-point source pollution associated with intensive dairy farming in the grassland areas (Woli et al. 2004; Hayakawa et al. 2006; Jiang et al., 2014).

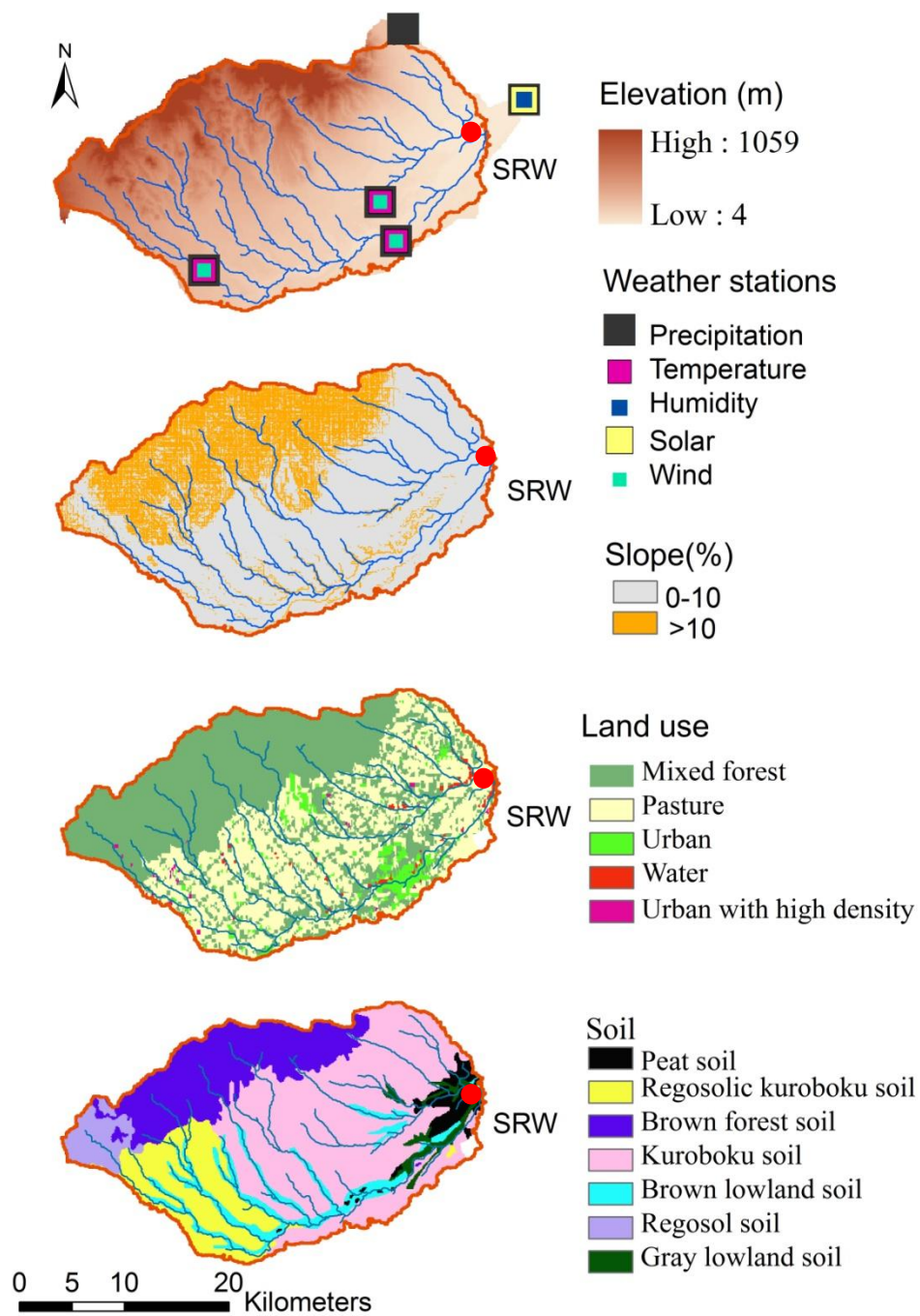


Figure 3.1 DEM, slope, land use and soil maps,
and locations of weather stations and sampling sites

Table 3.2 Characteristics of Shibetsu river watershed

Description		SRW (672 km ²) Percentage of basin area
Land use	Mixed forest	53.72
	Pasture	40.78
	Urban	4.21
	Urban with high residency	0.35
Soil type	Brown forest soil	20.56
	Brown lowland soil	9.10
	Gray lowland soil	2.55
	Kuroboku soil	46.08
	Peat soil	3.26
	Regosolic kuroboku soil	13.94
	Regosol soil	4.51
Slope	0–10	30
	>10	70

3.2 Sampling and monitoring

To get the streamflow, suspended sediment and nutrient concentration in stream, soil and plant, sampling and monitoring for the stream, soil and plant were conducted with the following methods:

Stream

Stream water samples were collected in stream outlets with 1 L polypropylene bottles during three periods over a year (baseflow, rainfall events and snowmelt season) from 2003 to 2008. Baseflow was sampled once a month manually. During rainfall events and snowmelt season, an automated water sampler (ISCO 6712, Isco, Lincoln, NE, USA) was installed at the outlet of each watershed, and water samples were collected. The auto-sampler was triggered when rainfall was $>4 \text{ mm } 30 \text{ min}^{-1}$ with intervals of 15 min to 1 h at the rising stage of discharge, while from 2 to 6 h at the receding stage. During the snowmelt season, the sampler was triggered manually with a sampling interval of 12 h. Water samples were transported to the laboratory immediately after the collection and then stored at 4°C until analysis. Streamflow rates during the baseflow events were measured by a flow velocity meter (TK-105; Toho Dentan, Tokyo) at the outlets of sub-watersheds. The water discharge rates were calculated by multiplying the sectional area of the stream by flow velocity along a transect across the

stream. Daily stream water table was recorded at every 15 min by water sensors equipped with data loggers (MC-1100W, STS, Sirmach, Switzerland). Daily discharge was calculated by daily stream water table and calibrated H-Q equations (quadratic curve), which are the relationships between runoff (Q) and the water table (H) (Tachibana and Nasu, 2003). The discharge for Shibetsu watershed was recorded at the website of Ministry of Land, Infrastructure and Transport, Japan (<http://www1.river.go.jp>). Meteorological data was obtained from Japan Meteorological Agency (<http://www.jma.go.jp/ima/indexe.html>). Stream sediment samples were collected during the snowmelt season (May, 2013).

Soil and Plant

Soil samples were collected at all the seven soil types of the watershed (Figure 3.1) in October, 2013. Soil sample was taken at top 5 cm. Each soil sample was sieved through a 2 mm sieve to remove the roots and other organic matters, also the sample were mixed and homogenized during sieving in the laboratory. The forest leaves and harvested grass were collected around the end of growing season (October, 2013) at the soil types of Peat soils, Kuroboku soils and Gray Lowland soils (Figure 3.1). In the laboratory, the soil samples were air dried and plant sample were oven dried at 80°C before analysis because the dry weight was used calculate nutrient concentration.

3.3 Laboratory analysis

Stream sediment, particulate organic nitrogen and phosphorous, and potassium

Water samples were filtered through 0.7 μm glass microfibre filters for the analysis of suspended sediments (Hayakawa et al., 2009). A portion of the water samples were filtered through 0.2 μm membrane filters within a few days, and analysed for total dissolved nitrogen (TDN) and total dissolved phosphorous (TDP) (Hayakawa et al., 2009). The remaining non-filtered samples were used for total nitrogen (TN), total phosphorous (TP) and potassium analysis (Hayakawa et al., 2009). Concentrations of TN, TP, TDN and TDP were determined using the method of alkaline persulfate digestion and HCl-acidified UV detection (Hayakawa et al., 2009). The potassium concentration was measured with the Schollenberger method (Schollenberger and Simon, 1945). The

particulate organic nitrogen (PON) and particulate organic phosphorous (POP) concentrations were calculated by subtracting the concentration of TDN from TN, and TDP from TP respectively (Hayakawa et al., 2009). Sediment and PON and POP yields were calculated from the sediment concentrations and discharge data. Stream sediment exchangeable potassium was analyzed with the Schollenberger method (Schollenberger and Simon, 1945).

Potassium concentration in plant, soil

For the plant samples, 1 g of samples were weighed to a 100mL beaker, and 61% nitric acid (Wako Pure Chemical Industries, Oosaka) 10mL was added. After the intense reaction was settled, it heated to 110°C on the hot plate. Furthermore, 30% hydrogen peroxide (Wako Pure Chemical Industries, Ltd., Oosaka) was added and heated. Hydrogen peroxide processing was repeated until liquid became transparent. After making up to 100mL by deionized water, filtered and potassium concentration was measured with the Schollenberger method (Schollenberger and Simon, 1945).

Surface soil samples (0-5cm) were collected and air dried in the laboratory. To determine the exchangeable potassium in the soil, 8 g soil was extracted by 100 mL of 1 M ammonium acetate (pH7), and the potassium concentration of extract was measured with the Schollenberger method (Schollenberger and Simon, 1945).

3.4 Outline of SWAT model

SWAT (Soil and Water Assessment Tool) is a comprehensive continuous time watershed model that operates on a daily time step. The simulation incorporates weather, hydrology, sedimentation, soil temperature, plant growth, nutrients, pesticides and land management. It's developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time (Neitsch et al., 2011). SWAT applies in a semi-distributed manner supported by an ArcGIS interface. The input data for SWAT model includes weather, topography, land use and soil data. The basin is divided into subbasins defined by topography and river network. Each subbasin is further subdivided into several hydrologic response units (HRUs) having uniform

land cover, soil and management combinations (Figure 3.2). Input information for each subbasin is grouped or organized into the following categories: climate; HRUs; pond/wetlands; groundwater; and the main channel, or reach, draining the subbasin (Neitsch et al., 2011).

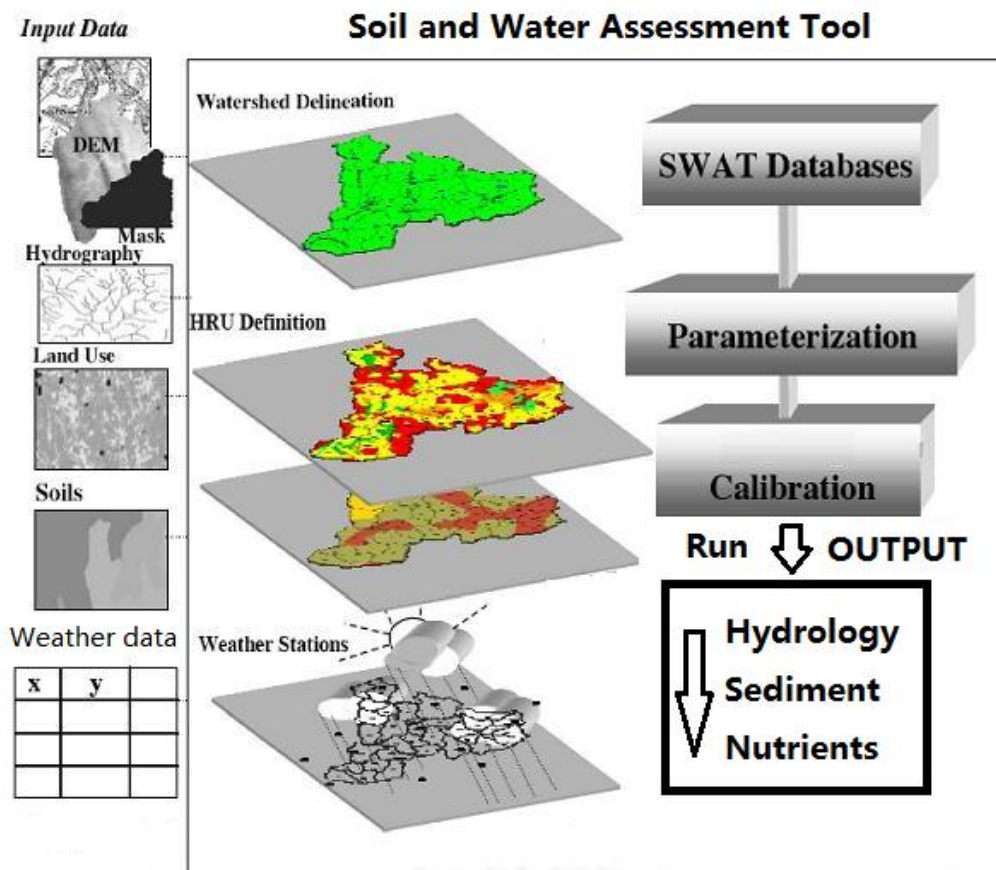


Figure 3.2 Schematic diagram of SWAT model

3.4.1 Hydrology

Water balance

Simulation of the hydrology of a watershed can be separated into two major divisions. The first division is the land phase of the hydrologic cycle (Figure 3.3) (Arnold et al., 1996), which controls the amount of water, sediment, nutrients and pesticide loadings to the main channel in each subbasin. The second division is the water or routing phase of the hydrologic cycle which can be defined as the movement of water, sediments, etc. through the channel network of the watershed to the outlet.

The water balance of a watershed is a deterministic relationship between the various components of the water balance (Figure 3.3). In conservative watershed, the

independent input variable is rainfall. The input variable rainfall and external water are transformed within the hydrological system into the dependent output variables of evaporation, streamflow soil storage change, and groundwater storage change (Salerno and Tartari 2009). The hydrologic cycle as simulated by SWAT is based on the water balance equation:

$$SW_t = SW_0 + \sum_{i=1}^t (P - ET - Q_{surf} - Q_{lat} - Q_{gw}) \quad \text{【3.1】}$$

Where SW_t is the final soil water content (mm H_2O), SW_0 is the initial soil water content on day i (mm H_2O), P is the amount of precipitation on day i (mm H_2O), ET is the amount of evapotranspiration on day i (mm H_2O), Q_{surf} is the amount of surface runoff on day i (mm H_2O), Q_{lat} is the amount of lateral flow on day i (mm H_2O), and Q_{gw} is the amount of groundwater recharge on day i (mm H_2O).

While in the non-conservative watershed; external water (EXT) source also is input variable. Water balance in a non-conservative watershed (with water losses or gains) can be expressed as:

$$SW_t = SW_0 + \sum_{i=1}^t (P + EXT - ET - Q_{surf} - Q_{lat} - Q_{gw}) \quad \text{【3.2】}$$

where EXT is the external water contribution (mm) that is positive for water gain and negative for water loss,

These equations enable the simulation of surface runoff, interflow, evapotranspiration, soil water redistribution, groundwater storage for each HRU. The aquifer system is conceptualized as a two-layer system consisting of a shallow and a deep aquifer. Water in the shallow aquifer can return to soil profile or enter into the stream as a groundwater flow, while water in the deep aquifer is supposed to leave the system (Figure 3.3). The surface, inter and groundwater flow are calculated for each subbasin as an area-weighted average of its composing HRUs. The runoff volumes generated from each subbasin are then routed to the basin outlet.

In conservative environment, total water entering channels per day from a HRU in the SWAT model can be derived from:

$$Q = q_{surf} + q_{lat} + q_{gw} \quad \text{【3.3】}$$

where q is total water entering channels as streamflow (mm), q_{surf} is surface runoff

contribution to streamflow (mm), q_{lat} is lateral flow contribution to streamflow (mm), q_{gw} is groundwater contribution to streamflow (mm).

For the non-conservative environment, in this study, the EXT contribution from each HRU was added into total water entering channels:

$$Q = q_{surf} + q_{lat} + q_{gw} + EXT \quad \text{【3.4】}$$

where q_{gw} here is internal groundwater contribution to streamflow (mm), EXT is external groundwater contribution from outside the watershed to streamflow (mm), the average value from 1980 to 2008 was 1.38 mm day^{-1} (504 mm year^{-1} , Table 3.1) calculated from annual water balance budget (Jiang et al. 2011).

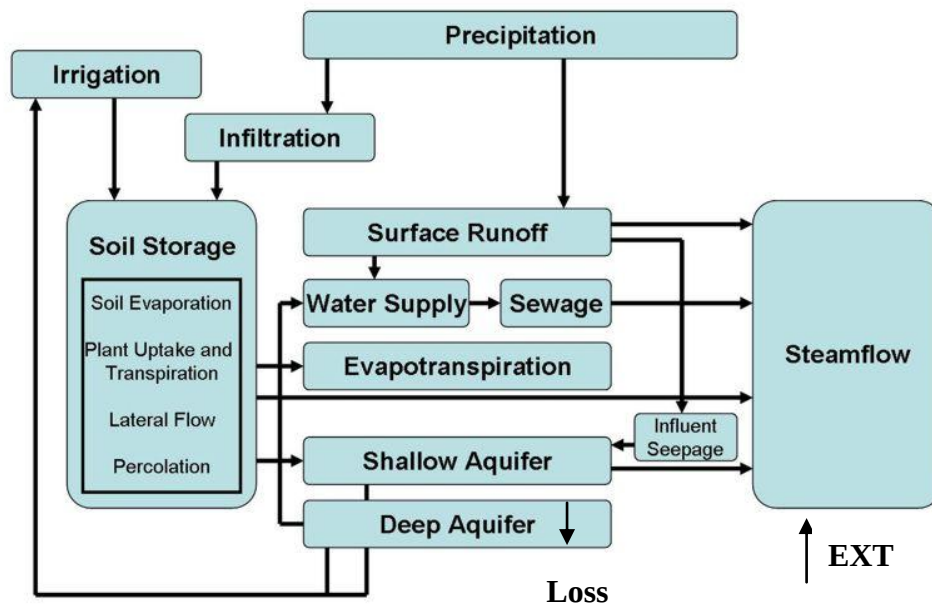


Figure 3.3 Components of water balance in a watershed

Snowmelt hydrology

In SWAT, snowmelt hydrology is realized on an HRU (hydrologic response unit) basis. When the mean daily air temperature is lower than the snowfall temperature, as specified by the variable SFTMP, the precipitation within an HRU is classified as snow and the liquid water equivalent of the snow precipitation is added to the snowpack. The snowpack increases with additional snowfall, but decreases with snowmelt or sublimation. The mass balance for the snowpack is computed as:

$$SNO_i = SNO_{i-1} + R_{sfi} - E_{subi} - SNO_{mli} \quad \text{【3.5】}$$

where SNO_i and SNO_{i-1} are the water equivalents of the snowpack on the current day (i) and previous day (i-1), respectively, R_{sfi} is the water equivalent of the snow precipitation on day i, E_{subi} is the water equivalent of the snow sublimation (as vapor) on day i, and SNO_{mli} is the water equivalent of the snowmelt (as liquid) on day i. All of these variables are reported in terms of the equivalent water depth (mm) over the total HRU area.

The snowpack is rarely uniformly distributed over the total area, resulting in a fraction of the area that is bare of snow. In SWAT, the areal coverage of snow over the total HRU area is defined using an areal depletion curve, which describes the seasonal growth and recession of the snowpack (Anderson, 1976) and is defined as:

$$sno_{covi} = \frac{SNO_i}{SNOCOV_{MX}} \left[\frac{SNO_i}{SNOCOV_{MX}} + \exp(cov_1 - cov_2 \times \frac{SNO_i}{SNOCOV_{MX}}) \right]^{-1} \quad \text{【3.6】}$$

where sno_{covi} is the fraction of the HRU area covered by snow on the current day (i), $SNOCOV_{MX}$ is the minimum snow water content that corresponds to 100% snow cover (mm H_2O), and cov_1 and cov_2 are the coefficients that define the shape of the equation. The values used for cov_1 and cov_2 are determined by solving the above equation using two known points: (1) 95% coverage at 95% $SNOCOV_{MX}$, and (2) 50% coverage at a user defined fraction of $SNOCOV_{MX}$. The value of sno_{covi} is assumed to be equal to 1.0 once the water content of the snowpack exceeds $SNOCOV_{MX}$, indicating a uniform depth of snow over the HRU area.

In addition to the areal coverage of snow, snowmelt is also controlled by the snowpack temperature and melting rate. The influence of the previous day's snowpack temperature on the current day's snowpack temperature is described by a lag factor, specified by the variable $TIMP$, which implicitly accounts for snowpack density, water content, and exposure.

The snowpack temperature is calculated as:

$$T_{spi} = T_{spi-1}(1-TIMP) + T_{ai} \cdot TIMP \quad \text{【 3.7】}$$

where T_{spi} and T_{spi-1} are the snowpack temperatures on the current day (i) and the

previous day ($i-1$) ($^{\circ}\text{C}$), respectively, and T_{ai} is the mean air temperature on day i ($^{\circ}\text{C}$). As TIMP approaches 1.0, T_{ai} exerts an increasingly greater influence on T_{spi} ; conversely, as TIMP moves away from 1.0, T_{spi-1} becomes more important.

The amount of snowmelt on the current day (i), SNO_{mlti} , expressed in terms of the equivalent amount of water in mm, or melting rate, is calculated in SWAT as:

$$\text{SNO}_{\text{mlti}} = b_{\text{mlti}} \times \text{sno}_{\text{covi}} \left(\frac{T_{\text{spi}} + T_{\text{maxi}}}{2} - \text{SMTMP} \right) \quad \text{【3.8】}$$

where T_{maxi} is the maximum air temperature on day i ($^{\circ}\text{C}$), SMTMP is the base temperature above which snowmelt is allowed ($^{\circ}\text{C}$), and b_{mlti} is the melt factor on day i ($\text{mm H}_2\text{O } ^{\circ}\text{C}^{-1} \text{ day}^{-1}$), which is calculated as:

$$b_{\text{mlti}} = \frac{\text{SMFMX} + \text{SMFMN}}{2} + \frac{\text{SMFMX} - \text{SMFMN}}{2} \times \sin \left[\frac{2\pi}{365} (i - 81) \right] \quad \text{【3.9】}$$

where SMFMX and SMFMN are the maximum and minimum snowmelt factors, respectively ($\text{mm H}_2\text{O } ^{\circ}\text{C}^{-1} \text{ day}^{-1}$).

3.4.2 Sediment

Landscape erosion

The SWAT model estimates soil erosion and sediment yield from the landscape and in-stream depositional and degrading processes. The sediment yield from the landscape is calculated by the MUSLE (Williams, 1975). The MUSLE is:

$$\text{SED} = 11.8 \times (Q_{\text{surf}} \times q_{\text{peak}} \times \text{area}_{\text{hru}})^{0.56} \times K_{\text{USLE}} \times C_{\text{USLE}} \times P_{\text{USLE}} \times L_{\text{USLE}} \times \text{CFRG} \quad \text{【3.10】}$$

where SED is the sediment yield (Mg day^{-1}), Q_{surf} is the surface runoff volume ($\text{mm ha}^{-1} \text{ day}^{-1}$), q_{peak} is the peak runoff rate ($\text{m}^3 \text{ s}^{-1}$), area_{hru} is the area of the HRU (ha), K_{USLE} is the USLE soil erodibility factor, C_{USLE} is the USLE cover and management factor, P_{USLE} is the USLE support practice factor, L_{USLE} is the USLE topographic factor, and CFRG is the coarse fragment factor. The K_{USLE} factor should be between 0 and 0.65. C_{USLE} is calculated from minimum C_{USLE} factor USLE_C which is between 0.001 and 0.5 for the land cover. The P_{USLE} factor should be between 0 and 1. L_{USLE} is calculated from SLSUBBSN.hru (slope length, m, ranged between 10 and 150) and HRU_SLP.hru (average slope of the subbasin, % or m m^{-1} , ranged between 0 and 0.6). CFRG is

between 0 and 100 in percentage.

Chanel routing of sediment

Sediment deposition and degradation in the stream channel are both calculated during the sediment routing. The channel sediment routing equation uses a modification of Bagnold's sediment transport equation (Bagnold, 1977) that estimates maximum concentration of sediment as a function of velocity. The model either deposits excess sediment or re-entrains sediment through channel erosion depending on the sediment load entering the channel and the maximum amount of sediment that can be transported from a reach segment.

$$\text{CONC}_{\text{sed,ch,mx}} = \text{SPCON} \times V_c^{\text{SPEXP}} \quad \text{【3.11】}$$

where $\text{CONC}_{\text{sed,ch,mx}}$ is the maximum concentration of sediment that can be transported (Mg m^{-3} or kg L^{-1}), SPCON is the coefficient in this equation defined by the user, V_c is the peak flow velocity (m s^{-1}) in the channel, and SPEXP is an exponent parameter in the equation. The coefficient (SPCON) should be between 0.0001 and 0.01. The exponent (SPEXP) normally ranges from 1.0 to 1.5.

The final amount of sediment in the reach per day was determined as:

$$\text{sed}_{\text{ch}} = \text{sed}_{\text{ch,i}} - \text{sed}_{\text{dep}} + \text{sed}_{\text{deg}} \quad \text{【3.12】}$$

where sed_{ch} is the amount of sediment in the reach (Mg), $\text{sed}_{\text{ch,i}}$ is the amount of sediment in the reach at the beginning of the time period (Mg), sed_{dep} is the amount of sediment deposited in the reach segment (Mg), and sed_{deg} is the amount of sediment reentrained in the reach segment (Mg).

The SWAT model was modified by taking into account the EXT contribution to channel water and the effect of EXT on sediment routing in channels. The model with EXT was used to estimate sediment, PON, and POP yields at the main outlet of SRW. Results were investigated and compared with observed data estimated from the rating curve method to evaluate its performance for sediment yield estimation in the SRW.

3.4.3 Potassium

Potassium (K) may be applied to the watershed through atmospheric deposition or to the agricultural land through fertilizer application because it is an essential nutrient for

plant growth. The potassium cycle was added to the SWAT model and this modified SWAT model was named as SWAT-K in the rest of the paper. The equations used to model the movement of potassium in the land phase and stream network of the hydrologic cycle were adopted from pesticide module (Leonard et al., 1987) in SWAT model. The solid-liquid distribution of the potassium is controlled by soil clay adsorption coefficient. Potassium transport by water and sediment is calculated for each runoff event and potassium leaching is estimated for each soil layer and groundwater when percolation occurs. SWAT-K address potassium input (wet deposition, fertilizer and manure application), potassium goes into the soluble and absorbed forms in the soil. Plants can uptake potassium from soil during growing seasons or returns potassium to soil as residue after harvest or forest litter fall during the dormant period. Water-soluble potassium can be transported surface runoff, lateral flow or with percolation deep into the soil profile and enter into shallow groundwater systems. Soil particles absorbed potassium may loss from soil erosion with surface runoff. SWAT simulates K movement into the stream network via surface runoff and groundwater flow to streams. After K entering streams, routing of sediment begins.

The total potassium load in the channel is partitioned into dissolved and sediment-attached components. While the dissolved potassium is transported with water, the potassium attached to sediment is affected by sediment transport and deposition processes. The major in-stream processes simulated by the model are settling, burial, resuspension and diffusion. Figure 3.4 shows potassium fate and transport in SWAT model. Potassium in a reach segment is increased through addition of mass in inflow as well as resuspension and diffusion of potassium from the sediment layer. The amount of potassium in a reach segment is reduced through removal in outflow as well as settling and diffusion into the underlying sediment. Potassium in the sediment layer underlying a reach segment is increased through addition of mass by settling and diffusion from the water. The amount of potassium in the sediment layer is reduced through removal by resuspension and diffusion into the overlying water, and burial.

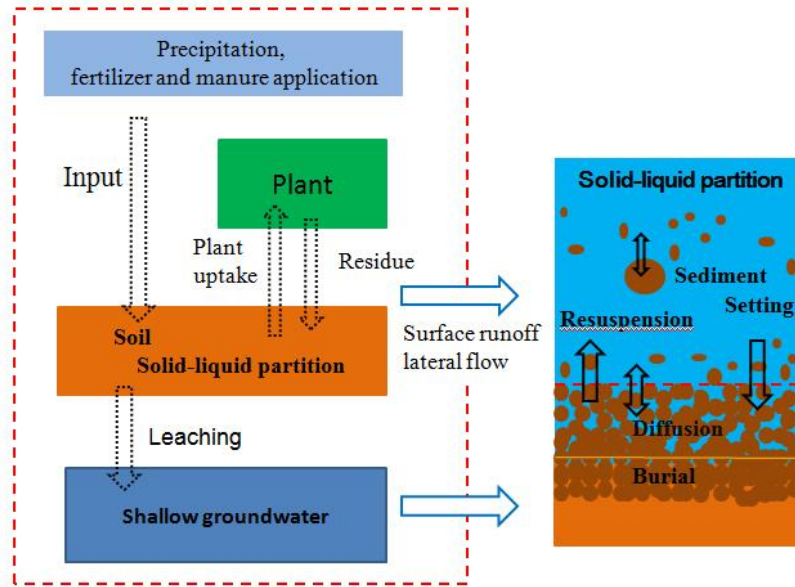


Figure 3.4 Potassium fate and transport in SWAT-K

Potassium transport in soil

Solid-liquid distribution of potassium

Potassium in the soil environment can be transported in soil solution or attached to sediment. The partitioning of potassium between the solid and liquid phases in the soil is defined by the solid-liquid distribution coefficient for the potassium. The soil solid-liquid distribution coefficient is the ratio of the potassium concentration in soil particles or solid phase to the potassium concentration in soil solution or liquid phase:

$$K_p = C_{\text{solidphase}} / C_{\text{liquidphase}} \quad \text{【3.13】}$$

where K_p is the solid-liquid distribution coefficient ($(\text{mg kg}^{-1})/(\text{mg L}^{-1})$ or $\text{m}^3 \text{Mg}^{-1}$), $C_{\text{solidphase}}$ is the concentration of the potassium absorbed to the solid phase (mg kg^{-1} solid material or g Mg^{-1}), and $C_{\text{liquidphase}}$ is the concentration of the potassium in liquid phase (mg L^{-1} solution or g m^{-3}). The definition of the solid-liquid distribution coefficient in assumes that the potassium sorption process is linear with concentration and instantaneously reversible.

Bangroo et al. (2012) found that partitioning of potassium is dependent upon the amount of clay in the soil, the solid-liquid distribution coefficient input to the model is normalized for soil clay content. The relationship between the solid-liquid distribution coefficient and the solid-liquid distribution coefficient normalized for soil clay content

is:

$$K_p = K_{\text{clay}} \times (\text{clay}/100) \quad \text{【3.14】}$$

where K_p is the solid-liquid distribution coefficient ((mg kg⁻¹)/(mg L⁻¹)), K_{clay} is the solid-liquid distribution coefficient normalized for soil clay content ((mg kg⁻¹)/(mg L⁻¹ or m³ Mg⁻¹), and clay is the percent clay present in the soil. Note that in this study a simplified equation was used to describe the K adsorption in soil, actually K adsorption in soil is quite complex and controlled by many factors, such as pH, soil organic matter, hydroxide aluminum, soil moisture status, cation exchange capacity, fertilization and tillage system (Bangroo et al., 2012).

Plant uptake potassium

Equation for potassium uptake by plant is based on soil to plant transfer factor F_v , defined as the ratio of the potassium content in the plant (mg kg⁻¹ dry weight plant tissue) to that in the soil first soil layer (mg kg dry⁻¹ weight soil) (IAEA, 2010). It assumes a linear relationship between the potassium concentration in plant (K_{plant}) and potassium concentration in soil (K_{soil}) (Kurashima et al., 1980). In SWAT, the first soil layer is typically delineated with a 1cm thickness, so as to better represent the rapid change of moisture, temperature, and nutrient at soil surface as influenced by weather conditions and human activities (e.g. precipitation, irrigation, and fertilizer application). Here, to be commensurate with the existing structure of SWAT, SWAT-K simultaneously addresses potassium content in top 1cm layer Potassium in crop residues and forest litter fall, can be washed easily and directly to the soil via rainfall, independent of the degree of organic matter mineralization because it is not tied up in organic forms in the plant (Rosolem and Calonego, 2013). Therefore, in SWAT-K model, potassium in crop residues and forest litter fall was added to the first soil layer as soon as the residue happens. Potassium content in the plant is described by:

$$K_{\text{plant}} = F_v * K_{\text{soil}} \quad \text{【3.15】}$$

The total amount of uptake of potassium depends on the plant potassium concentration and biomass growth.

$$K_{\text{up}} = K_{\text{plant}} * \text{bioday} \quad \text{【3.16】}$$

where K_{up} is daily uptake amount by plant in HRU (kg K ha^{-1}), $bioday$ is biomass generated on current day in HRU (kg K ha^{-1}), K_{soil} is average concentration of potassium in first soil layer (with root) in HRU (kg K kg^{-1}).

In this study, it might have uncertainty by assuming soil to plant transfer ratio of potassium as a constant value because it might vary with the plant growth stage and organs. Claassen et al. (1986) reported that the amount of potassium in the plant decreases as it matures; it is therefore important to know the plant's stage of growth to properly interpret the results of potassium analysis. Also, the potassium concentration usually decreases from top to bottom of the plant, from new organs to the old organs, so the portion of the plant sampled affects the interpretation as well. In SWAT-K, potassium in crop residues and forest litter fall was assumed to be returned to the first soil layer as soon as the residue happens. This assumption might also cause uncertainty in modeling the potassium cycling processes at short time steps (hourly or daily), because potassium release in crop residues and forest litter fall might depend on soil temperature and moisture. However, this study focuses on the potassium losses to streams and annual potassium cycling, the simplified assumption of the plant uptake and residue release of potassium were used in this study, future researches could be conducted to investigate factors controlling these processes.

Root density is greatest near the surface, and potassium uptake in the upper portion of the soil will be greater than in the lower portion. The depth distribution of potassium uptake is controlled by β_K , the potassium uptake distribution parameter, a variable users are allowed to adjust. The depth distribution of potassium uptake is calculated with the function:

$$K_{up,z} = \frac{K_{up}}{1 - \exp(-\beta_K)} \left\{ 1 - \exp\left(-\beta_K \frac{z}{z_{root}}\right) \right\} \quad \text{【3.17】}$$

where $K_{up,z}$ is potential potassium uptake from the soil surface to depth z (kg ha^{-1}), K_{up} is potential potassium uptake (kg ha^{-1}), β_K is the potassium uptake distribution parameter, z is the depth from the soil surface (mm), z_{root} is the depth of root development in the soil (mm).

Movement of soluble potassium

Potassium in the liquid phase may be transported with surface runoff, lateral flow or percolation. The amount of potassium removed in solution with the flow can be obtained by:

$$K_{\text{flow}} = K_{s,ly,o} \left(1 - \exp \left[\frac{-w_{\text{mobile}}}{SAT_{ly} + K_p \rho_b \text{depth}_{ly}} \right] \right) \quad \text{【3.18】}$$

where K_{flow} is the amount of potassium removed in the flow (kg ha^{-1}), $K_{s,ly,o}$ is the initial amount of potassium in the soil layer (kg ha^{-1}), w_{mobile} is the amount of mobile water in the layer ($\text{mm H}_2\text{O}$), SAT_{ly} is the amount of water in the soil layer at saturation ($\text{mm H}_2\text{O}$), K_p is the solid-liquid distribution coefficient ($(\text{mg kg}^{-1})/(\text{mg L}^{-1})$), ρ_b is the bulk density of the soil layer (Mg m^{-3}), and depth_{ly} is the depth of the soil layer (mm).

The amount of mobile water in the layer is the amount of water lost by surface runoff, lateral flow or percolation:

$$w_{\text{mobile}} = Q_{\text{surf}} + Q_{\text{lat,surf}} + Q_{\text{perc,surf}}, \quad \text{for top 10 mm} \quad \text{【3.19】}$$

$$w_{\text{mobile}} = Q_{\text{lat,ly}} + Q_{\text{perc,ly}}, \quad \text{for lower soil layers} \quad \text{【3.20】}$$

where w_{mobile} is the amount of mobile water in the layer ($\text{mm H}_2\text{O}$), Q_{surf} is the surface runoff generated on a given day ($\text{mm H}_2\text{O}$), $Q_{\text{lat,ly}}$ is the water discharged from the layer by lateral flow ($\text{mm H}_2\text{O}$), and $w_{\text{perc,ly}}$ is the amount of water percolating to the underlying soil layer on a given day ($\text{mm H}_2\text{O}$).

For the top 10 mm that interacts with surface runoff, the potassium concentration in the mobile water is calculated:

$$\text{conc}_{K,\text{flow}} = \min \left\{ \frac{K_{\text{flow}}}{[w_{\text{perc,surf}} + \beta(Q_{\text{surf}} + Q_{\text{lat,surf}})]}, \frac{K_{\text{sol}}}{100} \right\} \quad \text{【3.21】}$$

while for the lower layers

$$\text{conc}_{K,\text{flow}} = \min \left\{ \frac{K_{\text{flow}}/w_{\text{mobile}}}{K_{\text{sol}}/100}, \frac{K_{\text{sol}}}{100} \right\} \quad \text{【3.22】}$$

where $\text{conc}_{K,\text{flow}}$ is the concentration of potassium in the mobile water ($\text{kg ha}^{-1} \text{mm}^{-1} \text{H}_2\text{O}$), β is the potassium percolation coefficient, and K_{sol} is the solubility of potassium in water (mg L^{-1}).

Potassium removed in surface runoff is calculated:

$$K_{\text{surf}} = \beta \times \text{conc}_{K,\text{flow}} \times Q_{\text{surf}} \quad \text{【3.23】}$$

where K_{surf} is the potassium removed in surface runoff (kg ha^{-1}).

Potassium removed in lateral flow is calculated:

$$K_{\text{lat,surf}} = \beta \times \text{conc}_{K,\text{flow}} \times Q_{\text{lat,surf}}, \quad \text{for top 10 mm} \quad \text{【3.24】}$$

$$K_{\text{lat,ly}} = \text{conc}_{K,\text{flow}} \times Q_{\text{lat,ly}}, \quad \text{for lower layers} \quad \text{【3.25】}$$

where $K_{\text{lat,surf}}$ is the potassium removed in lateral flow from a layer (kg ha^{-1}),

Potassium moved to the underlying layer by percolation is calculated:

$$K_{\text{perc,ly}} = \text{conc}_{K,\text{flow}} \times w_{\text{perc,ly}}, \quad \text{【3.26】}$$

where $K_{\text{perc,ly}}$ is the potassium removed in surface runoff (kg ha^{-1}).

Transport of absorbed potassium

Potassium attached to soil particles may be transported by surface runoff to the main channel. The solid phase of potassium is associated with the sediment loading from the HRU and changes in sediment loading will impact the loading of absorbed potassium. The amount of potassium transported with sediment to the stream is calculated with a loading function developed by McElroy et al. (1976) and modified by Williams and Hann (1978).

$$K_{\text{sed}} = 0.001 \times C_{\text{solidphase}} \times \frac{\text{sed}}{\text{area}_{\text{hru}}} \epsilon_{K\text{sed}} \quad \text{【3.27】}$$

where K_{sed} is the amount of absorbed potassium transported to the main channel in surface runoff (kg ha^{-1}), $C_{\text{solidphase}}$ is the concentration of potassium on sediment in the top 10 mm (g Mg^{-1} soil), sed is the sediment yield on a given day (Mg), area_{hru} is the HRU area (ha), and $\epsilon_{K:\text{sed}}$ is the potassium enrichment ratio. The enrichment ratio is the concentration of potassium in the sediment divide the potassium concentration in the soil.

Concentration of the potassium absorbed to the solid phase is calculated by:

$$C_{\text{solidphase}} = \frac{100 \times K_p \times K_{s,\text{ly}}}{\text{SAT}_{\text{ly}} + K_p \times \rho_b \times \text{depth}_{\text{ly}}} \quad \text{【3.28】}$$

where $C_{\text{solidphase}}$ is the concentration of the potassium absorbed to the solid phase (mg kg^{-1} or g Mg^{-1}), K_p is the solid-liquid distribution coefficient ($(\text{mg kg}^{-1})/(\text{mg L}^{-1})$ or $\text{m}^3 \text{Mg}^{-1}$), $K_{s,ly}$ is the amount of potassium in the soil layer (kg ha^{-1}), SAT_{ly} is the amount of water in the soil layer at saturation ($\text{mm H}_2\text{O}$), ρ_b is the bulk density of the soil layer (Mg m^{-3}), and depth_{ly} is the depth of the soil layer (mm).

Potassium lag in surface runoff and lateral flow

In large subbasins with a time of concentration greater than one day, only a portion of the surface runoff and lateral flow will reach the main channel on the day it is generated, thus, potassium lag in surface runoff and lateral flow happens and potassium enters into streams after the day it is generated. SWAT incorporates a storage feature to lag a portion of the surface runoff and lateral flow release to the main channel. Potassium in the surface runoff and lateral flow are lagged as well. Once the potassium load in surface runoff and lateral flow is determined, the amount of potassium released to the main channel is calculated:

$$K_{\text{surf}} = (K'_{\text{surf}} + K_{\text{surstor},i-1}) \times \left(1 - \exp\left[\frac{-\text{surlag}}{t_{\text{conc}}}\right]\right) \quad \text{【3.29】}$$

$$K_{\text{lat}} = (K'_{\text{lat}} + K_{\text{latstor},i-1}) \times \left(1 - \exp\left[\frac{-1}{t_{\text{Tlat}}}\right]\right) \quad \text{【3.30】}$$

$$K_{\text{sed}} = (K'_{\text{sed}} + K_{\text{sedstor},i-1}) \times \left(1 - \exp\left[\frac{-\text{surlag}}{t_{\text{conc}}}\right]\right) \quad \text{【3.31】}$$

where K_{surf} is the amount of soluble potassium discharged to the main channel in surface runoff on a given day (kg ha^{-1}), K'_{surf} is the amount of surface runoff soluble potassium generated in HRU on a given day (kg ha^{-1}), $K_{\text{surstor},i-1}$ is the surface runoff soluble potassium stored or lagged from the previous day (kg ha^{-1}), K_{lat} is the amount of soluble potassium discharged to the main channel in lateral flow on a given day (kg ha^{-1}), K'_{lat} is the amount of lateral flow soluble potassium generated in HRU on a given day (kg ha^{-1}), $K_{\text{latstor},i-1}$ is the lateral flow potassium stored or lagged from the previous day (kg ha^{-1}), K_{sed} is the amount of absorbed potassium discharged to the main channel in surface runoff on a given day (kg ha^{-1}), K'_{sed} is the absorbed potassium loading generated in HRU on a given day (kg ha^{-1}), $K_{\text{sedstor},i-1}$ is the

absorbed potassium stored or lagged from the previous day (kg ha^{-1}), surlag is the surface runoff lag coefficient, t_{conc} is the time of concentration for the HRU (hrs) and TT_{lat} is the lateral flow travel time (days).

Potassium in the stream water layer

Solid-liquid distribution between suspended sediment and stream water

Potassium will partition into solid and liquid phases. The fraction of potassium in each phase is a function of the potassium's distribution coefficient and the reach segment's suspended solid concentration:

$$F_d = \frac{1}{1 + K_d \times \text{conc}_{\text{sed}}} \quad \text{【3.32】}$$

$$F_p = \frac{K_d \times \text{conc}_{\text{sed}}}{1 + K_d \times \text{conc}_{\text{sed}}} = 1 - F_d \quad \text{【3.33】}$$

where F_d is the fraction of total potassium in the liquid phase, F_p is the fraction of total potassium in the solid phase, K_d is the potassium distribution coefficient ($\text{m}^3 \text{g}^{-1}$), and conc_{sed} is the concentration of suspended solids in the water (g m^{-3}).

Setting from stream water layer to sediment layer

Potassium in the particulate phase may be removed from the water layer by settling. Settling transfers potassium from the water to the sediment layer. The amount of potassium that is removed from the water via settling is:

$$K_{\text{stl,wtr}} = \frac{v_s}{\text{depth}} \times F_p \times K_{\text{rchwtr}} \times TT \quad \text{【3.34】}$$

where $K_{\text{stl,wtr}}$ is the amount of potassium removed from the water due to settling (mg), v_s is the settling velocity (m day^{-1}), depth is the flow depth (m), F_p is the fraction of total potassium in the particulate phase, K_{rchwtr} is the amount of potassium in the water (mg), and TT is the flow travel time (days).

Outflow in stream

Potassium is removed from the reach segment in outflow. The amount of dissolved and particulate potassium removed from the reach segment in outflow is:

$$K_{\text{sol,o}} = Q \times \frac{F_d \times K_{\text{rchwtr}}}{V} \quad \text{【3.35】}$$

$$K_{\text{sorb,o}} = Q \times \frac{F_p \times K_{\text{rchwtr}}}{V} \quad \text{【3.36】}$$

where $K_{sol,o}$ is the amount of dissolved potassium removed via outflow (mg), $K_{sorb,o}$ is the amount of particulate potassium removed via outflow (mg), Q is the rate of outflow from the reach segment ($m^3 H_2O day^{-1}$), and V is the volume of water in the reach segment ($m^3 H_2O$).

Potassium in the stream sediment layer

Solid-liquid distribution in stream sediment layer

The fraction of potassium in solid and liquid phases is calculated:

$$F_{d,sed} = \frac{1}{1+(1-\phi) \times \rho_s \times K_d} \quad \text{【3.37】}$$

$$F_{p,sed} = 1 - F_{d,sed} \quad \text{【3.38】}$$

where $F_{d,sed}$ is the fraction of total sediment potassium in the dissolved phase, $F_{p,sed}$ the fraction of total sediment potassium in the particulate phase, ϕ is the porosity, ρ_s is the particle density ($g m^{-3}$), and K_d is the potassium distribution coefficient ($m^3 g^{-1}$). The potassium distribution coefficient used for the stream water layer is also used for the sediment layer.

Resuspension of potassium absorbed in sediment layer into stream water layer

Potassium in the sediment layer is available for resuspension. The amount of potassium that is removed from the sediment via resuspension is:

$$K_{rsp,wtr} = \frac{V_r}{depth} \times K_{rchsed} \times TT \quad \text{【3.39】}$$

where $K_{rsp,wtr}$ is the amount of potassium removed via resuspension (mg), V_r is the resuspension velocity ($m day^{-1}$), $depth$ is the flow depth (m), K_{rchsed} is the amount of potassium in the sediment (mg), and TT is the flow travel time (days). Potassium removed from the sediment layer by resuspension is added to the water layer.

Diffusion between stream sediment layer and stream water layer

Potassium in the dissolved phase is available for diffusion. Diffusion transfers potassium between the water and sediment layers. The direction of movement is controlled by the potassium concentration. Potassium will move from areas of high concentration to areas of low concentration. The amount of potassium that is transferred between the water and sediment by diffusion is:

$$K_{dif} = \left| \frac{V_d}{\text{depth}} \times (F_{d, \text{sed}} \times K_{rchsed} - F_d \times K_{rchwtr}) \times TT \right| \quad \text{【3.40】}$$

where K_{dif} is the amount of potassium transferred between the water and sediment by diffusion (mg), V_d is the rate of diffusion or mixing velocity (m day⁻¹), depth is the flow depth (m), $F_{d, \text{sed}}$ is the fraction of total sediment potassium in the dissolved phase, K_{rchsed} is the amount of potassium in the sediment (mg), F_d is the fraction of total water layer potassium in the dissolved phase, K_{rchwtr} is the amount of potassium in the water (mg), and TT is the flow duration (days). If $F_{d, \text{sed}} \times K_{rchsed} > F_d \cdot K_{rchwtr}$, K_{dif} is transferred from the sediment to the water layer. If $F_{d, \text{sed}} \times K_{rchsed} < F_d \cdot K_{rchwtr}$, K_{dif} is transferred from the water to the sediment layer.

Burial in stream sediment layer

Potassium in the sediment layer may be lost by burial. The amount of potassium that is removed from the sediment via burial is:

$$K_{bur} = \frac{V_b}{D_{sed}} \times K_{rchsed} \quad \text{【3.41】}$$

where K_{bur} is the amount of potassium removed via burial (mg), V_b is the burial velocity (m day⁻¹), D_{sed} is the depth of the active sediment layer (m), and K_{rchsed} is the amount of potassium in the sediment (mg).

Mass balance

The processes described above can be combined into mass balance equations for the well-mixed reach segment and the well-mixed sediment layer:

$$\Delta K_{rchwtr} = K_{in} - (K_{sol, o} - K_{sor, b, o}) - K_{stl, wtr} + K_{rsp, wtr} \pm K_{dif} \quad \text{【3.42】}$$

$$\Delta K_{rchsed} = K_{stl, wtr} - K_{rsp, wtr} - K_{bury} \pm K_{dif} \quad \text{【3.43】}$$

where ΔK_{rchwtr} is the change in potassium mass in the water layer (mg), ΔK_{rchsed} is the change in potassium mass in the sediment layer (mg), K_{in} is the potassium added to the reach segment via inflow (mg), $K_{sol, o}$ is the amount of dissolved potassium removed via outflow (mg), $K_{sor, b, o}$ is the amount of particulate potassium removed via outflow (mg), $K_{stl, wtr}$ is the amount of potassium removed from the water due to settling (mg), $K_{rsp, wtr}$ is the amount of potassium removed via resuspension (mg), K_{dif} is the amount of potassium transferred between the water and sediment by

diffusion (mg), K_{bury} is the amount of potassium removed via burial (mg).

3.4.4 Model calibration and validation

In SWAT, the HRU is the smallest unit of spatial disaggregation. As a watershed is divided into HRUs based on slope, soil, and land use, a distributed parameter such as hydraulic conductivity can potentially be defined for each HRU. The physical factors such as soil, land use, slope are the identifiers of the parameter in HRU. Several calibration techniques have been developed for SWAT, including manual calibration procedures and automated procedures. Automated procedures using SWAT-CUP includes the shuffled complex evolution method and other common methods (Abbaspour, 2007). Manually calibration was also commonly used where users can manually adjust parameters based on their experiences.

When users are faced with data-limited situations, calibration and validation using a single gauge was commonly conducted. When users have available data from multiple sites, calibration and validation using multi-site data can be conducted (e.g., Arnold et al., 2001; Van Liew et al., 2003; Cao et al., 2006; Parajuli et al., 2009). The definitions of the single site and multi-site calibration are shown in Table 3.3

Calibration and validation are typically performed by splitting the available observed data into two periods: one for calibration, and another for validation. Data may also be available spatially, with available data at one site or all the sites assigned to the calibration phase and correspondingly performing the validation at these sites or more other gauges within the watershed (Abbaspour, 2007).

In this study, single site calibration using data of the main outlet was conducted, the calibration and validation are performed by splitting the available observed data into two periods: one for calibration, and another for validation. For streamflow, 2003-2005 was selected as calibration period, and 2006-2008 was selected as validation period. For sediment and potassium load, 2003 was selected as calibration period, and 2004 was selected as validation period.

Table 3.3 Definitions of different calibration approaches

Calibration approaches		Definition
Available data	Single site	Using data from a single site for parameters calibration.
	Multi-site	Using data from multiple sites for parameters calibration.

3.4.5 Model performance evaluation

The accuracy of SWAT simulation results was determined by examining the coefficient of determination (R^2), the Nash and Sutcliffe (1970) efficiency (E_{NS}) and relative error (Re). The R^2 value is an indicator of the strength of the linear relationship between the observed and simulated values. The E_{NS} simulation coefficient indicates how well the plot of observed values versus simulated values fits the 1:1 line. If the R^2 and E_{NS} values are less than or very close to zero, the model prediction is unacceptable or poor. If the values are one, then the model prediction is perfect (Santhi et al., 2001). Re also indicates how close the observed values versus the simulated values are. Re can range from zero to a very large value, with zero representing perfect agreement between the model and real data. Essentially, when the model efficiency R^2 and E_{NS} are close to one, and when the model efficiency Re is close to zero, the models are considered more accurate.

R^2 is statistically defined as

$$R^2 = \left\{ \frac{\sum_{i=1}^n (X_{oi} - \overline{X_{oi}})(X_{si} - \overline{X_{si}})}{[\sum_{i=1}^n (X_{oi} - \overline{X_{oi}})^2]^{0.5} [\sum_{i=1}^n (X_{si} - \overline{X_{si}})^2]^{0.5}} \right\} \quad \text{【3.44】}$$

E_{NS} is statistically defined as

$$E_{NS} = 1 - \frac{\sum_{i=1}^n (X_{oi} - X_{si})^2}{\sum_{i=1}^n (X_{oi} - \overline{X_{oi}})^2} \quad \text{【3.45】}$$

Re (in percentage) at the gauge locations can be derived from

$$\text{Re}(\%) = \left| \frac{\sum_{i=1}^n X_{si} - \sum_{i=1}^n X_{oi}}{\sum_{i=1}^n X_{oi}} \right| * 100 \quad \text{【3.46】}$$

where X_{oi} is the observed data on day i , X_{si} is the simulated output on day i , $\overline{X_{oi}}$ is the average measured value during the study period, and n is total number of observed data.

4 SWAT modeling of water flow

4.1 Introduction

The Soil and Water Assessment Tool (SWAT) has been developed and widely used to predict impacts of land management practices on water, sediment, and agricultural chemical yields in large complex watersheds with varying soils, land use, and management conditions over long periods of time (Arnold et al. 1993; Van Griensven et al., 2012). The SWAT model incorporates weather, hydrology, sedimentation, soil temperature, plant growth, nutrients, pesticides and land management. The hydrology subcomponent within SWAT is the driving force behind other components and subcomponents. Evaluation of the hydrological performance needs to be performed prior to attempting any water quality simulations (Levesque et al., 2008). Thus, SWAT has been widely evaluated by researchers at daily, monthly and annual scales for watersheds with different characteristics. Some of these evaluations have indicated that SWAT gives an acceptable prediction on daily, monthly and annual streamflows (Spruill et al., 2000; Bingner, 1996), but some researches indicated that it cannot satisfactorily predict daily streamflows (Bosch et al., 2004) and annual streamflows (Arnold et al., 1999). These evaluations were conducted in areas where streamflows were dominantly generated from rainfall events, with negligible contributions from melting snow.

Snow plays an important role in the water cycle (Adam et al., 2009). Snowmelt hydrology is a very important component for applying SWAT. Some studies reported excellent SWAT snowmelt discharge estimates. For instance, Abbaspour et al. (2007) successfully simulated the snowmelt streamflow in their SWAT simulation of the Thur watershed in Switzerland. Wang and Melesse (2005) reported acceptable model performance of the SWAT model's snowmelt hydrology for simulating spring streamflow generated from melting snow for the Wild Rice River watershed, located in northwestern Minnesota. However, some studies reported SWAT's poor performance in modeling watersheds where streamflows are predominantly generated from melting snow in spring, while streamflows in summer and fall are predominantly generated

from rainfall runoff. Qi and Grunwald (2005) found that SWAT did not predict winter season precipitation-runoff events well for the Sandusky River watershed. Chanasyk et al. (2003) found that SWAT was not able to replicate snowmelt dominated runoff for three small grassland watersheds in Alberta that were managed with different grazing intensities. Snowmelt and rainfall runoff are two very different kinds of hydrologic processes. Compared with the rainfall runoff process, snowmelt is a slow and gradual process, and melting snow is treated as rainfall with zero energy in SWAT (Arnold et al., 1993). For watersheds where melting snow is the dominant source for streamflows in spring but rainfall runoff is the dominant source for streamflows in summer and fall, which is the situation for the study watershed Shibetsu river watershed, located in eastern Hokkaido Japan (Figure 3.1). The SWAT model needs to be calibrated for all four seasons because the hydrologic conditions in one season may influence the subsequent season. In the Shibetsu river watershed, snowpack accumulates in winter (December to February) and melts in the following spring, so the streamflows in spring (March to May) are predominately generated from melting snow, whereas the streamflows in summer (June to August) and fall (September to November) are mainly generated from rainfall runoff. The watershed in this study differs from the watershed studied by others (Wang and Melesse, 2005; Abbaspour et al., 2007) because it is characterised by volcanogeneous soils with high hydraulic conductivities and porosities. Surface runoff due to infiltration excess is probably low and dominating flow processes are likely to happen in the subsurface during snowmelt and rainfall seasons (Blume, 2008). It is important to set up a SWAT model to simultaneously predict streamflows from these two hydrological processes with an acceptable accuracy in Shibetsu river watershed.

The main objectives of this study were to understand the hydrological processes and to evaluate the performance of the SWAT model for simulating snowmelt seasons hydrology in the Shibetsu river watershed.

4.2 Materials and methods

4.2.1 Study site

Watershed description

The Shibetsu river watershed (main outlet SRW, 672 km²) is located in eastern Hokkaido, Japan. The detail descriptions of SRW are shown in section 3.1. The locations of the weather stations and outlets, slope, land use, and soil types in the Shibetsu river watershed are shown in Figure 3.1. There is snowfall from December to March, snowmelt from April to May, and rainfall from June to November in Shibetsu river watershed. The snowmelt season from April to May was separated from other periods to study the snowmelt hydrology. It was observed that there was large contribution of snowmelt water flows to streams as groundwater in June, so June was add to the snowmelt season to study the snowmelt hydrology in this chapter.

Instrumentation and sampling

The location of the measurement outlet SRW is shown in Figure 3.1. Hourly stream water table data (2003–2008) were obtained for the main SRW outlet from the Water Information System (Japan Ministry of Land, Infrastructure, and Transport, <http://www1.river.go.jp/>). Hourly streamflow (Q) was calculated from the stream water table (H) and calibrated H-Q equations (quadratic curve), which define the relationships between streamflow and the water table. The H-Q curve, which was obtained from 5-11 measurements of stream velocity and river section over three periods (monthly base flow, daily snowmelt, and daily storm events), yielded a value of R^2 greater than 0.95 ($P < 0.001$) every year, showing the reliability of the estimated river discharge (Hayakawa et al. 2009; Jiang et al. 2011). Daily streamflow was calculated by integrating the hourly streamflow.

4.2.2 Model description

Model input

In this study, daily weather data, digital elevation model data, land uses and soils of the study area are detailed in section 3.1. There are five weather stations used for this study, all located outside the mountainous area with high elevations (29% of the

watershed) which might have significantly different weather conditions from other areas. The Soil Conservation Service Runoff Curve Number estimation method and the Penman-Monteith evapotranspiration estimation method were used in this study. In the SRW catchment, the drainage area does not correspond to the hydrological watershed and it is recognized that it receives external groundwater (EXT) from other watersheds. The EXT contribution was simplified as 1.38 mm day^{-1} without temporal and spatial variation and added to the total water entering stream of the SWAT model.

Model calibration and validation

Previous studies by Jiang et al. (2011) suggested the sensitive hydrological parameters (Table 4.1) in Shibetsu river watershed from sensitivity analysis with a Latin hypercube one-at-a-time (LH-OAT) method (Van Griensven et al., 2006). For the parameters (Initial SCS runoff curve number for moisture condition, CN2; Baseflow alpha factor, ALPHABF and Groundwater delay, GW_DELAY), the physical factor land use were considered during calibration. Parameters calibration was conducted using the observed streamflow from single site of main outlet SRW. Firstly, the calibration of flow was performed manually to obtain a good match between the observed and simulated values. Then, the snow-related parameters (snowfall temperature, SFTMP; snowmelt temperature, SMTMP; maximum snowmelt rate, SMFMX; minimum snowmelt rate, SMFMN; and snowpack temperature lag factor, TIMP) were calibrated using SWAT CUP with the SUFI-2 calibration and uncertainty analysis routine (Abbaspour, 2007). The streamflow was calibrated from 2003 to 2005 and validated from 2006 to 2008.

Model performance evaluation

The accuracy of the results of the SWAT simulation were determined by examination of the coefficient of determination (R^2) and the Nash and Sutcliffe (1970) model efficiency coefficient (E_{NS}). The definitions of the two statistic criteria were explained in section 3.4.5.

4.3 Results and discussion

4.3.1 Watershed Hydrology.

Table 4.1 presents the calibrated parameters for streamflow. Figure 4.1 graphically illustrates the comparison between the observed and simulated daily streamflow at the main outlet of the SRW. The simulated streamflow followed a similar trend to the observed streamflow. The statistical performance of the SWAT for daily streamflow estimation was satisfactory (calibration period: $R^2 = 0.60$, $E_{NS} = 0.40$ and $Re = 14\%$; validation period: $R^2 = 0.87$, $E_{NS} = 0.61$ and $Re = 7\%$). The SWAT model yielded a mean annual streamflow of 1140 mm for the period studied (2003-2008), which was close to the observed value of 1054 mm. However, the simulated peak discharge was underestimated during some heavy rainfall periods such as events in August 2003 and October 2006. This was primarily due to the surface runoff was underestimated for these events in this study. It might be because precipitation duration and intensity are not being considered by the soil conservation services (SCS) curve number (CN) method (SCS, 1972) for simulation of streamflow in SWAT model as reported by Phomcha et al. (2011). This limitation might be more profound for the heavy rainfall events.

Mean annual rainfall for the total simulation period (2003 to 2008) over the area of the catchment was 1055 mm. Simulated results showed that about 430 mm (41 %) was removed through evapotranspiration (ET). Simulated mean annual water yield was 1140 mm, including surface runoff of 30 mm (2.5 %), lateral flow of 145 mm (13 %) and groundwater recharge of 965 mm (84.5 %), including EXT of 480 mm (42 %). In the SRW, most stream water was recharged by subsurface flow throughout the year. The computed water balance components indicated low surface runoff (2.5 % of total water yield) that subsequently caused landscape erosion. Since the watershed studied is characterised by volcanogeneous soils with high hydraulic conductivities (Table 2) and porosities, surface runoff due to infiltration excess is probably of little importance and dominating flow processes are likely to happen in the subsurface (Blume, 2008). The large subsurface storage can retain most of the incident rainfall during events (>90 %, 54

often even >95 %) as reported by Blume (2008).

Table 4.1 Calibrated parameters for stream streamflow

No	Parameters	Definition of parameters	Fitted value
1	CN2.mgt	Initial SCS runoff curve number for moisture condition	Forest (36) Pasture (45) Urban (55)
2	ALPHA_BF.gw	Baseflow alpha factor (days)	Forest (0.02) Pasture (0.5) Urban (0.5)
3	REVAPMN.gw	Threshold depth of water in the shallow aquifer for revap to occur (mm)	210
4	SOL_AWC.sol (all soil layers)	Available water capacity of the soil layer (mm H ₂ O mm soil)	0.10
5	ESCO.hru	Soil evaporation compensation factor	0.29
6	CANMX.hru	Maximum canopy storage (mm H ₂ O)	39
7	GW_DELAY.gw	Groundwater delay (days)	Forest (3) Pasture (1) Urban (1)
8	CH_N2.rte	Surface runoff lag coefficient	0.09
9	SFTMP.bsn	Snowfall temperature (°C)	1.93
10	SMTMP.bsn	Snowmelt base temperature (°C)	1.57
11	SMFMX.bsn	Maximum melt rate for snow during years (mm °C ⁻¹ day ⁻¹)	1.19
12	SMFMN.bsn	Minimum melt rate for snow during years (mm °C ⁻¹ day ⁻¹)	0.23
13	TIMP.bsn	Snowpack temperature lag factor	0.59
14	SURLAG.bsn	Manning's "n" value for the tributary channels	1

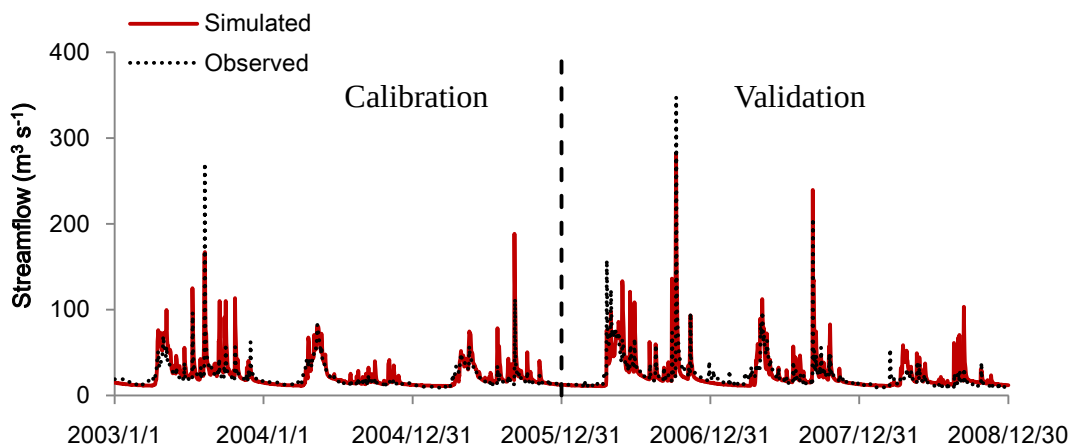
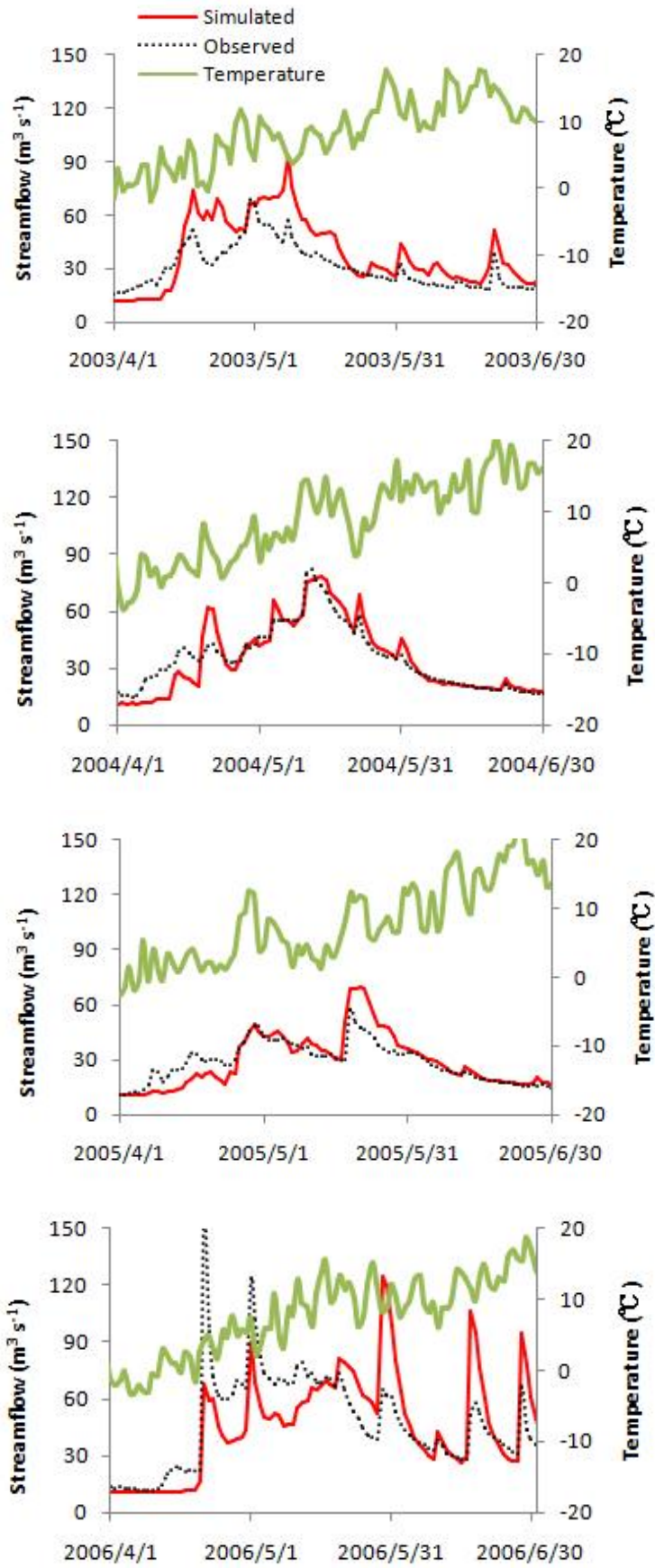


Figure 4.1 Observed and simulated streamflow comparison from 2003-2008

4.3.2 Snowmelt hydrology

Figure 4.2 shows the simulated and observed daily streamflows from the snowmelt process during from 2003 to 2008 at SRW. The SWAT model simulated daily streamflows exhibited similar variation trend with observed daily streamflows, which is indicated by the high R^2 values (above 0.45) (Table 5.2). SWAT model produced satisfactory performances for the years 2004, 2005 and 2007, with E_{NS} values of 0.82, 0.51 and 0.71, respectively. SWAT model produced acceptable performance for year 2006, with E_{NS} value of 0.31. However, SWAT model produced poor performances for years 2003 and 2008, with E_{NS} values of -0.14 and -2.83, respectively. The acceptable and poor model performance might be because the assumed EXT as constant value (1.38 mm day^{-1}) caused biased estimation of streamflow. For example, in April, 2006, the streamflow was significantly underestimated, which might be because the EXT was under assumed. While the streamflow was significantly overestimated in June, which could be due to the EXT was over assumed. The five weather stations used in this study were all located outside the mountainous areas, the temperature and precipitation in the forest area were unknown, and this might resulted in poor estimation of the snowmelt processes. According to Sawano et al. (2005), the precipitation in the forested area was about 330 mm higher than that in the agricultural area in Japan during 1979 to 2003. Another study conducted in Hokkaido, northern Japan, showed that the precipitation in the mountainous area was significantly correlated with elevation (Yamada et al., 1995). In Shibetsu, the forest land use was mainly located under steep topography, applying the temperature band and precipitation band maybe can improve the model performance in future studies. Fontaine et al. (2002) reported that setting precipitation and temperature as a function of elevation bands can result in greatly improved streamflow estimates for the mountainous area.



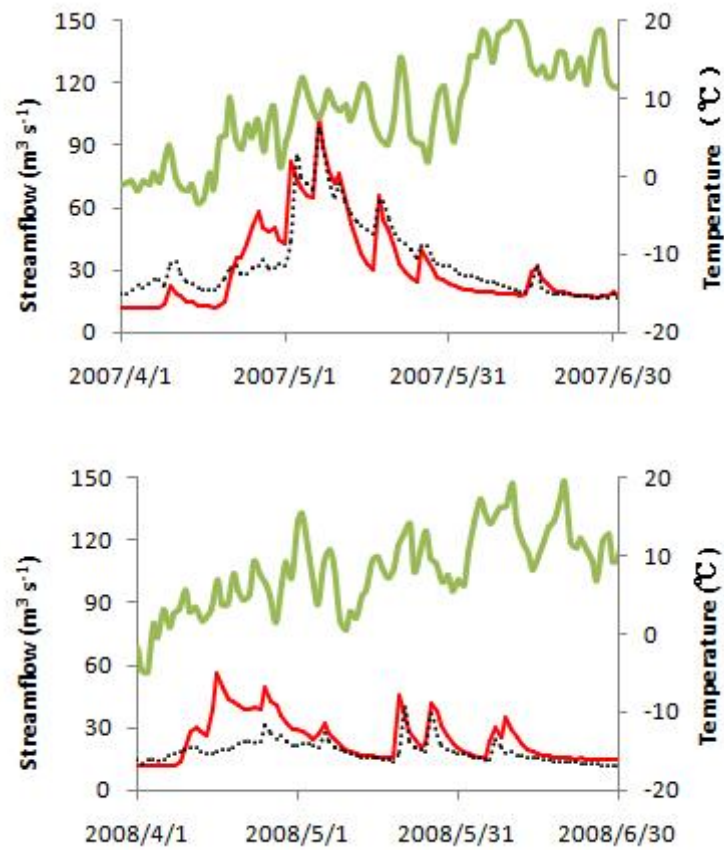


Figure 4.2 Observed and simulated streamflow comparison during snowmelt season

Table 4.2 Model performance during snowmelt season

Time	R^2	E_{NS}
2003 (March-June)	0.76	-0.14
2004 (March-June)	0.88	0.82
2005 (March-June)	0.79	0.51
2006 (March-June)	0.46	0.32
2007 (March-June)	0.79	0.71
2008 (March-June)	0.49	-2.83

4.4 Conclusion

In Shibetsu river watershed, streamflows in spring are mainly generated from melting snow although streamflows in summer and autumn are generated from rainfall. Hydrology in rainfall events and snowmelt season was evaluated using the Soil and Water Assessment Tool (SWAT). In the Shibetsu river catchment, the drainage area

does not correspond to the hydrological watershed and it is recognized that it receives external groundwater (EXT) from other watersheds. The EXT contribution was simplified as 1.38 mm day^{-1} without temporal and spatial variation and added to the total water entering stream of the SWAT model. SWAT model performance results indicated that for the study watershed, the SWAT model had a good performance on simulating the daily streamflows. When the snowmelt season was analyzed alone, the daily streamflows in spring, which were predominantly generated from melting snow, could be predicted with an acceptable accuracy, and the corresponding daily streamflows could be simulated very well during most of the studied years. In some years the SWAT model cannot describe the daily streamflows well, which could be attributed to the uncertainty in the weather data and assumed EXT.

5 SWAT simulation of sediment, PON and POP export

5.1 Introduction

Sediment and sediment-bound pollutants including pesticides, particulate nutrients, heavy metals and other toxic substances transported from the land surface to stream networks are responsible for reservoir sedimentation and aquatic habitat degradation (Haag et al., 2001; Kerr et al., 2011; Boithias et al., 2011, 2013; Cerro et al., 2013 a and b). Several adverse economic and environmental impacts due to the damaging effects of soil erosion have been reported. The on-site effect of soil erosion in terms of declining soil fertility and decreased agricultural yields are well known around the world. Environmental consequences are primarily off-site effects due to the pollution of natural waters (Lal, 1998). Understanding the dynamics of sediment transfer from land to watercourses and quantifying sediment yields are essential for controlling land soil erosion and implementing appropriate mitigation practices to reduce stream sediment and associated pollutant loads, and hence improve surface water quality downstream (Heathwaite et al., 2005). Downstream erosion and sedimentation implications are of increasing interest for catchment management, such as the design of dam reservoirs, river restoration, the design of stable channels and the protection of fish and wildlife habitat.

It has remained a challenge to estimate changes in sediment yield over time in a catchment owing to the complexity of the processes involved in the detachment and transport of fluvial sediment. Different approaches have been adopted for sediment yield estimation. The most reliable method for sediment load estimation is direct measurement at catchment level. Sediment concentrations are usually measured infrequently because very frequent monitoring over the long term is costly. It has also been noted that a sediment sampling strategy should be designed to capture high sediment concentrations for long-term monitoring to provide better results (Thomas, 1988).

The applications of empirical models for estimating sediment load have shown promise. Estimation of sediment load is commonly achieved by establishing a sediment

rating curve. Empirical rating curves describing relationships between sediment load and instantaneous water streamflow are often used. Some researchers have suggested that an excellent sediment rating curve could be constructed using a limited set of data (Gao, 2008). Sediment rating curves are useful in predicting sediment yield, but they are site specific and have limitations when it comes to interpreting erosion processes (landscape erosion and in-stream erosion/sedimentation). Distributed and process-based watershed models are capable of capturing these complex processes both spatially and temporally. This category of models can be used to provide an enhanced understanding of the relationship between hydrologic processes, landforms, land management, soil factors and erosion/sedimentation (Van Rompaey et al., 2001; Easton et al., 2010). Many of the model parameters have a physical meaning and can be measured in the field, and therefore model validation can be concluded on the basis of a short field survey and a short time series of meteorological and hydrological data. Various hydrological models have been proposed to predict sediment export to rivers, such as the European soil erosion model (EUROSEM) (Morgan et al., 1998), the water erosion prediction project (WEPP) (Nearing et al., 1989) and the Soil and Water Assessment Tool (SWAT) (Neitsch et al., 2005).

The sediment yield estimation model used in this study is the SWAT model. It is a comprehensive process-based model that simulates water, sediment and chemical fluxes in watersheds under varying climatic conditions, soil properties, stream channel characteristics, land use and agricultural management (Jayakrishnan et al., 2005; Talebizadeh et al., 2010). The SWAT model has been applied to enhance understanding of sediment loss and transport processes over a wide range of environments around the world (Oeurng et al., 2011). For sediment yield modelling, Mukundan et al. (2010) examined the suitability of SWAT at the North Fork Broad River catchment located in the Piedmont region of Georgia, and their results suggested that the SWAT model is a better substitute than the sediment rating curve for estimating sediment yield. Many researchers have reported that the SWAT model predicted reasonable results for sediment yield estimation (especially on monthly and yearly timescales) when provided

accurate input data and model parameterisation (Chu et al., 2004; Saghafian et al., 2012).

The SWAT model can estimate soil erosion from the landscape and in-stream depositional and degrading processes. The sediment yield from the landscape is calculated using the modified universal soil loss equation (MUSLE) (Williams, 1975). Sediment deposition and degradation in the stream channel are both calculated during sediment routing. The maximum amount of sediment that can be transported from a reach segment during the channel sediment routing is determined by the modified Bagnold's equation (Bagnold, 1977). However, both MUSLE and the modified Bagnold's equation in the SWAT model are empirical equations, therefore SWAT may not produce accurate results in all situations. As a surface hydrological model, SWAT also has limited applicability in complex hydrological environments, such as non-conservative watersheds where the drainage area does not correspond to the hydrological watershed. Non-conservative watersheds may either lose internal groundwater to neighbouring watersheds or gain external groundwater (EXT) originating from outside the watershed. The evaluation of the hydrological component of SWAT completed in previous studies has pointed out that SWAT has no mechanism to account for external water (EXT) contributions through subsurface flow from outside the watershed (Chu et al., 2004; Salerno and Tartari, 2009). Consequently, the SWAT model cannot consider the effect of EXT on sediment routing in reaches where EXT finally enters. Jiang et al. (2011, 2014) examined the possibility of using the SWAT model in a non-conservative watershed, the Shibetsu river watershed (SRW, 672 km², Hokkaido, Japan), which is recognised to receive external groundwater. They reported that the SWAT model could successfully be used to understand components of stream streamflow and nitrate export by assuming EXT as a constant value (1.38 mm day⁻¹, estimated from a long-term annual water balance budget) and including it as a point source of water and nitrate in the model. However, the suitability of the SWAT model for estimating sediment and associated particulate organic nitrogen (PON) and particulate organic phosphorous (POP) yields and for understanding the soil erosion mechanism by

taking into account the EXT contribution to streams is still unclear in the Shibetsu river watershed.

The main objective of this study was to apply the SWAT model by taking into account the EXT contribution to channel water so as to estimate sediment and associated particulate organic nitrogen (PON) and particulate organic phosphorous (POP) yields accurately and to understand soil erosion mechanisms in Shibetsu river watershed containing forest and agriculture, which is dominated by volcanic soils with an external groundwater source.

5.2 Materials and methods

5.2.1 Study site description

The Shibetsu river watershed (main outlet SRW, 672 km²) is located in eastern Hokkaido, Japan. The detail descriptions of SRW are shown in section 3.1. The locations of the weather stations and outlets, slope, land use, and soil types in the Shibetsu river watershed are shown in Figure 3.1.

5.2.2 Instrumentation and sampling

For the whole watershed outlet of the SRW, hourly stream water table data (2003-2008) was obtained from the Water Information System (Ministry of Land, Infrastructure and Transport, Japan). Then it was used to calculate the daily streamflow based on the method introduced in section 3.2. Water samples were collected for the analysis of suspended sediments, particulate organic nitrogen (PON) and particulate organic phosphorous (POP) concentrations. A detailed description of the methods of water sample analysis is shown in section 3.3.

5.2.3 Model description and model input

The input data (weather data, DEM, land use and soil) required by SWAT model is shown in section 3.1. It should be note that, for this non-conservative environment, the EXT flow into the subbasin channel from each HRU was added to the total water entering the channels in the SWAT model in this study, which is explained in section 3.4.1.

The SWAT model with EXT was used to estimate sediment yield at the main outlet of

SRW. Then, based on the estimated sediment yield and its relationships with PON and POP yields, PON and POP yields were estimated further. The SWAT results were investigated and compared with the observed values to evaluate its performance for estimating sediment and PON and POP yields in the SRW.

5.2.4 Model calibration and validation

The SWAT model was first calibrated using SWAT CUP with the SUFI-2 calibration and uncertainty analysis routine (Abbaspour, 2007). Then the calibration of flow and sediment was performed manually to obtain a good match between the observed and simulated values. Previous study by Phomcha et al. (2011) showed the sensitive sediment-related parameters were used in this study (Table 5.1). The streamflow was calibrated from 2003 to 2005 and validated from 2006 to 2008, as explained in section 4.2.2. The SWAT model was further used for sediment yield calibration following completion of the streamflow calibration process. The USLE soil erodibility factor (K_{USLE}) was calculated from an equation proposed by Williams (1995) based on clay, silt, sand and organic carbon contents in soil (Table 5.2). The USLE topographic factor (LS_{USLE}) based on SLSUBBSN.hru and HRU_SLP.hru was automatically calculated from the GIS interface in the SWAT model. Other model parameters were calibrated within their acceptable ranges to match the simulated sediment loadings with the observed loadings (Table 5.1). Observed sediment loads were used to calibrate SWAT in 2003 and the model was validated in 2004. The study period for sediment was only two years (2003-2004) because most samples were collected in these two years. Note that the calibration and validation periods were short due to the lack of long-term observed data. Longer calibration and validation periods would provide more confidence in the model parameters.

Table 5.1 Parameters for sediment calibration at the Shibetsu river watershed

No	Parameter	Definition of parameters	Fitted value
1	USLE_C(FRST)	Minimum value for the cover and management factor for the land cover	0.01
2	USLE_C(PAST)	Minimum value for the cover and management factor for the land cover	0.02
3	USLE_P(FRST)	USLE support practice factor	0.9
4	USLE_P(PAST)	USLE support practice factor	0.85
5	CH_EROD	Channel erodibility factor	0.01
6	CH_COV	Channel cover factor	1
7	PRF	Peak rate adjustment factor in main channel	0.56
8	SPCON	Coefficient in sediment transport equation	0.01
9	SPEXP	Exponent in sediment transport equation	2

Table 5.2 Soil saturated hydraulic conductivity (K) and USLE soil erodibility (K_{USLE})

Soil type	FAO soil type	K (mm hr ⁻¹)	K_{USLE}
Peat soil	Histosols	59	0.24
Regosolic kuroboku soil	Vitric Andosols	123	0.23
Brown forest soil	Cambisols	46	0.18
Brown lowland soil	Haplic Fluvisols	122	0.23
Regosol soil	Regosols	151	0.19
Gray lowland soil	Gleyic Fluvisols	75	0.25
Kuroboku soil	Silandic Andosols	114	0.22

5.2.5 Model performance evaluation

The accuracy of SWAT simulation results was determined by examining the coefficient of determination (R^2), the Nash and Sutcliffe (1970) efficiency (E_{NS}) and relative error (Re), as introduced in section 3.4.5.

5.3 Results and discussion

5.3.1 Particulate organic nitrogen (PON) and particulate organic phosphorous (POP)

The partition of dissolved and particulate nitrogen and phosphorous in 2003, 2004 and 2007 is shown in Table 5.3. These three years data were analysed because most of the PON and POP analysis from the water samples were conducted during this period. Results showed that PON accounted for about 10 %, 20 %, and 70 % of total nitrogen during snowfall (December to March), snowmelt (April to May) and rainfall (June to November) seasons respectively. PON partitioned only 4 % of the total nitrogen during

the snowfall season, but its partition increased to 23 % during the snowmelt season and to 32 % during the rainfall season. POP is the main form of phosphorous, which accounted for 92 % of total phosphorous during the snowfall season, and it decreased to 67 % and 64 % during the snowmelt and rainfall seasons respectively. These results indicate that PON and POP are important forms of nitrogen and phosphorous loss from the land that need to be quantified and understood. In this study, a significant linear relationship was found between PON, POP and sediment concentration (Figure 5.1). It indicates that PON and POP were mainly transported with suspended sediment, because particulate nutrient losses from land to rivers were mainly caused by land surface soil erosion.

Table 5.3 Partition of dissolved and particulate nitrogen and phosphorous for snowfall, snowmelt and rainfall seasons during 2003, 2004 and 2007

Hydrological events		TN	TDN	PON	TP	TDP	POP
		Concentration (mg L ⁻¹)					
Snowfall season (Dec-Mar) N=17	Mean	1.042	1.004	0.039	0.028	0.002	0.026
	%		96 %	4 %		8 %	92 %
Snowmelt season (Apr-May) N=139	Mean	1.264	0.972	0.292	0.058	0.019	0.039
	%		77 %	23 %		33 %	67 %
Rainfall season (Jun-Nov) N=431	Mean	1.349	0.916	0.433	0.083	0.030	0.054
	%		68 %	32 %		36 %	64 %

TN: total nitrogen, TDN: total dissolved nitrogen, PON: particulate organic nitrogen, TP: total phosphorous, TDP: total dissolved phosphorous, POP: particulate organic phosphorous

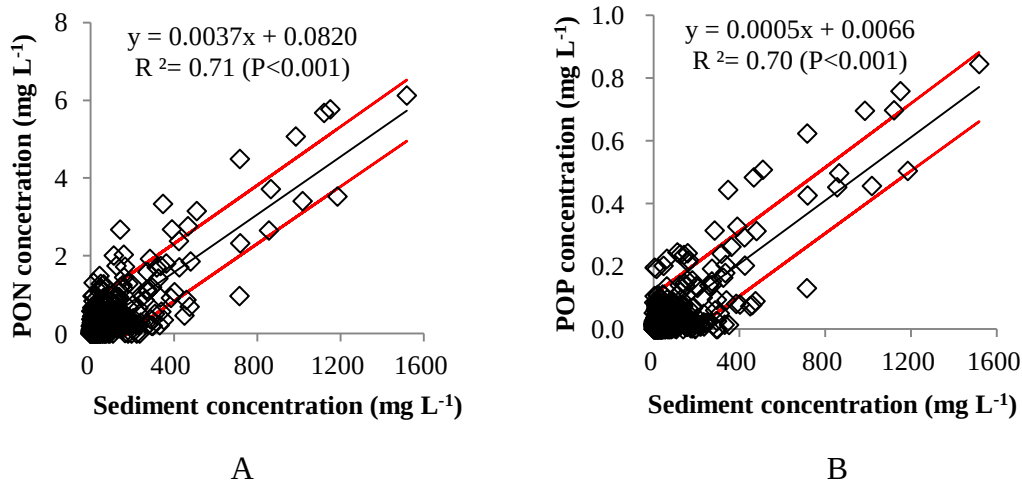


Figure 5.1 Relationships between measured instantaneous sediment and (A) particulate organic nitrogen (PON), and (B) particulate organic phosphorous (POP) concentration.

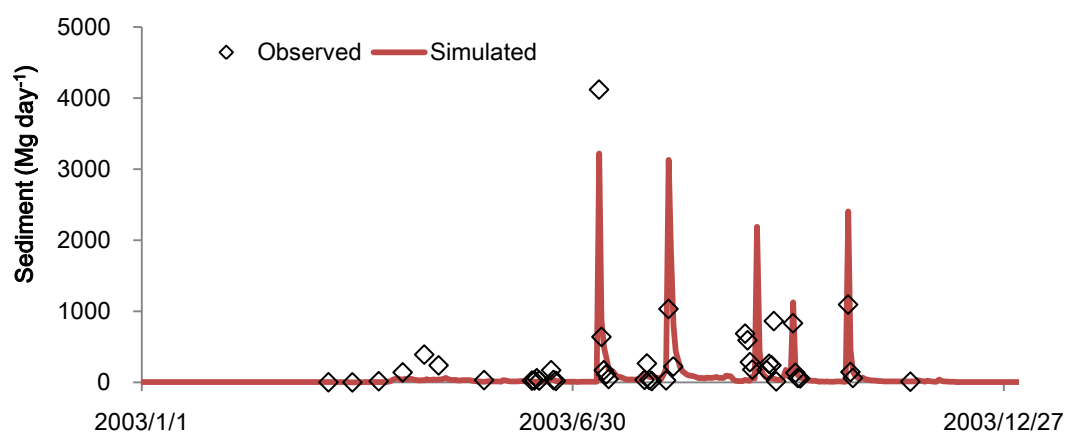
Red line shows 95 % prediction interval

5.3.2 Modelling performance of SWAT for sediment yield estimation

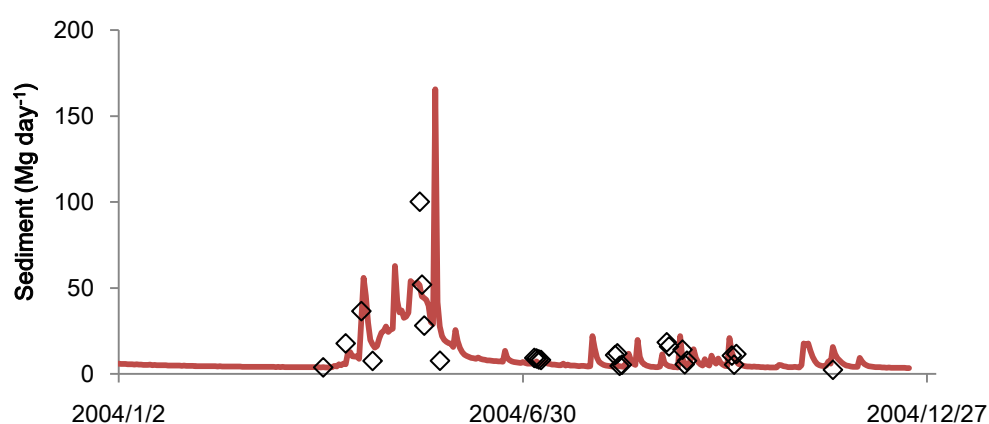
Figure 5.2 shows the comparison between simulated daily sediment loads of the SWAT model and the observed values. Table 5.1 presents the calibrated parameters for sediment yield simulation. Figure 5.2 generally indicates that the simulated daily sediment loads and the observed values are comparable. The SWAT model performance statistics are shown in Table 5.4, yielding R^2 of 0.62, E_{NS} of 0.48 and Re of 10 % in the calibration period, and R^2 of 0.64, E_{NS} of 0.61 and Re of 14 % in the validation period (Table 5.4). Overall, the SWAT model was able to simulate sediment yield with reasonable accuracy on a daily time step. Simulated annual sediment yield from 2003 to 2008 ranged from 5 to 45 Mg km⁻² yr⁻¹ (annual mean of 24 Mg km⁻² yr⁻¹) (Figure 5.3A). The SWAT model predicted acceptable model performance with a short time step (daily), indicating that this model can be considered an appropriate tool for estimating sediment yield in the SRW.

The snowfall, snowmelt and rainfall seasons contributed around 10 %, 20 %, and 70 % respectively to total sediment and associated PON and POP yields. Rainfall season play an important role in sediment transport, as most of the annual sediment yield from a watershed can be transported by a stream during a small number of rainfall events that occur in a relatively short period of time within a year. However, a comparison of

the results indicates that the SWAT model might overestimate the sediment load for some high-flow events (Figure 5.2) because the SWAT model allows all the soil eroded by runoff to reach the river directly, without considering sediment deposition remaining on surface catchment areas. The results also indicate that the SWAT model underestimated the sediment load of some peak events (Figure 5.2). This might be because the sediment routing algorithm used in SWAT is very simplified. The topographic factor (LS_{USLE}) automatically estimated from the DEM in the SWAT model was found to contain errors (Kim et al., 2009; Babel et al., 2011), it partially explains the model inaccuracies for sediment yield estimation. With better accuracy and resolution of DEM and more reliable methods for derivation of the topographical variables related to LS_{USLE} , such as slope length and steepness, it would be possible to enhance precision of the model (Van Remortel et al. 2001). The land cover and management factor ($USLE_C$) was classified and assigned corresponding values in the SWAT model (Table 3). This method, however, results in $USLE_C$ factor that is homogeneous for each HRU which might cover relatively large areas and do not adequately reflect spatial variations in vegetation density within cover classes or over large geographic areas (Wang et al. 2002; Yang et al., 2014). Determining $USLE_C$ factor value as a function of fractional bare soil and vegetation cover could be implemented in SWAT model to improve sediment predictions (Benkobi et al., 2009; Yang et al., 2014). The MUSLE method improves upon the USLE and RUSLE methods by explicitly considering runoff (Kinnell, 2005). However, Qiu et al. (2012) pointed out that the SWAT model sediment predication error most likely resulted from the limitations of the existing SCS-CN method and MUSLE method. The studied watershed had intense rainfall and heavy storm events with high potential to erode surface soil, but SCS-CN and MUSLE do not account for detailed characteristics of rainfall as reported by Phomcha et al. (2011). Modification of SWAT components may be needed to take rainfall intensity and its duration into account to enhance the model performance on peak flow and sediment load simulation during the heavy rainfall season.



A

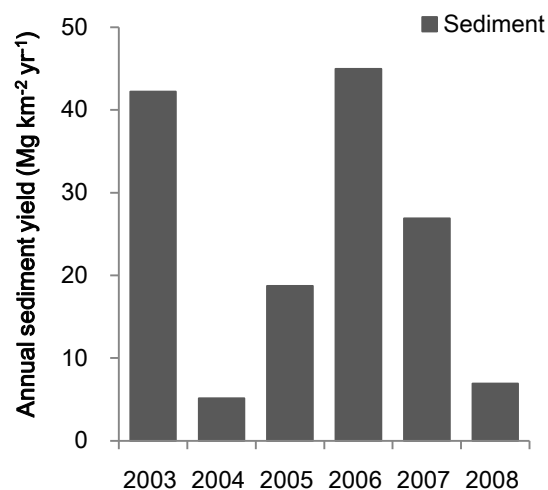


B

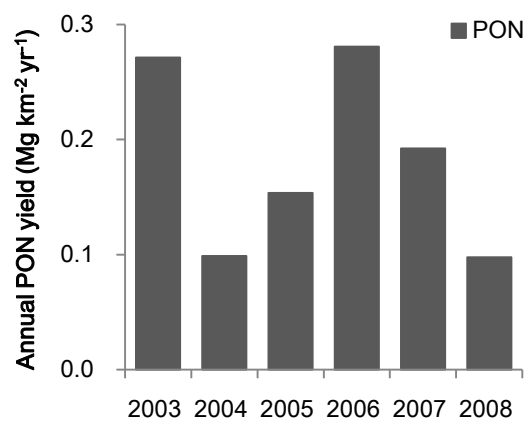
Figure 5.2 Comparison between observed daily sediment yield and simulated values obtained by the SWAT model during (A) calibration and (B) validation periods

Table 5.4 Performance of the SWAT model for estimating daily sediment and particulate organic nitrogen (PON) and phosphorous (POP) yields

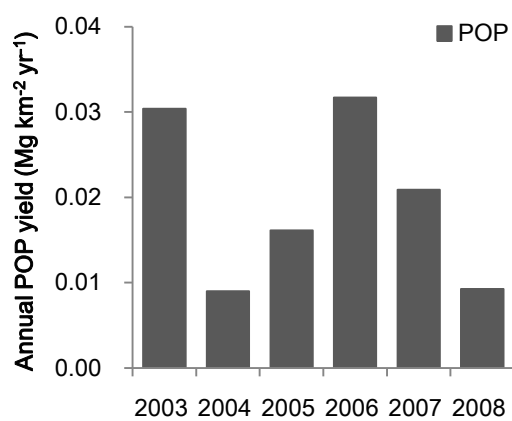
Model performance	Sediment		PON	POP
	2003	2004	2003-2004	2003-2004
R^2	0.62	0.64	0.65	0.70
E_{NS}	0.48	0.61	0.65	0.70
Re	10 %	14 %	1 %	7%



A



B

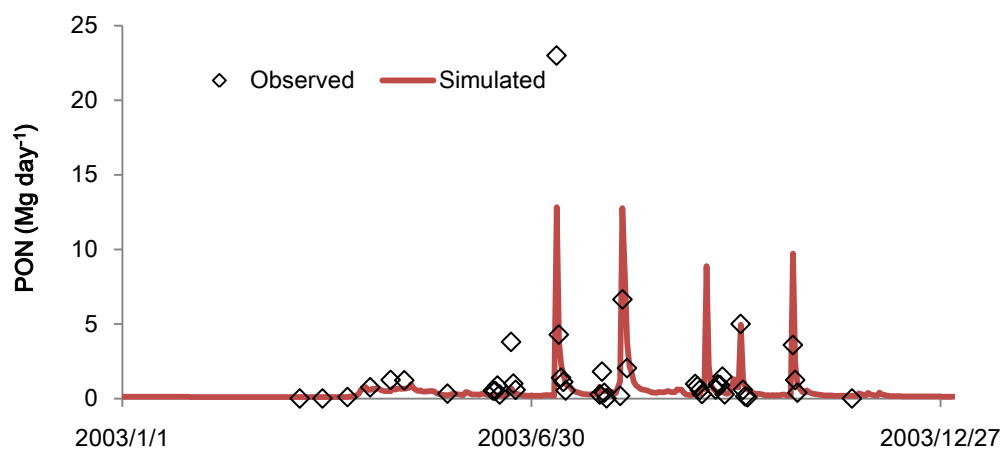


C

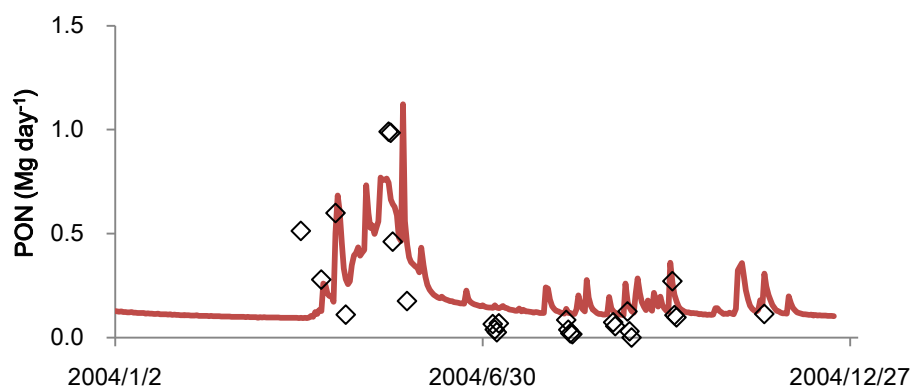
Figure 5.3 Simulated (A) annual sediment, (B) particulate organic nitrogen (PON) and (C) particulate organic phosphorous (POP) yields (2003-2008)

5.3.3 Estimation of PON and POP yields with SWAT

Statistically significant relationships were found between PON and POP concentration and sediment concentration (Figure 5.1). Strong linear relationships between sediment and particulate nutrients have also been found by other researchers (Kronvang et al., 1997; Oeurng et al., 2011). Based on these relationships, temporal variation in PON and POP yields could be computed from simulated daily sediment yield obtained from the SWAT model (Figure 5.2). The daily PON and POP yields showed a strong variability due to the variability in sediment yield within the catchment. Figures 5.4 and 5.5 show that simulated daily PON and POP yields are comparable with the observed results during 2003-2004, which yielded R^2 of 0.65, E_{NS} of 0.65 and Re of 1 % for PON yield, and R^2 of 0.70, E_{NS} of 0.70 and Re of 7 % for POP yield (Table 5.4). Simulated annual PON and POP yields from 2003-2008 showed that the annual PON yield ranged from 0.1 to 0.3 $Mg\ km^{-2}\ yr^{-1}$ (annual mean of 0.18 $Mg\ km^{-2}\ yr^{-1}$), and annual POP yield ranged from 0.01 to 0.03 $Mg\ km^{-2}\ yr^{-1}$ (annual mean of 0.02 $Mg\ km^{-2}\ yr^{-1}$) at the main outlet of the SRW. Appropriate strategies should be advised to protect critical areas with high soil erosion that also are the critical source area for PON and POP exports.

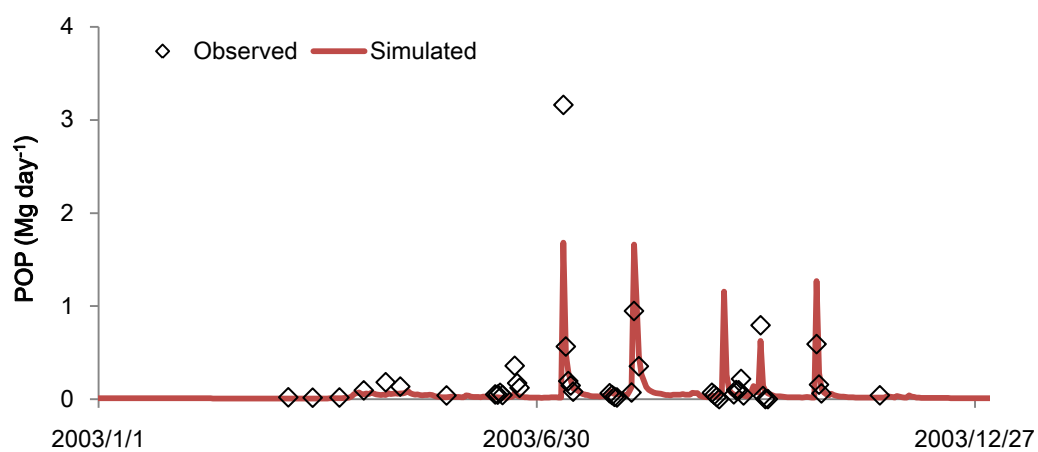


A

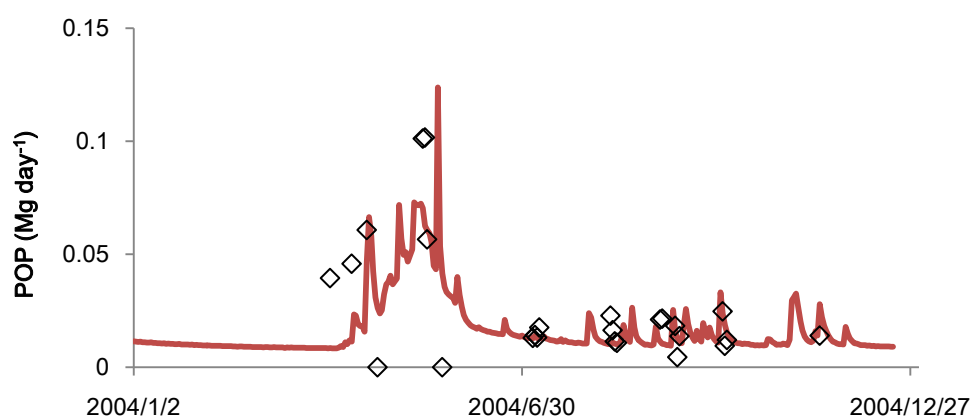


B

Figure 5.4 Comparison between simulated and observed daily particulate organic nitrogen (PON) in (A) 2003 and (B) 2004



A



B

Figure 5.5 Comparison between simulated and observed daily particulate organic phosphorous (POP) yields in (A) 2003 (B) and 2004

5.3.4 Identification of critical source areas of land surface erosion

During the studied years (2003 to 2008), SWAT model simulation results showed that sediment delivery ratio in each channel was close to one, it indicates that in-stream erosion/sedimentation might be of little importance in SRW. The average annual sediment contribution from the individual subbasin was investigated to determine its relative source contribution with the SWAT model. With the current fitted parameters (Tables 5.1 and 5.2), results showed that subbasins 3, 4, 5, 6, and 10 in the forest area had the highest sediment yield (60.1 to $110.0 \text{ Mg km}^{-2} \text{ yr}^{-1}$) compared to the neighbouring subbasins (Figure 5.6). The sediment yield value was similar to the results reported by Saghafian et al. (2012). These subbasins with the highest elevation and slopes greater than 10 degrees (Figure 3.1) were identified as the most critical source areas of land surface erosion, even though they are covered by forest. Other subbasins in the forest area with steep slopes above 10 degrees were also found to have a relatively high sediment contribution (20.1 to $60.0 \text{ Mg km}^{-2} \text{ yr}^{-1}$). Under agricultural pastureland, sediment yield increased with the distance from the watershed outlet. For example the sediment contribution was higher in subbasins 46, near 66 and 67 due to a small proportion of steep slopes in this area. Topography had an influence in that subbasins further from the outlet had a relatively high elevation and featured slopes under agricultural pastureland. Soil erosion increased with increase of steepness slope and elevation, which is most likely the reason for a higher sediment yield in these subbasins (Wu and Chen, 2012). Results from this study indicated that topography (Figure 3.1) might play an important role in land surface erosion. Agricultural land use with a small proportion of steep slopes can be critical sediment source area, even though flat terrain is found in most areas. Best management practices for effective anti-erosion, such as reduced tillage, contour cropping, the establishment of buffer strips and riparian zones, and the construction of settling ponds and wetlands, could be important in preventing soil detachment and transport from cultivated fields (Boardman et al., 2009; Ekholm and Lehtoranta, 2012). Riparian forests have been reported to play a function in soil conservation by sequestering hillslope-derived sediments at the watershed scale (Jolley

et al., 2010). In the present study site of the SRW, about 7 % to 9 % of land use consists of riparian forests, which would lead to the uncertainty in model simulation of land surface erosion because the soil conservation function of riparian forests was not considered during the simulation (Jiang et al., 2011). Results from this study indicated that forest and pasture covers were not sufficient to protect slopes from soil erosion. Watershed managers should pay attention to areas with steep slopes when implementing best management practices to reduce non-point source pollution in the SRW from land surface erosion.

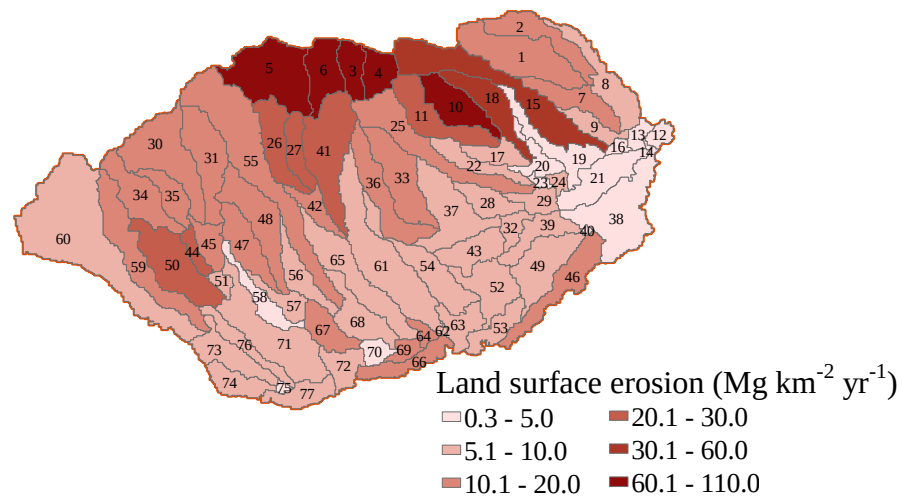


Figure 5.6 Average annual land surface erosion. Numbers show locations of the subbasins in the watershed.

5.4 Summary and conclusions

Sediment and PON and POP yields were investigated in the Shibetsu river watershed in Hokkaido (Japan), which is characterised by agricultural land use and forest, dominated by volcanic soils and recognised as the recipient of external groundwater.

The SWAT model, which includes the external groundwater contribution from HRUs to channels, was used successfully to quantify sediment and PON and POP yields at the main outlet of the SRW. Subbasins located in the upper part of the watershed were identified as critical source areas of land surface erosion. Effective anti-erosion management practices should be introduced here. The SWAT model could be used as an appropriate tool for estimating sediment and PON and POP yields and understanding soil erosion mechanisms in the SRW.

6 Modifying SWAT model to simulate the potassium export

6.1 Introduction

Potassium (K) is the seventh most abundant element in the Earth's crust. It is an essential and major nutrient for plant growth. For example, various research efforts have shown that potassium stimulates early growth, increases protein production, improves the efficiency of water use, and improves plant resistance to diseases and insects. Importance of potassium in agriculture is well recognized (Sparks and Huang, 1985).

Soil potassium is typically divided into three forms according to its availability: unavailable, slowly available (fixed) and readily available (exchangeable) (Raheb and Heidari, 2012). Unavailable form comprises a largest amount (approximately 90-98%) of total soil potassium, it exists as a component of primary minerals and becomes available upon long-term weathering. Slowly available potassium is a form potassium that has trapped between the layers or sheets of certain kinds of clay minerals. It can serve as a reservoir for readily available potassium. However, potassium release from weathering or the interlayer space of such minerals is very slow, this process is far too slow to supply the full potassium needs of plant growth (Barré et al., 2007). Readily available potassium is that which is dissolved in soil water or held on the surface of clay particles. Plant can absorb dissolved potassium readily, and as soon as the concentration of potassium in the soil solution drops, more is released into solution from the exchangeable forms. Thus, most soil tests for available potassium measure the readily available (exchangeable) forms but not the unavailable or slowly available forms. However, exchangeable potassium usually accounts for only a small fraction of the total soil potassium which is principally locked up in the structural framework of soil minerals. Cropping systems with continuously large negative balances are not sustainable in the long-term. Increasing exchangeable potassium by slurry, fertilizer or manure application can be an effective alternative in providing an available potassium pool in soils to optimize crop recovery.

Exchangeable potassium is a mobile ion in soils and consequently significant amounts can be lost with water movement. Alfaro et al. (2004) conducted a set of lysimeter based

experiments and found that although clay particles hold potassium ions in an exchangeable or available form, total potassium losses in the clay soil were more than twice those in the sand soil because of the development of preferential flow in the clay soil. Potassium can also easily leach from sandy soils than from clayey soils because sandy soils do not contain enough clay to hold the potassium (Malavolta 1985; Dassannayake, 1990). Pal et al. (1999) reported that soluble potassium is negatively related to the proportion of coarse sand and positively related to the amounts of clay and silt. Thus, greater potassium leaching losses might be expected from sandy soils than from clayey soils. Potassium leaching losses can occur by percolation after rainfall. Loss of potassium by leaching is one reason sandy often test relatively low in available potassium, especially when tested in the spring. These soils require precise annual potassium applications, since it is not possible to build up high potassium reserves. Fertilizers are the main anthropogenic source of potassium for improving potassium deficiency, however, application of fertilizer, slurry, manure or other potassium sources could increase potassium losses (Askegaard et al. 2004; Kayser et al. 2012). Nutrient runoff and leaching from agricultural fields are a major source of nutrient loss and nonpoint source pollution (US Environmental Protection Agency, 1996). Contaminated groundwater can have a negative impact when consumed, or it can contribute to surface water degradation by moving laterally into streams, lakes or bays. Direct runoff from fields also contributes to surface water degradation. Potassium leaching from soil to groundwater can affect groundwater quality and suitability for consumption, irrigation and industrial activities. When groundwater recharges into rivers, river water quality are further affected. The dissolved potassium in surface runoff and absorbed potassium on sediment during rainfall events also increased losses of exchangeable potassium from soil to rivers. Intensive crop and animal production and frequent application of slurry, manure, and fertilizer resulted in constant potassium surpluses, potassium levels might further increase with the consequence of even larger potassium losses to rivers. As a consequence, potassium concentrations might increase to above the permissible limits (12 mg L^{-1}) for drinking water recommended by WHO (1984) (Taouil et al., 2013).

Agronomic management needs balanced nutrient systems, which requires consideration of nutrients other than nitrogen. Potassium is needed in large amounts and is easily lost with hydrologic processes. In order to avoid potassium deficiency or the unnecessary waste of a limited resource in potassium balances, accurate determination of potassium losses to river systems (Askegaard et al., 2004) is necessary. However, K loss to rivers in complex watersheds has received relatively little attention in research and practice in recent years. Understanding the K losses at large watershed scale by field experiment for a long period requires tremendous investment of time or money. Modeling can be used as a critical tool to support the work and enables its users to study long-term assessment of K losses. However, assessment of K loss from large complex watersheds with modeling approach is scarce.

Watershed models have been widely used by researchers and decision makers to understand hydrologic, ecological, and biogeochemical processes as well as to examine effects of human activities and climate change/variability on water quantity and quality. Soil and Water Assessment Tool (SWAT) attempts to incorporate spatially distributed and physically distributed watershed inputs to simulate a set of comprehensive processes, such as hydrology, sediments and nutrients in large complex watersheds with varying soils, land uses and management conditions over long periods of time (Arnold et al., 1998). SWAT has been widely applied for quantify and manage riverine nitrogen, phosphorous, pesticides and heavy metals loads. However, SWAT has never been developed and assessed for potassium cycling linked to riverine processes.

In this study, SWAT model was further modified to simulate exchangeable potassium losses from land to rivers considering the agriculture management operations (harvest and applications of fertilizer and manure) at watershed scale. The continuous application of available potassium from fertilizers and manure in the agricultural land was the focus of this study. Although it was beyond the scope of this research to trace dynamics of exchangeable potassium release from soil minerals, the modified SWAT model is expected to pave a solid foundation for future research in this aspect. Given the apparent emphasis of this version of SWAT on potassium, we refer to it as SWAT-K throughout

the rest of this paper.

6.2 Methods and materials

6.2.1 Data used to evaluate SWAT-K simulation

Study site description

The study site is Shibetsu river watershed, located in eastern Hokkaido, Japan. Weather stations, main outlet location, land use and soil type map are shown in Figure 3.1. A detail description of the study site is shown in section 3.1.

Data sets

Weather data including precipitation, daily temperature, wind speed, and relative humidity were collected. Daily stream water level (H) and sediment and soluble K concentration in stream water were measured from 2003 to 2008 at the outlet of SRW. Then, daily stream discharge (Q) was calculated from H-Q equations and daily soluble K load was calculated from K concentration and stream discharge. Potassium inputs from fertilizer, application, manure application and precipitation were investigated. Potassium concentration in harvested pasture, surface soil and sediment were measured. The total K amount in the harvested was calculated based on yield and average K concentration of pasture. A detail description of the sampling and analysis methods is shown in section 3.2 and 3.3.

6.2.2 Model description

Potassium (K) cycle in soil in SWAT-K

SWAT-K address potassium input (wet deposition, fertilizer and manure application), potassium goes into the soluble and absorbed forms in the soil. Then, between plant and soil, plant uptakes potassium from soil during growing seasons or returns potassium to soil as residue after harvest or forest litter fall during the dormant period. Water-soluble potassium can be transported surface runoff, lateral flow or with percolation deep into the soil profile and enter into shallow groundwater systems. Potassium in the groundwater may loss through process of recharging streamflow. Soil particles absorbed potassium may loss from soil erosion. The plant system will take up a large amount of potassium through processes of plant harvest. A detailed description of K cycle in the

soil is shown in section 3.4.3 and Figure 3.3. After potassium entering streams, routing of sediment begins.

Potassium (K) routing in streams in SWAT-K

SWAT simulates potassium movement into the stream network via surface runoff (in solution and absorbed to sediment transported by the runoff), and into the soil profile and aquifer by percolation (in solution). Potassium in aquifer can also be transported to streams via groundwater contribution to streamflow. The potassium load in the channel is partitioned into dissolved and sediment-attached forms. The potassium in each phase is a function of its partition coefficient. While the dissolved potassium is transported with water, the potassium attached to sediment is affected by sediment transport and deposition processes. Potassium transformations in the dissolved and absorbed phases are governed by first-order decay relationships. The major in-stream processes simulated by the model are settling, resuspension, diffusion and burial. potassium in the particulate phase may be removed from the water layer by settling. Settling transfers potassium from the water layer to the sediment layer. Potassium in the sediment layer is available for resuspension. Potassium in the dissolved phase is available for diffusion. Diffusion transfers potassium between the water and sediment layers. The direction of movement is controlled by the dissolved potassium concentration, potassium will move from areas of high concentration to areas of low concentration. Potassium in the sediment layer may be lost by burial. A detailed description of K cycle in stream is shown in section 3.4.3 and Figure 3.3.

Calibration and validation of SWAT-K

All the potassium related parameters for in SWAT-K model were listed in Table 6.1. Following the streamflow and sediment calibration, the potassium related parameters were manually calibrated within their acceptable ranges to match the simulated potassium loadings with the observed daily potassium loadings. These parameters were identified as “C” and their fitted values are shown in Table 6.1. The parameter PST_KG which is the amount of potassium applied was calculated for precipitation based on potassium concentration in rainfall and total precipitation amount, for manure and

fertilizer application based on government recommendations. Some parameters were measured from the field. The parameter SOLK which is the initial potassium amount in soil was measured in 1970 and 2013. Potassium amount in soil between 1970 and 2013 was interpolated based on the two year's data. The potassium concentration of the pasture was measured in 2013, and then the parameter Fv which is the soil to plant transfer potassium factor was calculated based on the potassium concentration of plants and soils in 2013. For the parameter Fv, the measured average value from different land uses and soils were used as references in the SWAT-K model. Fitted parameters related with potassium loading simulation in the SWAT model are shown in Table 6.1.

For streamflow, sediment load and potassium load simulation, a 12-year warm-up period (1991-2002) was used to initialize the potassium partition in the soil profile (dissolved and absorbed phases) before calibration and validation of SWAT model. The long-term warm-up period was used to provide a reasonable distribution of K in soil (solid and liquid phases). After the calibration for streamflow and sediment, the model was further calibrated for one year period in 2003 and validated for one year period on 2004 for potassium load at SRW.

Table 6.1 Potassium related parameters in SWAT-K model

Parameters		Fitted method	Fitted value
K movement in soil and transport from soil to reach			
SK_{clay}	Solid-liquid distribution coefficient normalized for soil clay content (ml g^{-1})	C^*	320
WSOL	Solubility of the potassium in water (mg L^{-1})	R^*	∞^*
PERCOK	Potassium percolation coefficient	C	0.9
KENR	Potassium enrichment ratio	C	5
Variables in potassium management			
K_KG	Amount of potassium applied (kg ha^{-1})	M	90
Initial condition			
PLTK	Initial potassium amount in plant.		0
SOLK	Initial potassium amount in soil (mg kg^{-1}).	C	300
K in reach			
CHK-Kp(.swq)	Potassium partition coefficient ($\text{m}^3 \text{g}^{-1}$)	C	1E-05
CHK_STL(.swq)	Potassium settling velocity (m day^{-1})	C	0.01
CHK_RSP(.swq)	Resuspension velocity (m day^{-1})	C	1
SEDK_ACT(.swq)	Depth of the active sediment layer (m)	C	1
CHK_MIX(.swq)	Rate of diffusion or mixing velocity (m day^{-1})	C	0.1
SEDK_BRY(.swq)	Potassium burial velocity (m day^{-1})	C	0.01
Soil to plant transfer of potassium			
Fv	Soil to plant transfer factor	C	2.6
Depth distribution of potassium uptake			
β_K	Potassium uptake distribution parameter	C	5

*: M-Measure; C- Calibration from SWAT model; R-Reference;

∞ : A very large value was used for the solubility of the potassium in water.

Model performance evaluation

Model performance was analyzed by using coefficient of determination (R^2) and the Nash and Sutcliffe (1970) model efficiency coefficient (E_{NS}) and relative error (Re).

6.3 Results and Discussion

6.3.1 Field data

Potassium input

Table 6.2 shows the total K input from deposition of precipitation, fertilizer and manure application in Shibetsu river watershed. Annual K input ranged between 8.8 to 9.2 Mg K $\text{km}^{-2} \text{yr}^{-1}$ during 2003-2008. Average annual K input was about 9.0 Mg K $\text{km}^{-2} \text{yr}^{-1}$. The average annual K input was used in SWAT-K model to simulate the K export and understand the K cycling in Shibetsu river watershed.

Table 6.2 Potassium input at SRW

	Chemical fertilizer	Manure application	Deposition (Rain)	Sum of input
	Mg K $\text{km}^{-2}\text{yr}^{-1}$	Mg K $\text{km}^{-2}\text{yr}^{-1}$	Mg K $\text{km}^{-2}\text{yr}^{-1}$	Mg K $\text{km}^{-2}\text{yr}^{-1}$
2003	3.3	5.0	0.52	8.8
2004	3.3	5.1	0.38	8.8
2005	3.3	5.2	0.40	8.9
2006	3.3	5.4	0.54	9.2
2007	3.3	5.4	0.43	9.1
2008	3.3	5.4	0.31	9.0
Average	3.3	5.3	0.43	9.0

Potassium distribution coefficient in soil and stream

Soil adsorption coefficient of K (SK_{clay}) was measured for Kuroboku soil ranged from 75.5 (ml g^{-1}) to 124 (ml g^{-1}), with average value of 102.1 (ml g^{-1}). In-stream soil adsorption coefficient of K (CHK-Kp) was also measured, which is 10 ml g^{-1} (or $1\text{E-}05 \text{ m}^3 \text{ g}^{-1}$, as Table 6.1 shows). Soil adsorption coefficient of K in stream (CHK-Kp) only accounted for 10% of soil adsorption coefficient of K in land surface soil (0-5cm). Significant correlation was found between PON, POP and sediment concentrations, indicating that sediment particles bound well with organic matter. Soil organic matter is the major contributor to adsorption (Wang and Huang, 2001), other soil constituents (especially clay) can play a role in adsorption (Bangroo et al., 2012). The relative contribution is dependent on the extent of organic matter coverage on soil clay

particles. This may be a reason why sediment adsorption coefficient of K was smaller than land soil adsorption coefficient of K because K is weakly absorbed by negative charge of organic matter. The K adsorption can be stronger in high clay content low organic carbon subsurface soils than in certain high organic carbon top soils which can be washed into rivers as sediment.

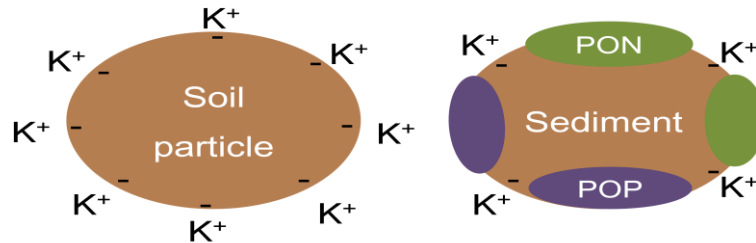


Figure 6.1 Mechanism of K adsorption coefficient in land soil and stream sediment

Soil-plant potassium transfer ratio (F_v)

Equation for potassium uptake by plant is based on soil to plant transfer factor F_v , defined as the ratio of the potassium content in the plant (mg kg^{-1} dry weight plant tissue) to that in the soil first soil layer (mg kg dry^{-1} weight soil) (IAEA, 2010). It assumes a linear relationship between the potassium concentration in plant (K_{plant}) and potassium concentration in soil (K_{soil}) (Kurashima et al., 1980). Average K concentration in pasture was about 5560 mg kg^{-1} . Soil (0-5cm) exchangeable K concentration in the seven soil types was shown in Table 6.3. Pasture K concentration under the soil types GLS, KS and PS was measured and used for calculated the soil-plant K transfer ratio (F_v) for the soil types GLS, KS and PS. Average K concentration of pasture in the soil types GLS, KS and PS was used to calculated the soil-plant K transfer ratio for other soils. The measured soil-plant K transfer ratio ranged from 6.2 to 18.6. In SWAT, the first soil layer is typically delineated with a 1cm thickness, while the average soil exchangeable K concentration was measure at depth 0-5cm, this resulted in that F_v value (2.6) in SWAT-K model (Table 6.1) is smaller than the measured value (Table 6.3) because the K concentration tends to decrease with soil depth (Figure 6.2).

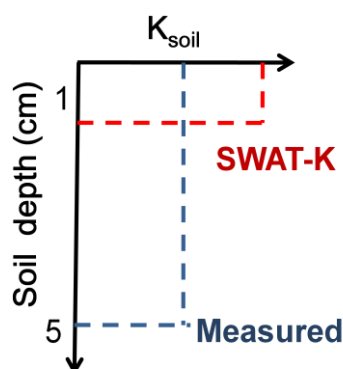


Figure 6.2 Soil exchangeable potassium concentration (K_{soil}) changes with soil depth

Table 6.3 Soil and pasture K concentration and soil-plant K transfer ratio

Soil type	Soil K concentration (mg kg^{-1})	Plant K concentration (mg kg^{-1})	Fv ($\text{mg kg}^{-1} / \text{mg kg}^{-1}$)
BFS	895	5560	6.2
BLS	794	5560	7
GLS	411	7230	17.6
KS	383	4123	10.8
PS	285	5313	18.6
RKS	379	5560	14.7
RS	403	5560	13.8
Average	507	5560	12.7

Pasture yield and potassium uptake

Pasture yield were obtained from statistics of Nakashibetsu town (Hokkaido Statistics and Information Office, Ministry of Agriculture, Forestry and Fisheries of Japan 2004, 2005). Average K content of (0.556%, 5560 mg kg^{-1}) in pasture was used to estimate the K yield in the harvested pasture (Table 6.4). The average harvested pasture yield was about $3.8 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in Nakashibetsu town (located in Shibetsu river watershed), and $4.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in Betsukai town (located near Shibetsu river watershed).

6.3.2 SWAT-K simulated potassium export to stream

K load calibration (2003) and validation (2004) were based on daily time step. Soil weathering processes was not considered because it was assumed to be very slow. The external groundwater amount of 1.38 mm day^{-1} with K concentration of 1.6 mg L^{-1} was used during the simulation. Figure 6.3 graphically illustrates comparison between observed and simulated daily soluble K export at the SRW. The simulated daily K load variation followed a similar trend to the observed values. The statistical performance of

the SWAT-K model for daily streamflow estimation was satisfactory ($R^2 = 0.82$, $E_{NS} = 0.27$ and $Re = 9\%$ for calibration period, and $R^2 = 0.80$, $E_{NS} = 0.50$, and $Re = 11\%$ for validation period). The daily soluble K export simulated by the SWAT-K model was accumulated at annual time step and yielded mean annual soluble K export of $2.1 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ for the studied period from 2003 to 2004 ($1.7 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in 2003 and $2.5 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in 2004). The simulated mean annual sediment absorbed K export was $0.0005 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ for the studied period from 2003 to 2004 ($0.0009 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in 2003 and $0.00009 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in 2004).

During the studied period from 2003 to 2008, the model simulated that about $2.2 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ entering streams through surface runoff, lateral flow, and internal and external groundwater recharge. Simulated K loss through surface runoff to stream was about $0.103 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ including dissolved form of $0.098 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ and absorbed form of $0.005 \text{ Mg K km}^{-2} \text{ yr}^{-1}$, which accounted for 5% of the total K loss to streams. K in external groundwater contribution was about $0.8 \text{ Mg K km}^{-2} \text{ yr}^{-1}$, which accounted for 36% of the total K loss to streams. K in lateral flow and groundwater was the main source of streams K loads to streams.

Table 6.4 Harvested pasture and K yields

Place	Time	Pasture yield ($\text{Mg K km}^{-2} \text{ yr}^{-1}$)	K yield ($\text{Mg K km}^{-2} \text{ yr}^{-1}$)
Nakashibetsu	2003	625	3.5
	2004	785	4.4
	2005	698	3.8
	2006	689	3.6
	2007	667	3.7
	2008	651	3.6
	Average	686	3.8
Betsukai	2003	727	4.0
	2004	865	4.8
	2005	742	4.1
	2006	759	4.2
	2007	766	4.3
	2008	725	4.0
	Average	764	4.3

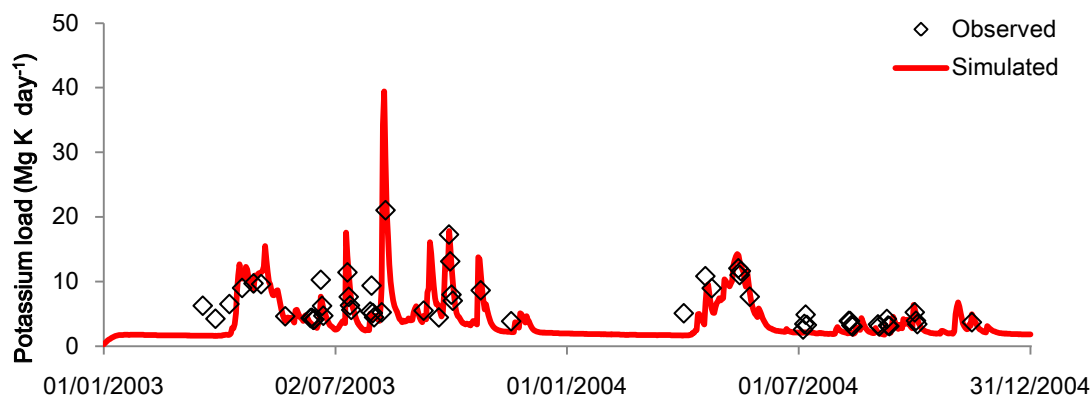


Figure 6.3 Comparison between observed and simulated daily stream dissolved potassium export at SRW

6.3.3 Potassium concentration in stream.

Figure 6.4 shows the comparison between observed and simulated stream dissolved K concentration at SRW. This figure shows that the simulated stream dissolved K concentration varied similarly with the observed data. Both the observed and simulated stream dissolved K concentration ranged between 1.5-4 mg L⁻¹. At the beginning of the snowmelt season (April) and rainfall events, both the observed and simulated results showed that dissolved K concentration tended to increase, because of the K transport with the water movement from land to stream increased significantly. The increase of K concentration happened slightly slowly than the observed data during snowmelt season of 2003, which could be explained by the poor performance of snowmelt hydrology simulation by SWAT-K model during this period.

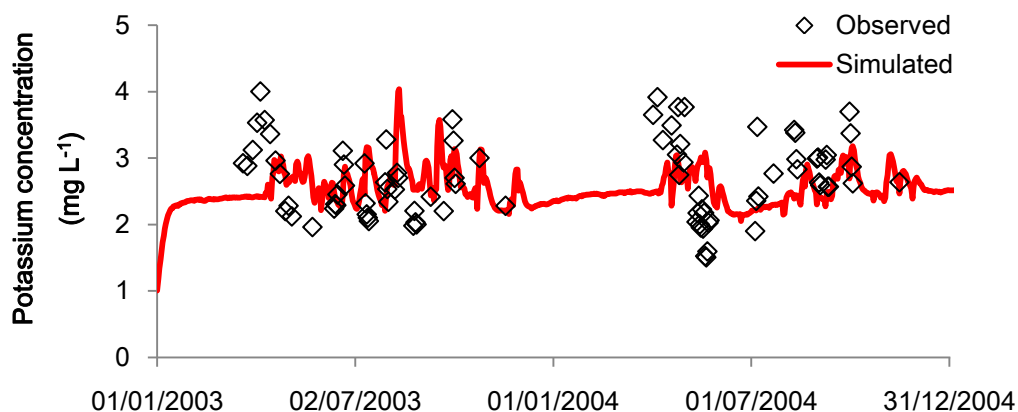


Figure 6.4 Comparison between observed and simulated stream dissolved potassium concentration at SRW

6.3.4 SWAT-K simulated potassium cycle at SRW

The K budget in Shibetsu river watershed during 2003-2008 was simulated in SWAT-K model. The simulated total K input was about $9.1 \text{ Mg K km}^{-2} \text{ yr}^{-1}$, which is close to the observed value of $9.0 \text{ Mg K km}^{-2} \text{ yr}^{-1}$. The model estimated that about $1.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ K leached out of soil profile and entered into the groundwater. Simulated amount of K loss by pasture uptake was about $2.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ at whole Shibetsu river watershed scale. Simulated K loss through surface runoff to stream was about $0.103 \text{ Mg K km}^{-2} \text{ yr}^{-1}$. Overall, the model estimated that about $5.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ was stored in the soil every year.

Under the pasture land, simulated pasture biomass was about $764 \text{ Mg km}^{-2} \text{ yr}^{-1}$ and about $686 \text{ Mg km}^{-2} \text{ yr}^{-1}$ (76% of the biomass) was harvested. The pasture yield was underestimated in the model compared with the field data of $685 \text{ Mg km}^{-2} \text{ yr}^{-1}$, however the model performance is acceptable with Re of 10%. Simulated K yield in the harvested pasture was about $4.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$, which was slightly overestimated, but close to the field data of $3.8 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in Nakashibetsu town (Re of 12%) and $4.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in Betsukai town (Re of 0.2%).

6.4 Conclusions

A simple hypothesis of 1.38 mm day^{-1} with K concentration of 1.6 mg L^{-1} was used to account the external groundwater and associated K contribution to streamflow. Also, K release from rock weathering process and forest uptake of K were assumed to be small and ignored in the SWAT-K model. The SWAT-K model was successfully used for estimating the pasture yield, pasture K uptake and K export to stream in Shibetsu river watershed, Hokkaido, Japan. The K cycle was further investigated based on the simulation results.

It should be note that the hypothetical external groundwater without temporal variation might cause uncertainties and decrease the model performance for streamflow and K export simulation when applied the SWAT-K model in Shibetsu river watershed.

7. General Discussion

7.1 Achievements of this study

Watershed scale nonpoint source pollution is the leading remaining cause of agricultural nutrients loss (for example, nitrogen, phosphorous and potassium) and water quality problems (for example, eutrophication) in freshwater and estuarine ecosystems (Liu et al., 2005; Basnyat et al., 2000; Nikolaidis et al., 1998). Understanding the watershed scale sediment and nutrients transport with water flow is important to control nonpoint source pollution and develop management practices in minimizing the agricultural nutrient loss and riverine water pollution. Watershed scale models have become critical tools in support of understanding the watershed scale water flow, sediment and nutrient transport mechanisms and supporting watershed management, such as reducing fertilizer and manure applications. Commonly used watershed scale hydrologic, sediment and nutrients transport conceptual models include the Agricultural Non-Point Source model (AGNPS) (Young et al., 1987); the Environmental Management Support System (EMSS) (Vertessey et al., 2001; Watson et al., 2001); the Hydrologic Simulation Program, Fortran (HSPF) (Johanson et al., 1980); the Integrated Water Quantity and Quality Model (IQQM) (DLWC, 1995); the Simulator for Water Resources in Rural Basins (SWRRB) (USEPA, 1994) and the Soil and Water Assessment Tool (SWAT) (Neitsch et al., 2005). Among these models, SWAT has been widely applied for quantify and manage water resources, riverine nitrogen, phosphorous, pesticides and heavy metals loads.

In this study, field study was coupled with the simulation model of SWAT to understand the water flow, soil erosion and sediment associated particulate nitrogen and phosphorous transport and losses in Shibetsu river watershed, Hokkaido, Japan. Model performance evaluation showed that SWAT model can be successfully used for quantifying the water flow, soil erosion and sediment associated particulate nitrogen and phosphorous losses, understanding the hydrology components and soil erosion mechanisms, and identifying critical land surface erosion areas in Shibetsu river watershed (Table 7.1).

SWAT has never been developed and assessed for potassium cycling linked to riverine processes. In this study, the SWAT model was modified to address the following potassium transport processes: potassium input from rainfall deposition, fertilizer and manure applications; potassium goes into the soluble and absorbed forms in the soil; Plant uptakes potassium from soil during growing seasons or returns potassium to soil as residue after harvest or forest litter fall during the dormant period; Water-soluble potassium transports with surface runoff, lateral flow or with percolation deep into the soil profile and enter into shallow groundwater systems; potassium in the surface runoff and groundwater flow to rivers. After potassium entering streams, routing of sediment begins, the following processes happen: the potassium load in the channel is partitioned into dissolved and sediment-attached forms; Settling transfers potassium from the water layer to the sediment layer; potassium in the sediment layer is available for resuspension; potassium in the dissolved phase is available for diffusion between the water and sediment layers; potassium in the sediment layer may be lost by burial. A detailed description of potassium cycle in the soil and stream are shown in section 3.4.3. SWAT-K model was applied in Shibetsu river watershed to test the model and applied the model for understanding potassium export and cycling processes based on the field data. SWAT-K well predicted the soluble K loads to stream from 2003 to 2004. The model performance evaluation showed that the modified SWAT model (SWAT-K) can be successfully used to quantify the potassium transport and potassium cycling (Table 7.1). Predicted mean annual sediment absorbed K load was only accounted for 0.02% of the total K load from 2003 to 2004. Significant correlation was found between PON, POP and sediment concentrations, indicating that sediment particles bound well with organic matter (Figure 6.1). K is weakly absorbed by negative charge of organic matter. This may be a reason why sediment adsorption coefficient of K was smaller than land soil adsorption coefficient of K and sediment absorbed K load was very low.

Table 7.1 SWAT model simulated streamflow, sediment and associated particulate organic nitrogen (PON) and phosphorous (POP), and potassium export (SWAT-K) in Shibetsu river watershed. (A: Model performance; and B: Simulation results)

Model performance		R^2	E_{NS}	Re
Streamflow	Calibration	0.60	0.40	14 %
	Validation	0.87	0.61	7 %
Sediment export	Calibration	0.62	0.48	10 %
	Validation	0.64	0.61	14 %
PON export	Validation	0.65	0.65	1 %
POP export	Validation	0.70	0.70	7 %
Potassium export	Calibration	0.82	0.27	9 %
	Validation	0.80	0.50	11 %

A Model performance

Items	Period	Simulated results
Hydrology	2003- 2008	Annual rainfall
		EXT of 480 mm
		Evapotranspiration (430 mm, 41 % of rainfall).
		Annual streamflow 1140 mm,
		Surface runoff of 30 mm (2.5 %), Lateral flow of 145 mm (13 %) Groundwater recharge of 965 mm (84.5 %).
Sediment	2003- 2008	Annual mean sediment export: 24g km ⁻²
PON	2003- 2008	Annual PON export: 0.18 Mg N km ⁻²
POP	2003- 2008	Annual POP export: 0.02 Mg P km ⁻²
Potassium	2003- 2008	Simulated total potassium input 9.1 Mg K km ⁻²
		Potassium leaching out of soil profile: 1.3 Mg K km ⁻²
		Plant uptake was about 2.3 Mg K km ⁻²
		Annual potassium export: 2.2 Mg K km ⁻²
		Surface runoff: 0.1 Mg K km ⁻² (5%) External groundwater: 0.8 Mg K km ⁻² (36%) Internal groundwater: 1.3 Mg K km ⁻² (59%)

B Simulation results

7.2 Shortages & perspective

During the application of SWAT model and the SWAT-K model for modeling water flow, soil erosion and sediment associated particulate nitrogen and phosphorous, and potassium transport and losses in Shibetsu river watershed, Hokkaido, Japan, a simple hypothesis of 1.38 mm day^{-1} calculated from annual water balance budget with potassium concentration of 1.6 mg L^{-1} was used to account the external groundwater and associated potassium contribution to streamflow. It should be note that the hypothetical external groundwater without temporal variation might cause uncertainties and decrease the model performance for streamflow and potassium export simulation. The SWAT-K model was expected to be applied in other conservative watershed around the world for understanding the potassium transfer and cycling processes.

Chemical weathering of soil minerals can contribute variable quantities of potassium depend on factors, such as soil mineralogy, moisture and temperature. However, in this study, weathering as a source of potassium input was assumed to be small and ignored in this study. The SWAT-K model was expected to be coupled with soil weathering models, such as the biogeochemical steady-state model PROFILE (Holmqvist et al., 2003) to quantify the potassium release dynamics from soil weathering.

In this study, SWAT model has been validated as an efficient tool to quantify the water flow, sediment, PON, POP export and to identify the critical source areas of the land surface erosion, and SWAT-K model has been validated with applicability for quantifying the potassium transfer, export and cycling. The management practices need to be further studied and proposed for the watershed managers to reduce the agricultural nutrients losses and improving riverine water quality.

8. Summary

Introduction

There is an increasing concern of water scarcity, nutrient loss and water quality pollution around the world due to climate change or inappropriate agriculture management (Oki and Kanae, 2006). Sediment, sediment-bound particulate nutrients and water dissolved nutrients load to rivers are responsible for declining soil fertility, decreased agricultural yields and water quality pollution. The assessment, management, and understanding of water resource availability and quality are the principal issues because of the close relationship between water resources and human life, agriculture, and the environment. Thus, it is imperative to conduct research that can contribute towards understanding the hydrology, sediment and nutrient loads.

Hydrological models can be used to support the understanding of the mechanisms of hydrology sediment and nutrient loads to rivers. Soil and water assessment tool (SWAT) is a spatially distributed and physically process based watershed simulation model that operates on a daily time step for long-term simulation (Arnold et al., 1998). Major components of SWAT include hydrology, weather, erosion, soil temperature, crop growth, nutrients, pesticides and agricultural management practices (Neitsch et al., 2005). This model has the ability to predict changes in sediment, nutrient and pesticide load to rivers with respect to the different management conditions in watershed.

The objective of this study was to investigate the hydrology, sediment, sediment-bound particulate nutrients and potassium transport processes in this watershed by using a combined field study and SWAT model simulation approach.

Material and methods

Shibetsu river watersheds (SRW, 672 km²), located in Hokkaido, Japan, is a volcanic watershed characterised by forest and agriculture. Streamflow in spring are predominantly generated from melting snow in SRW. The water quality of SRW has been impacted by sediment and nutrient loads from forest and agriculture lands.

For the whole watershed outlet of the SRW, hourly streamflow data (2003-2008) was

obtained from the Water Information System (Ministry of Land, Infrastructure and Transport, Japan). Stream water tables (2003–2008) were measured every 15 min by a water table sensor (MC-1100W; STS, Sirnach, Switzerland) installed at the river outlets. Streamflow (Q) was calculated from the stream water table (H) and calibrated H-Q equations (quadratic curve). Water samples were collected using an autosampler. This sampling method generated a high sampling frequency during storm events. Water samples were used for the analysis of suspended sediments, particulate organic nitrogen (PON) and particulate organic phosphorous (POP), and potassium concentrations.

SWAT model was further modified (SWAT-K) to simulate exchangeable potassium losses from land to rivers considering the agriculture management operations (harvest and applications of fertilizer and manure) at watershed scale. Simulation was performed by SWAT model using 90 m DEM, 1:50000 soil map, 1:25000 land use map, weather data (from 5 weather stations), and management data (fertilization and harvest). Key hydrology and sediment and potassium related parameters were selected. The SWAT model was first calibrated using SWAT CUP with the SUFI-2 calibration and uncertainty analysis routine (Abbaspour, 2007). Then the calibration of flow and sediment was performed manually to obtain a good match between the observed and simulated values. The accuracy of SWAT simulation results was determined by examining the coefficient of determination (R^2), the Nash and Sutcliffe (1970) efficiency (E_{NS}) and relative error (Re).

1. SWAT modeling of water flow

In SRW, streamflow in spring are mainly generated from melting snow, and streamflow in summer and autumn are generated from rainfall. Snowmelt and rain fall seasons hydrology was evaluated using SWAT model. In the SRW catchment, the drainage area does not correspond to the hydrological watershed and it is recognized that it receives external groundwater (EXT) from other watersheds. The EXT contribution was simplified as 1.38mm day^{-1} without temporal and spatial variation and added to the total water entering stream of the SWAT model. Along with the five snowmelt-related parameters determined to be sensitive for the simulation (snowfall temperature, snowmelt

temperature, maximum snowmelt rate, minimum snowmelt rate, and snowpack temperature lag factor), nine additional parameters were adjusted. This evaluation indicated that for the study watershed, the SWAT model had a good performance on simulating the daily, monthly, and annual mean streamflows and a satisfactory performance on predicting the daily streamflows. When analyzed alone, the daily streamflow in spring, which were predominantly generated from melting snow, could be predicted with an acceptable accuracy, and the corresponding daily streamflows could be simulated very well.

2. SWAT model simulation of sediment, particulate organic nitrogen (PON) and particulate organic phosphorous (POP) export

Sediment and sediment-bound pollutants can decline soil fertility, decrease agricultural yields and pollute natural waters. To provide an understanding of sediment and particulate organic nitrogen (PON) and particulate organic phosphorous (POP) yields at SRW. SWAT model was used to simulate daily sediment and PON and POP yields. Results showed a strong agreement between the simulated data and the observed values. The simulated annual sediment yield ranged from 20 to 230 (annual mean of 120) Mg km⁻² yr⁻¹. Annual PON yield ranged from 0.2 to 1.0 (annual mean of 0.6) Mg km⁻² yr⁻¹. Annual POP yield ranged from 0.02 to 0.13 (annual mean of 0.07) Mg km⁻² yr⁻¹. Snowfall, snowmelt and rainfall seasons contributed about 10 %, 20 %, and 70 % respectively to total sediment and associated PON and POP yields. The SWAT model identified that subbasins located in the upper part of the watershed were critical source area of land surface erosion. This research demonstrates the ability of the SWAT model to estimate sediment and associated PON and POP yields, and to improve the understanding of soil erosion mechanisms at catchment scale receiving external water.

3. SWAT model simulation of Potassium export

Potassium (K) is an essential and major nutrient for plant growth. Potassium is needed in large amounts in agriculture, however, and it is highly soluble and easily lost with hydrologic processes. In order to avoid deficiency or the unnecessary waste of a limited

resource in K, accurate determination of potassium losses to river systems is necessary. SWAT has never been developed and assessed for potassium cycling linked to riverine processes. SWAT model was further modified (SWAT-K) to simulate exchangeable potassium losses from land to rivers considering the agriculture management operations (harvest and applications of fertilizer and manure) at Shibetsu river watershed. A simple hypothesis of 1.38 mm day^{-1} with potassium concentration of 1.6 mg L^{-1} was used to account the external groundwater and associated potassium contribution to streamflow. The continuous exchangeable potassium release from unavailable and slowly available potassium from soil minerals was assumed to be small and not considered in the SWAT-K. The simulated annual soluble potassium export was $2.1 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ for the studied period from 2003 to 2004 ($2.5 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in 2003 and $1.7 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ in 2004). Results showed a strong agreement between the simulated and the observed soluble potassium export from SRW. The simulated potassium budget during 2003-2008, showed that $1.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ potassium leached out of soil profile and entered into the groundwater, $0.1 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ potassium entered into streams through surface runoff. Simulated amount of potassium loss by pasture uptake was $2.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ at whole Shibetsu river watershed scale. The model estimated that $5.3 \text{ Mg K km}^{-2} \text{ yr}^{-1}$ potassium was stored in the soil every year.

Conclusion

In conclusion, SWAT model was well applied to understand the hydrology, sediment, particulate organic nitrogen, particulate organic phosphorous, and potassium yields at SRW. The best management practices should be taken to effectively control soil erosion in the upper part of SRW and the luxury local agricultural potassium application, based on the understanding of the sediment and potassium export mechanisms.

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