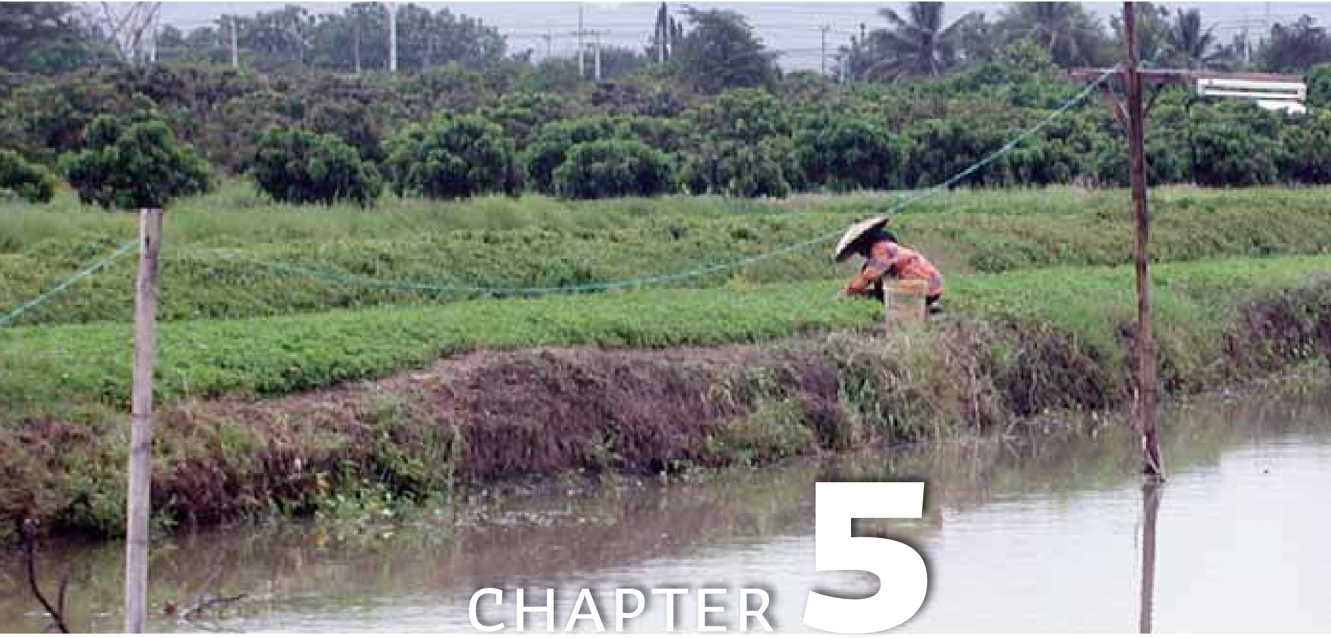

CHAPTER 5

Modelling socioeconomic impacts and decision processes

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Summary

Social and economic sciences play an important role in any integrated assessment of natural resource management issues. Land and water use and management decisions made by farmers, industry and community groups impact on the environment and on individuals in, and sometimes outside, the catchment. These groups are also affected by the decisions of others and are constrained by characteristics of the landscape in which they live. Economics and the other social sciences offer tools to assist in understanding these drivers and impacts. They can assist in developing models of decision-making, to help simulate changes in land and water use and management in response to changes in climate, policy, prices or other influences. They can provide models for simulating the impact of changes on economic prosperity and subsistence production and they can be used to design better participatory processes used to

develop integrated models. This chapter describes the role of social science and economics in the modelling and analysis aspects of the Integrated Water Resources Assessment and Management (IWRAM) project. Participation in the IWRAM project was described separately in earlier chapters.

Introduction

At the most basic level, the role of economics and the social sciences in the development of an integrated water resource assessment model can be considered to be comprised of three main tasks: representing and understanding decision-making processes and their impact on the catchment system; understanding and evaluating the impacts of changes in management and catchment conditions on the community and the values they place on different outcomes; and designing and implementing participatory approaches to ensure greater stakeholder involvement in assessment and management. This chapter considers the representation of decision-making and understanding of socioeconomic impacts. Participation in the Integrated Water Resources Assessment and Management (IWRAM) project was dealt with in Chapters 3 and 4.

Decision-making

In order to understand the impact of changes in policy or the management of a catchment, the way in which decisions affecting resource use and management are made and respond to changes in factors such as climate, policy, prices, taxes and subsidies must be understood. Decisions that need to be understood and represented may include agricultural production decisions, industrial and urban water-use decisions as well as decisions to plant areas of the catchment to forestry, to clear areas for agricultural use, or to capture run-off for production purposes before it reaches the stream. For any given type of decision, many different approaches may be taken to modelling this decision.

Considerations in developing decision models

The key issues to consider when developing decision models are as follows:

1. The spatial scales at which decisions are to be modelled and how decisions should be spatially disaggregated. Many decision models simulate decisions for representative households, farms or firms. These decisions do not usually correspond to specific areas in the landscape. In the IWRAM project, two separate approaches to simulating decisions were developed. The first was a lumped approach, which aggregated household decisions by representative households, then disaggregated these decisions by land unit. The second was a grid-based approach where decisions were simulated by grid cell.
2. The temporal scales at which decisions are to be considered. These time scales may include the representation of tactical or strategic decisions and the time step over which decisions are updated, the nature of cropping decisions being made (e.g. perennial crops versus annual or seasonal

cropping decisions) and the choice between simulating short-run decisions, where farmers are constrained by their available capital and infrastructure, or long-run decisions where decision-makers are able to adjust the amount of capital available to them.

3. The appropriate level of complexity in the model representation. Optimisation-based decision models are often considered by stakeholder groups to be overly complex or difficult to understand. A simpler approach may be as accurate (or more accurate) and may be more intuitive. This can be a distinct advantage where the model needs to be transparent to a range of stakeholders.

4. The types of trade-offs that need to be considered will inform most of the conceptualisation of the model. The decision to aggregate the decisions of specific groups of decision-makers into a single decision model or to treat them separately will depend on the type of impacts and distributional effects being considered by the model, as well as the trade-offs that need to be investigated. For example, if the decisions and impacts of changes in resource availability are expected to differ between households based on their level of resource availability, and this is of concern to the assessment, then these different groups will need to be treated separately. Otherwise it may be possible to aggregate and treat them with a single decision model.

5. The level and source of uncertainty with which decision-makers are faced may be of concern in an assessment. Decisions can be modelled using various assumptions relating to both the types of uncertainty facing decision-makers and their attitudes to risk (e.g. risk neutral, risk loving, risk averse). The model representation and structure will depend on both the sources of uncertainty considered and the attitudes of decision-makers towards risk assumed.

6. Key decision groups affecting resource use or being affected by resource availability will affect the types of decisions to be modelled. Common decision-making groups represented are farmers or other agriculturalists, hydropower personnel, and industrial or urban water users.

Some common approaches to modelling decision-making

Frequently used methods for simulating decisions include optimisation-based approaches, based on the assumption that individuals and firms act to maximise profits or utility, and decision-tree approaches, where decisions are simulated using empirically derived 'rules of thumb'. This section reviews some common approaches to modelling decision-making. Later sections provide examples of the use of decision-tree and optimisation-based approaches to simulating decision-making in the IWRAM framework.

Regional-scale production models

Regional-scale production models are generally used to consider the regional-scale, spatial distribution of impacts and trade-offs resulting from changes in policy or other factors. These types of models normally divide an area, such as a catchment or basin, into a number of regions (e.g. sub-catchments) on the basis of 'relatively homogeneous' production systems and policy scales. Each of these regions is then treated as though it is managed by an individual farmer. This allows 'averaging' or aggregation of decision-making to a scale appropriate for the types of impacts being considered. This assumption basically means that resources such as land and water are assumed to be transferable between farmers within a region. These models place emphasis on the differences between farmers from different regions rather than on differences within regions. This enables large-scale water trading and reform issues to be considered. In particular, conflicts between upstream and downstream use can be identified.

Regional-scale production models are commonly used for integrative studies at a catchment scale. This is because the scales within these models are commensurate with the required catchment scale and because the types of questions that they are designed to answer are those most frequently asked of integrative studies at this scale. In particular, questions on the spatial distribution of socioeconomic and environmental impacts resulting from changes in water policy or water trading are frequently considered using this type of model. Letcher et al. (2004) developed an integrated model for considering a variety of water-allocation policy options in the Namoi River catchment. This model used a regional-scale production model underlain by a hydrologic network to assess spatial and temporal trade-offs associated with a number of water-allocation policy changes. Trade-offs considered were both economic (between regions) and environmental, with impacts of extraction on streamflow considered.

Regional-scale production models can also indicate where water is likely to be bought into or sold out of a region given alternative production options. They can be applied to consider 'optimal' allocation of water within a basin given an objective. Impacts are generally limited to first-order impacts (i.e. impacts on agricultural production in the region). This means that secondary impacts on towns and agriculture-dependent industries are not considered. These models may be used to identify whether or not first-order impacts are large enough to warrant further investigation of these types of second-order impacts.

Hall et al. (1994) used a spatial equilibrium model, a variant of a regional-scale production model, to consider water markets in the Murrumbidgee River catchment in southeastern Australia. They used 18 regional-scale linear programming submodels, linked with a model of the river system and a model of product supply and demand, to analyse the impacts on irrigated agriculture of changing water prices and trade between regions in the southern Murray–Darling Basin. This model was later updated, with the regional structure being altered for changes in water management. This type of spatial equilibrium model was also applied by Branson et al. (1998) in the southern Murray–Darling Basin to investigate the structural adjustment implications of water reform.

Jayasuriya and Crean (2000) and Jayasuriya et al. (2001) developed a regional-scale production model to consider the trade-offs between ecological benefits and reduced irrigation production associated with environmental flow rules in the Murrumbidgee valley. The model divided the Murrumbidgee catchment into eight separate production zones and then maximised gross margins for each of these, given resource constraints in the zone. This model was linked to hydrological data from a hydrological model.

Eigenraam (1999) and Branson et al. (1999) developed a spatial equilibrium model based on regional-scale production models for irrigation areas in New South Wales and Victoria, Australia. This model was used to consider the impacts of trade, environmental-flow rules and changes in water pricing on these irrigation districts. This model was able to show the pressures for water trading in these areas and to provide information on their likely extent and direction.

The strength of the approach of regional-scale production models is their ability to consider spatial trade-offs, both socioeconomic and environmental, at reasonably large scales. They do not, however, allow the user to consider impacts on individual farmers who are constrained by their resource availability within these regions. Nor do they consider the second-order impacts on towns, agriculture-dependent industries and employment. Limited information about first-order impacts on employment may be obtained, so long as regional labour-supply constraints are included in the model formulation.

Representative farm (household) models

Representative farm models are very commonly used to consider the impact of water reforms and other policy changes on individual farmers or households. This type of model relies on identification of a 'typical' or 'representative' farm (or household) in a given area. Production decisions made by this farm subject to various resource constraints are generally considered by the model. This model may take the form of a simple farm budget, or may be a complex simulation or optimisation-based procedure.

Jayasuriya and Crean (2001) used a representative farm modelling approach, and whole-farm budgeting, to evaluate the on-farm impacts of environmental flows in the Murrumbidgee catchment. Three representative farm types were used: one typical, rice-based farm in the Murrumbidgee Irrigation Area and two rice-based farms of differing sizes in the Coleambally Irrigation Area. Impacts on farm profitability were assessed in terms of whole-farm gross margin, net farm income, and business return.

Jayasuriya (2000) developed two representative farm-scale models (one rice-based, the other a non-rice-based farm undertaking maize, soybean, canola and wheat cropping) for considering the impacts of reduced groundwater availability in the lower Murrumbidgee catchment. These models focused on short-term responses to changes in groundwater availability and so did not include consideration of potential investments in irrigation infrastructure or water-saving technologies in the model formulation.

One common issue with developing representative farm models is deriving 'typical' or representative farms for an area. In some cases, clustering and analysis of statistical data, such as farm survey data, are used.

Jayasuriya and Crean (2001) used a local consensus data approach, relying on feedback from meetings with focus groups of farmers in the area to indicate the 'typical' or representative farmers that should be modelled. The information obtained using this method was then cross-checked against other sources. Jayasuriya and Crean (2001) say that this technique is able to produce typical figures for a target group that are more representative than simple averages of statistical data, given the distortions often present due to sampling errors arising from variability in the survey population. However, they also point out that figures derived in this way cannot be easily aggregated for a regional analysis.

The main strength of this approach is its ability to consider the way in which resource constraints at the farm level constrain decision-making and influence the impact of policy changes on farms.

Urban water demand models

These models are generally based on the estimation of demand curves for urban water, assuming a given functional form and using observations of water demand. Empirical relationships between household water demand and price are generally calculated. Factors such as rainfall or evaporation may also be used to explain seasonal fluctuations in demand. These models are generally constructed by water-supply authorities for demand forecasting and pricing purposes. They assume that all households in a city or some subgroup can be represented using a single demand function, which is generally of a specific form.

The most common, and simplest, functional form for estimating urban water demand is:

$$Q = aP^b$$

where Q is household water demand, P is price and a and b are parameters derived from analysing observed demand and price data. This form of the demand curve is often used, as it readily allows a constant price elasticity of demand to be estimated (b). In order to improve the fit of the model to observations, it is often assumed that this function holds only for excess water use, above some minimum necessary threshold (sometimes considered to be equal to indoor water use).

Many other forms of demand model have been used in the past, including more-complicated econometric models of demand as a function of both price and climate—see, for example, Renwick et al. (1998). Other functional forms of the demand curve have also been assumed.

Urban demand models allow predictions of demand to be made given changes in price (demand management). Where the model is being used to simulate future water demand, a model of population growth is also required to obtain total demand.

Ringler (2001) used a net-benefit function for municipal and industrial water use, derived from an inverse demand function for water, as part of an integrated study into the optimal allocation of water in the Mekong Basin. This model and other agronomic production models were integrated with a hydrologic model using a nodal network approach.

Agent-based models

An agent-based model considers a system to be made up of a number of individual 'agents' who interact with each other—see, for example, Hood (1999). These models are based on the theory that detailed knowledge and information are available only on the properties of individuals and that system properties are a potentially non-linear consequence of agent properties (Hood 1999). Agent-based models are used mainly to understand the consequences of these types of interactions between individuals for the whole system. Thus, the concept of investigating 'emergent behaviour' of the system as a result of individual interactions is considered to be a key concern of agent-based modelling.

Hood (1999) recommends that agent-based models be used to complement 'top-down' modelling approaches, where assumptions of linearity are often made, rather than as prescriptive models. One strength of agent-based models is said to be the way in which they are not constrained by the system, rather the system properties emerge from agent interactions (Hood 1999). Also, assumptions of linearity and equilibrium, common in economic models, do not need to be made.

These models rely on detailed knowledge of individual characteristics and representation of a large number of individuals. As such, data and computational limitations generally mean that only a relatively small number of individuals (e.g. hundreds) can be considered. This limits the spatial scale at which they can be used, restricting their capacity to consider catchment or basin-scale problems.

Hare et al. (2001) considered a number of agent-based modelling case studies from around the world to develop a taxonomy of agent-based models. This taxonomy was to aid modellers in choosing the agent-based technique that matched their modelling requirements. Other applications of agent-based methods for considering natural resource problems can be found in Barreteau and Bousquet (2001) and Becu et al. (2001).

One common use of agent-based models is as a negotiation support tool, to support 'bottom-up' or participatory decision-making. These models are generally used for investigation of the system rather than to estimate the impacts of policy changes on individuals or communities.

Decision-tree approaches

Decision trees generally consist of a set of 'if…then…' rules that define the way in which decisions change in response to specific triggers. These decision rules may be derived using data-mining techniques directly from data—see, for example, Whitten and Frank (1991)—or may be postulated from a mixture of theory and qualitative information derived from interviews with decision-makers.

Ashby and De Jong (1982) use information derived from interviews with farmers to derive a decision-tree model describing farmers' tillage decisions in Colombia. The model consists of a set of decision criteria dictating the form of tillage applied. This model was tested against a second set of data.

An example of the implementation of a decision-tree approach derived from data mining of survey information, used as part of the IWRAM model framework, is given later in this chapter.

Socioeconomic impacts

The other key socioeconomic consideration explicitly considered in many integrated assessment projects is the issue of socioeconomic impacts. A very simple approach to incorporating economic and social considerations, which is commonly applied in more biophysically focused projects, is to evaluate the direct costs associated with a change in land use or management. This is the most basic approach to considering socioeconomic impacts and is more often undertaken than any representation of decision-making. A more holistic approach includes evaluation of a much broader range of impacts and considers the capacity of people to adjust away from these impacts through the decisions they make. Key social and economic impacts that may be considered include impacts on household and firm incomes and financial viability, impacts on subsistence production, employment or leisure time, changes in the recreational, environmental or amenity value associated with natural or human-derived resources and impacts on the regional economy. Again, the scale and range of impacts to be considered dictates the type of modelling approach used.

There is a clear link between representing and understanding people's decisions and considering the social and economic impacts associated with any change in the catchment system. This is particularly important where people may change their decisions to adjust away from the social and economic impacts of any intervention. These adjustments may reduce the size of impacts, or may create new impacts on others in the catchment. They may also lead to second-order environmental or biophysical impacts, which will be unexpected if potential adjustments are not considered in the assessment. In some cases, the model representing decision-making may also calculate many of the socioeconomic impacts to be evaluated. Where decisions are modelled assuming perfect knowledge, then the decision model will also include assessment of the socioeconomic impacts associated with the change on the decision-maker. Where decisions are based on uncertain expectations, a separate socioeconomic impact model will be required to estimate impacts on the decision-maker. This model may be very simple, as with the examples shown at the end of this chapter.

Common impact models

The decision models described above can often be considered to be 'impact' models as well—that is, these models usually consider not only the decisions of individuals or groups in a catchment but also calculate the impact of changes in prices, climate, or policy on these groups. However, other types of models exist that provide information purely about the nature of socioeconomic impacts, not about decision-making. This section briefly outlines two types of socioeconomic impact models.

Input–output models

Input–output models are used to consider the flows of goods and services in the economy—see, for example, Black (1997). These models assume that the economy can be divided into a number of sectors. Horton (2002) states that:

> …the fundamental premise of this technique is that changes in production levels of an economy's basic industries, arising from either changes in output or changes in demand, will, through various and extensive inter-industry linkages, produce an iterative process of spending, income creation, and re-spending, thereby changing the production levels of other, directly and indirectly related industries.

Thus, when undertaking analysis of the impacts of water trading or changes in water-allocation policies, these models are often used to consider the second-order impacts on regional industries, employment and regional income. They assume fixed-input coefficients, which are generally derived from data at one point in time, as well as linearity (i.e. constant returns to scale and constant ratios of inputs to production for each sector). Multipliers are used to indicate the strength of linkages between a particular sector and the regional economy—see Morison and Zorzetto (1995). A lack of supply-side constraints is also assumed.

Woodlock (1996) used an input–output model to consider the impact of the introduction of environmental flows policies on the regional economy of the Namoi River catchment, Australia. This model did not consider the flow-on effects to the regional economy or the environmental impacts of policy implementation. It focused on the impacts on the agricultural industry of the Namoi region. The model used a linear programming formulation to optimise the present value of regional gross margins over a three-year period, subject to resource availability. The entire area sown to crops and pastures was treated as one large farm and cropping enterprise in the model.

Leistritz et al. (2002) consider the impact of a proposed emergency outlet for Devils Lake, North Dakota, USA. The regional economic impact of various management scenarios was estimated using an input–output model. The regional economic effects considered by the model include transportation, agriculture, residential relocations and outlet construction expenditures. These effects were measured in terms of gross receipts for different sectors, secondary employment and tax collection.

Fischer and Sun (2001) address the impact of future land use scenarios on China's economy using an input–output model. Impacts on the entire Chinese economy and on seven individual economic regions were produced.

DLWC (1999) warns that input–output models are primarily designed to support measurement of economic activity rather than to support the evaluation of changes in the economy itself. As such, it is suggested that these types of models are likely to overestimate the static flow-on effects on income or employment while potentially underestimating the long-term flow-on effects, because they ignore government and capital expenditure induced effects as well as demographic effects of population change. These models are useful, however, in indicating the likely magnitude of effects and points of pressure within the regional economy.

Choice models

Choice modelling, or a choice experiment, is one of a number of stated-preference techniques used to estimate the value that the community places on various environmental outcomes. This method is capable of producing estimates of the values of changes in individual attributes as well as the value of aggregate changes in environmental quality (Morrison et al. 1996). This method uses surveys to identify respondents' preferences for environmental outcomes. Respondents choose their most-preferred resource option from a number of alternatives. This allows estimation of the value of multiple resource options. Choice modelling is based on the assumption that consumers seek to maximise utility when they make choices.

Choice modelling differs from other stated-preference techniques, such as the contingent valuation method or contingent ranking, by the design of the survey used to elicit respondents' preferences and by the statistical models used to analyse the results of the survey. Morrison et al. (1996) reviewed a number of stated-preference techniques and concluded that choice modelling had considerable potential for providing useful and valid estimates of environmental values.

Whitten and Bennett (2001) used choice modelling to estimate the non-market values of wetlands in the upper southeast of South Australia and of wetlands on the Murrumbidgee floodplain in New South Wales. The values estimated in this report were used in a cost–benefit framework, including both monetary and non-monetary costs and benefits of wetland management, to advise policy makers on the aggregate benefits of pursuing alternative wetlands policies. Bennett and Morrisson (2001) used choice modelling to estimate the environmental values of a number of rivers in New South Wales, including the Murrumbidgee River catchment.

The strength of choice modelling is in its ability to consider the impacts of policy change on non-monetary values, such as recreational or environmental values.

Two examples of decision-models in the Integrated Water Resources Assessment and Management framework

Two separate approaches were used to represent decision-making in the IWRAM framework. These approaches represent very different ways of dealing with decision-making. Importantly, both can be used to link in with the same set of biophysical models, stressing the importance of the separation between the conceptual framework underlying the interactions between system components and the specific models chosen to represent individual processes. In many ways, the choice of individual models is secondary to determining the nature of interactions occurring between system components. To illustrate these considerations, this section outlines the two approaches to representing decision-making that have been implemented in the IWRAM framework.

Household decision models

The integrated modelling approach developed in the first phase of the IWRAM project considered resource-management decisions as taking place at the household scale. This scale was chosen as it was considered that the household was the main driver of agricultural production decisions in northern Thailand (Scoccimarro et al. 1999).

Decision model formulation

With decisions on land and water use being modelled in the IWRAM decision support system as taking place at the household level, decisions are made in response to expectations of the level of land, water and labour available to a household. Households are classified into a number of different types called resource management units (RMU), on the basis of biophysical, economic and sociocultural attributes. For a detailed discussion on RMUs and their application in the IWRAM project, see Scoccimarro et al. (1999). It should be noted that individual households are not modelled, but that separate household models are run for each household type, then the results aggregated by the number of households of each type. Thus, household models essentially estimate decisions on a 'per household per RMU' basis.

RMU types differ according to their access to land and water in the catchment. For example, one RMU type may contain households that own only irrigated paddy land, while households in another RMU may own some irrigated paddy and some rainfed upland fields. The types of RMU that may be seen in a catchment are summarised in Table 5.1. Classification of households into RMUs was undertaken using household survey data collected in the catchment. Only types 2, 3 and 8 are considered in this paper, as these were the only types seen in the survey data for the Mae Uam sub-catchment. Households of RMU2 have access to irrigated paddy land only. Households of RMU3 have access to rainfed upland fields only, while those of RMU8 have access to both irrigated paddy fields and rainfed upland fields.

Each household is assumed to be constrained in its activities by its access to land, water and labour. Households are modelled as aiming to generate as much household income as possible given a choice of crops, and expectations on the amount of land, water and labour that will be available to them. Social constraints, such as the desire to grow rice as a subsistence crop during the wet season, are included as constraints on household decision-making. For example, households are limited to growing mainly rice in the wet season in order to meet their subsistence needs. A level of 300 kg per year is assumed to be required per person to eliminate the subsistence deficit. Cash cropping is assumed to take place in the dry season. The model allows for different choices of fertiliser level on crops as well as for the choice of whether or not to irrigate a crop. The crops that can be chosen by each household type differ. For the Mae Uam: RMU2 can choose irrigated paddy rice in the wet and dry seasons, and irrigated sorghum in the dry season; RMU3 can choose rainfed upland rice in the wet season and rainfed sorghum in the dry season; and RMU8 can choose irrigated paddy rice and rainfed sorghum, upland rice and groundnut in the wet season and irrigated and rainfed sorghum in the dry season. These crop choices were derived from survey data.

It is possible to run the DSS over several years or for a single year. If the model is run over multiple years, then the expected volume of irrigation water available to a RMU for each successive year (used in the household decision model) is updated on the basis of events in previous years. In the first year, the expected quantity of irrigation water is that which is initially assumed by the user. In all other years, the expected value is the actual amount of irrigation water used by the household in the previous year (i.e. naive expectations are assumed). Climate data for each year also affect flows, erosion, crop yields and irrigation demands calculated by biophysical models in the DSS.

Table 5.1 Possible resource management unit (RMU) types for use in the Integrated Water Resources Assessment and Management decision support system

RMU	Description
1	rainfed paddy only
2	irrigated paddy only
3	rainfed upland only
4	irrigated upland only
5	rainfed and irrigated paddy
6	rainfed paddy and upland
7	rainfed paddy and irrigated upland
8	irrigated paddy and rainfed upland
9	irrigated paddy and upland
10	rainfed and irrigated upland
11	rainfed and irrigated paddy and rainfed upland
12	all types
13	irrigated paddy and upland and rainfed upland
14	rainfed paddy and upland and irrigated upland

Linear programming is invoked to solve the constrained optimisation, using separate components for wet- and dry-season decisions. At present, only seasonal cropping decisions can be accounted for in the model. Decisions to grow perennial produce, such as fruit trees, are not currently incorporated in the model. In most cases, Base-case values were determined from the household survey data.

Spatial disaggregation of household decisions and links to the biophysical models

The modelling uses a nodal-network where nodes represent aggregated points of extraction along the river system. Each node is associated with an area of land containing many households and land uses. This means that household extraction decisions in an area are aggregated and are modelled as occurring from a specific point along the river. Total water supply, simulated using the hydrological model (see Chapter 6), is also an output at this point or node. Households in an area are divided into a number of representative RMUs and the decisions of individual households are aggregated by summing the decisions of each RMU type present at the node.

Households of the same RMU type are modelled as having the same access to land, water and labour at a node. This means that within any one nodal area the same land use decision is assumed to be made by each of these households in a specific RMU type. Household decisions for each RMU type present at the node are aggregated across individual RMUs and then across RMU types. This aggregate land use decision is fed to the biophysical models as an aggregated land use and management decision for the node. These models consider biophysical processes on a land-unit basis. Land units correspond to unique soil types and slope classes (for further details see Chapter 3). Household decisions for each RMU type need to be disaggregated to individual land units then summed over the entire catchment in order to be passed to the biophysical models. The decision disaggregation model (DDM) uses a procedure to disaggregate crop decisions to each land unit, using the household decisions for each RMU and the number of households of each RMU type in the catchment. The DDM outputs the total area of crops in each season on each land unit as well as the total forest cover in the catchment.

Users of the model can change the total number of households of each RMU type at each node, as well as the access that each of these RMUs has to land, labour and water resources. In this way, they are able to explore changes that could occur in the catchment, for example as a result of forest clearing for agriculture, or migration into the catchment.

Decision-tree approach

In the second phase of the IWRAM project, an alternative, decision-tree approach was used to simulate production decisions in the catchment. This approach was chosen to overcome several limitations of the more-complex linear programming approach described above:

- The decision-tree approach can simulate grid-based land use decisions, a key desire of Thai management authorities (improving uptake and adoption).
- No assumptions (such as profit maximisation) are made about decision-making, instead the drivers of land use decisions are derived directly from detailed survey data collected in the catchment.

- The decision-tree approach is relatively simple to implement and can be readily understood by non-technical users. As such, it is more accessible to a broader range of stakeholders, increasing the likelihood that the decision support tool will be adopted.

These advantages have driven the choice of approach in this project. However, use of this approach relies on extensive data frequently not available in an integrated assessment. Thus, choice of the appropriate approach should be made considering the characteristics of the problem at hand, including available data, user requirements and the spatial and temporal scales required of simulations.

Data collection

Two surveys were conducted of households in the three catchment areas as part of the socioeconomic component of the second phase of the IWRAM project. In the first stage, a survey of farmers was conducted by the Land Development Department (LDD) covering 23 land units (312 households; 212 from Mae Ping Part II Watershed and 100 from Mae Kuang). This survey was conducted in 2000. In the second stage, another farmer survey was conducted (in 2001) by a team from Chiang Mai University covering 23 land units and 284 households (50 from Mae Rim Watershed, 109 from Mae Kuang and 125 from Mae Ping Part II watersheds). After major land units together with their administrative boundaries were identified, sample households were selected based on these land units. These households were chosen to supplement the survey previously done by LDD, so that land units surveyed did not overlap with those previously surveyed. Global positioning system (GPS) equipment together with detailed administrative maps were used to pinpoint the exact location, and farmers in these land units were selected for interviews. Approximately 4–8 households having the same cropping pattern were selected at each location.

Together, the two surveys covered 37 land units and 596 households. There were about eight farm households interviewed in each land unit. In addition, informal interviews and socio-logical studies were conducted to supplement understanding of farming systems in the area. Questions asked related to cropping patterns, problems of farming, use and management of irrigation systems and environmental problems.

Table 5.2 summarises the main information sought from households during the survey conducted by Chiang Mai University. The final data-set collected represents a comprehensive database of crop activities and household characteristics suitable for classifying decision-making behaviour in the study area. Data-mining techniques were then used to derive from this data-set a set of decision rules, describing wet- and dry-season cropping decisions using these household attributes.

Analysis of the survey data

In order to derive decision trees from the survey results, the crops were grouped into several categories. These were based not only on economic characteristics of the crops, but also on advice from agronomists in the project team. The labels used to identify crop categories are given in Table 5.3, with a suffix used to indicate if the crop is grown in the wet or the dry season.

The variables considered from the survey by the data-mining analysis as possible descriptors of crop choice were: the estimated profit level; cost of production; farm size per unit of household labour; total farm size; household labour units; the number of household members; whether or not the household would consider an alternative crop; farmers' willingness to participate in off-farm employment; whether the farm has livestock; the land-tenure status of the farm; the incidence of waterlogging on farm; the incidence of drought periods on farm; the availability of irrigation water; membership in a water-users association; and household capital availability. In some cases, these variables were grouped into discrete classes to aid with the analysis.

Wet- and dry-season crop choices were analysed separately using the data-mining algorithm. In both seasons, the data could be classified accurately using only four attributes: land unit, estimated cost of production, the land–labour ratio and estimated profit level. These four variables and the classes used for each in the analysis are described in Table 5.4.

Given these decision trees, each land unit can be divided into many wet- and dry-season crops depending on the farmers' profit expectations and their resources, e.g. capital (estimated cost of production, land and labour availability). The decision tree can be used predict what crops a representative farmer will grow in the study areas, given different assumptions about resource availability.

A brief summary of the decision trees is given below.

Wet- and dry-season decision trees

Separate decision trees were determined for wet- and dry-season cropping decisions. These decision-trees are shown in Figures 5.1 and 5.2.

These decision trees demonstrate that decisions on which crop to plant depend not only on the physical characteristics of the land but also on characteristics of the farmers, such as how much land they have, how much money they have, how much labour they have and how much their decision is driven by profit maximisation. The data-mining results indicate that many different crops can be grown on any land unit in each season, depending on the farmers' characteristics. This information can be used to simulate changes in farmer decision-making given changes in the distribution of many of these farm characteristics.

Table 5.2 Survey information collected by the Chiang Mai University team for use in socioeconomic modelling in phase II of the Integrated Water Resources Assessment and Management project

Information requested	
Part 1	General, household characteristics, farm and household size
Part 2	Land type, tenure and land utilisation, crop year 2001
Part 3	Production costs for annual crops and perennial crops including fertilisers, materials, machinery and labour use
Part 4	Output, product sold and income for annual or perennial crops
Part 5	Income for other sources and capital availability
Part 6	Environmental problems
Part 7	Past use of land, competition of annual crops, farmers' attitudes
Part 8	Use and management of irrigation water

Implementation of decision trees for simulating land use decisions

A decision simulation model was then constructed using these decision trees. The framework of this decision simulation model is show in Figure 5.3. This shows that GIS data are used to determining the farmers' level of investment. This then feeds into the wet-season crop decision tree, where wet-season crop choice is simulated. This choice is checked against system constraints before dry-season crop choices are simulated. Wet-season choice of perennial crops, including fruit trees and forest, is also passed as a constraint to the dry-season crop choice. The final output is a GIS-based output of wet- and dry-season crop choice.

An example of the type of output produced by this model is shown in Figures 5.4 and 5.5. These figures show different wet- and dry-season land use choice maps under two scenarios: scenario 1, in which households are without additional credit; and scenario 2, where each household has 10,000 baht in additional credit.

Table 5.3 Crop groupings used for analysis of survey data collected by Chiang Mai University during phase II of the Integrated Water Resources Assessment and Management project

Crop type	Category (Label)	Crops included in category
Rice	rice_wet, rice_dry	glutinous rice, non-glutinous rice
Non-rice field crops	maize_wet, maize_dry	maize (corn), baby corn and sweet corn
Non-rice field crops	bean_wet, bean_dry	green soybean, groundnut, sweet bean, soybean and yardlong bean
Vegetables	leafveg_wet, leafveg_dry	head lettuce, bakchoi cabbage, Chinese cabbage, spinach, kale, green cabbage, cabbage, cauliflower, michilli
Vegetables	rootveg_wet, rootveg_dry	carrot, Chinese radish, potato, gobo, garlic and shallot
	othveg_wet, othveg_dry	bitter gourd, chilli, bunching onion, tomato, sweet basil and basil
Other annual crops	flower_dry	marigold and curcuma
Other annual crops	tobacco_dry	tobacco
Tree crops	banana	banana
Tree crops	longan	longan
Tree crops	lychee	lychee
Tree crops	mango	mango
Tree crops	tea_coffee	tea, coffee
Tree crops	ornamental	ornamental trees

Table 5.4 Final decision-tree variables used to represent decision-making in phase II of the Integrated Water Resources Assessment and Management project

Variable	Description	Values used for analysis
LU	Land unit as defined by the Land Development Department (LDD)	Values as defined by LDD
Profitgrp	This is calculated from gross margin level. Profit aspiration is divided into five groups. Certainly, a farmer wants more profit rather than less, but usually more profit means more risk, skills and management. One can think of these as a variable indicating risk and skill levels. Level one of profitgrp is low risk, low return and easy skills. Level two and three being medium risk, return and medium level of skills. Level four and five being high risk, return and high skills level.	<=3000 baht: profitgrp=1 >3000 to ≤6000 baht: profitgrp=2 >9000 to ≤12000 baht: profitgrp =3 >12000 to ≤15000 baht: profitgrp 4 >15000: profitgrp=5
Costrd	This is redefined from the actual cost of production. This variable indicates the level of investment farmers want to make in a particular crop.	cost 2 ≤2000 baht: costrd=2000 >2000 to ≤4000 baht: costrd=4000 >4000 to ≤6000 baht: costrd =6000 >6000 to ≤8000 baht: costrd 8000 >8000 to ≤10000: costrd=10000 >10000 to ≤12000 baht: costrd= 12000 >12000 to ≤15000: costrd=15000 <15000: costrd=20000
Landlabor	This is farm size divided by the units of household labour. Low values indicate land scarcity in relation to labour. High values indicate relative land abundance in relation to labour.	

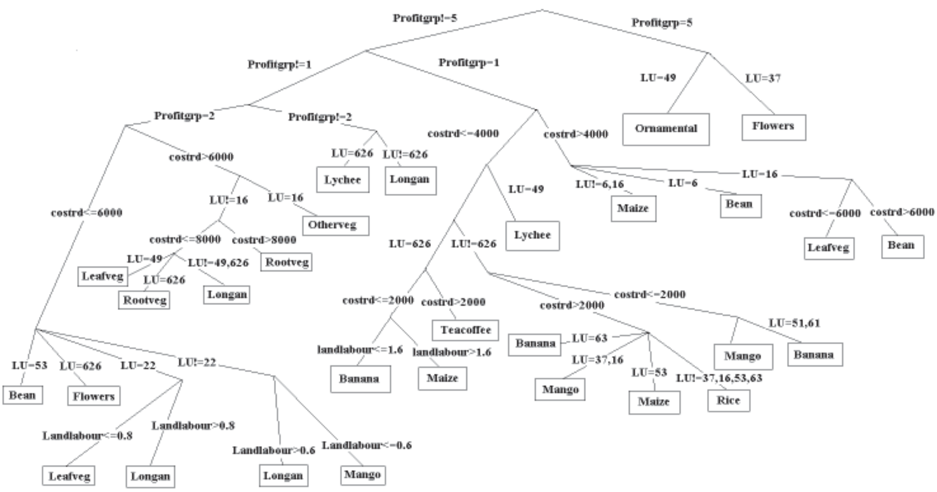

Figure 5.1 Wet-season decision tree derived from survey data collected during phase II of the Integrated Water Resources Assessment and Management project

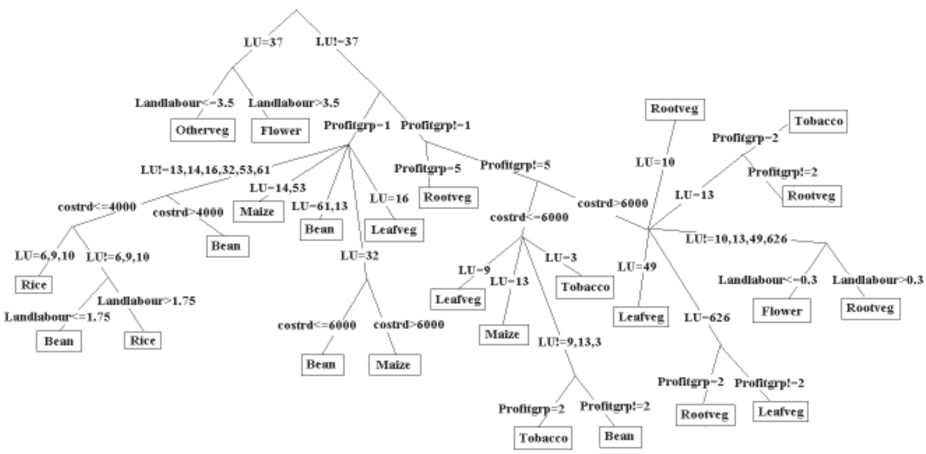

Figure 5.2 Dry-season decision tree derived from survey data collected during phase II of the Integrated Water Resources Assessment and Management project

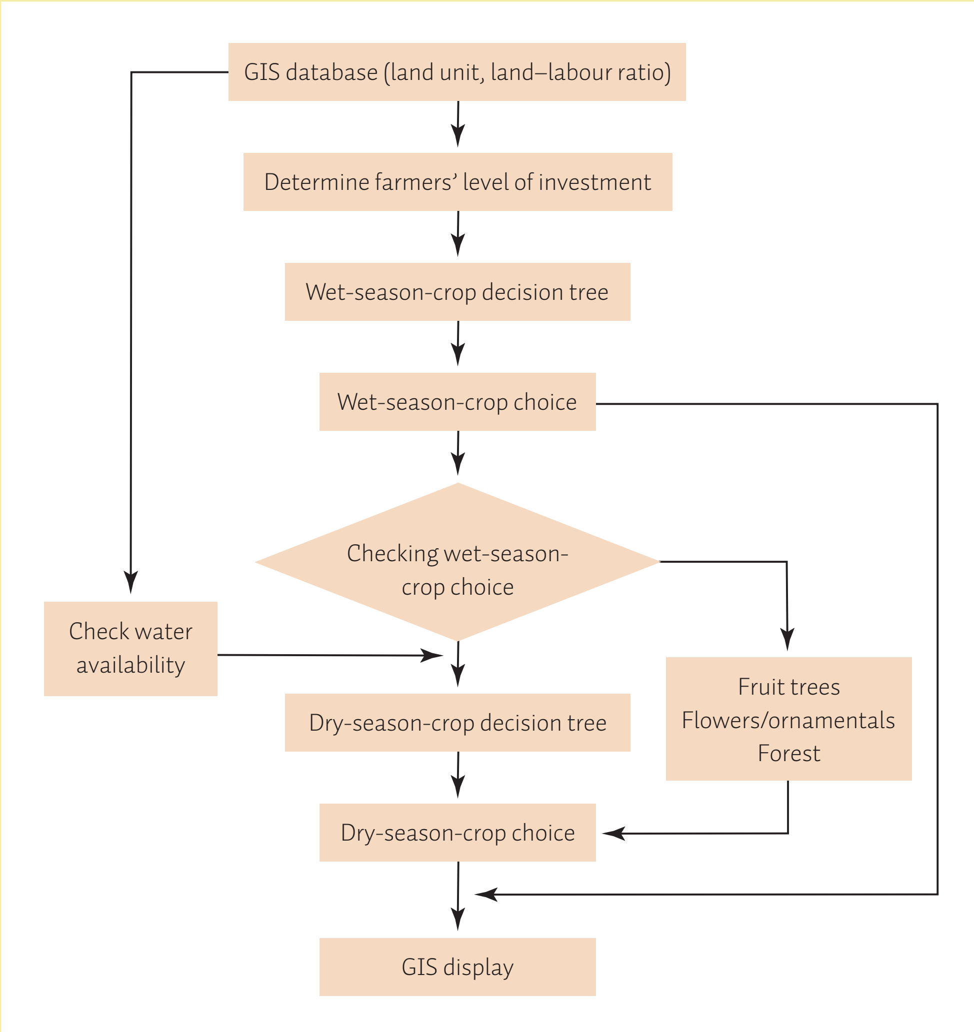

Figure 5.3 Decision model structure, incorporating wet- and dry-season decision trees, implemented in phase II of the Integrated Water Resources Assessment and Management project decision support system

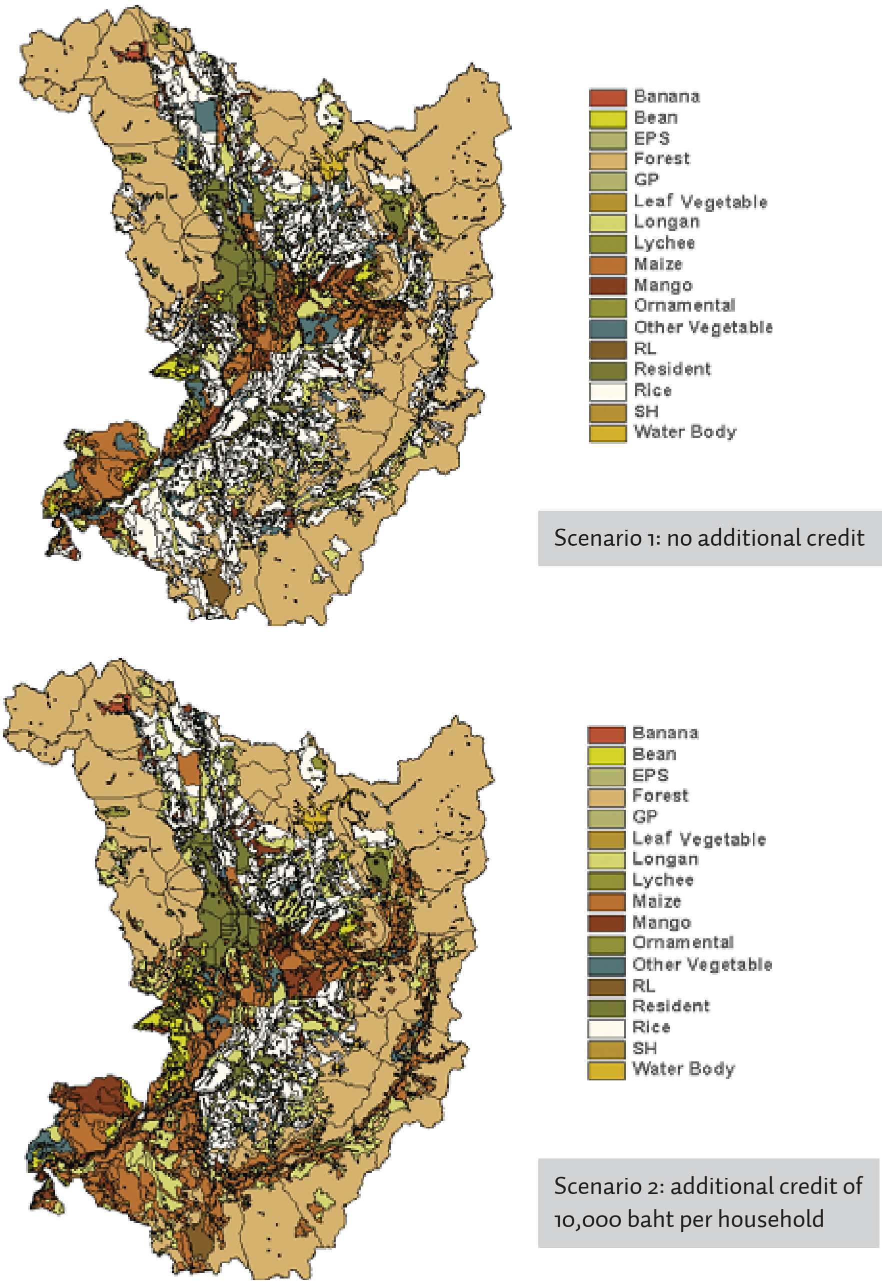

Figure 5.4 Example of decision model output for phase II of the Integrated Water Resources Assessment and Management project decision support system: wet-season crop choice

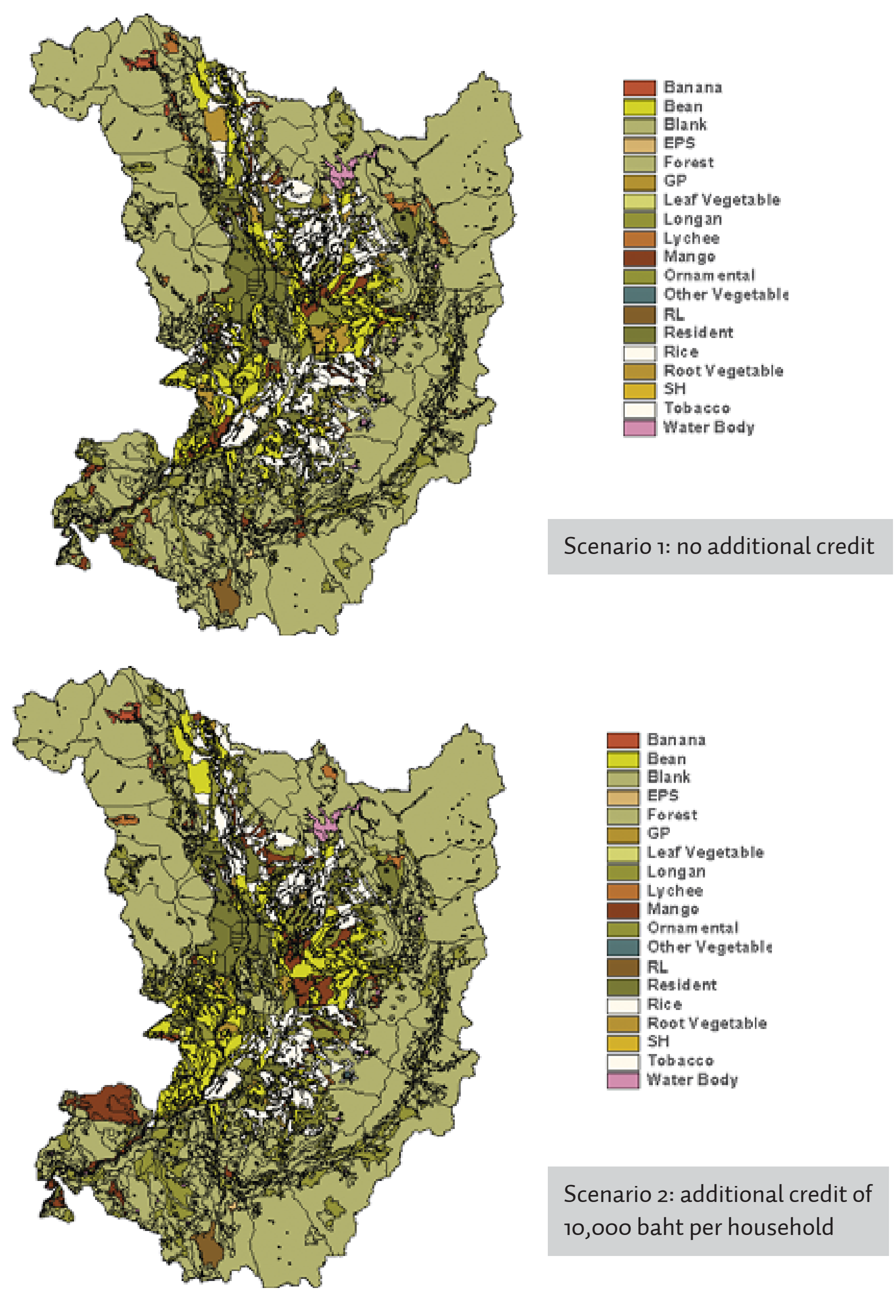

Figure 5.5 Example of decision model output for phase II of the Integrated Water Resources Assessment and Management project decision support system: dry-season crop choice

Treatment of socioeconomic impacts in the Integrated Water Resources Assessment and Management project

In the IWRAM project, assessment of socioeconomic impacts was limited to the first-order impacts of changes in land and water availability on agricultural households. The decision models used in both phases of the project assumed expectations-based decision-making, so a separate socioeconomic impact model was required to assess the final impact of decisions and resource availability on household performance. This impact was assessed in terms of farm gross margin and subsistence rice production, given crop yields simulated by the biophysical models in the framework and the areas planted to different crops.

The socioeconomic impacts relative to some base conditions, arising from a scenario where agricultural expansion leads to increases in the land available to individual households is shown in Figure 5.6. The model was run over five years using climate data from 1989 to 1993. Results from only the upstream node in the

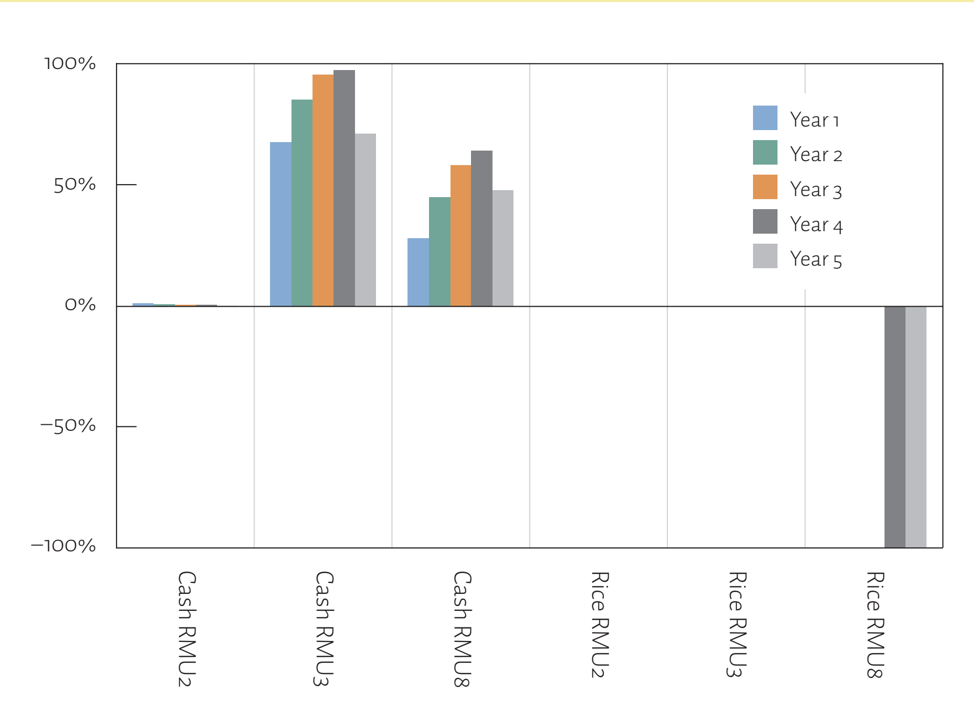

Figure 5.6 Socioeconomic impacts of agricultural expansion in the Mae Uam catchment, northern Thailand

Mae Uam catchment are shown. The socio-economic models (household decision models and socioeconomic impact models) from phase 1 of the IWRAM project were used to produce these results.

Households of RMU2 receive a very small benefit (i.e. increase in household cash) from the increase in land available. These households are constrained by their access to other resources (water and labour) more than land and so do not receive large benefits from the increase in area available. Households of both RMU3 and RMU8 have access to rainfed fields and benefit to a much greater extent than those of RMU2. In some years, household income in these RMUs more than doubles under these scenarios. Also, these households have a small rice deficit under the base-case assumptions. Increases in land lead to the removal of this rice deficit. This means that increasing the land area available to these households helped them meet their subsistence requirements and increased their cash wealth.

Conclusions

This chapter outlined the role of socioeconomic analysis and modelling in the IWRAM framework. In terms of modelling, this role can be divided into two components: modelling decision-making; and simulating socioeconomic impacts from changes in climate, policy, access or other drivers. Two approaches to modelling decision-making were applied in the IWRAM framework. The first was an optimisation-based approach to modelling decision-making on a household scale. The second used data-derived decision trees to simulate grid-based land use decisions. The choice of approach depends on many factors including the availability of survey data, the comfort levels of stakeholders and users with more complex optimisation-based approaches, the drivers of decision-making that need to be captured in the analysis and the need for spatially explicit model outputs. Regardless of the choice of model, the same framework is used to integrate decision-making with models of biophysical processes. This means that the focus in model development should be on developing the framework within which components will be integrated, and understanding the requirements that this and the problem place on each of the component models. Different models can then be used to meet these requirements.

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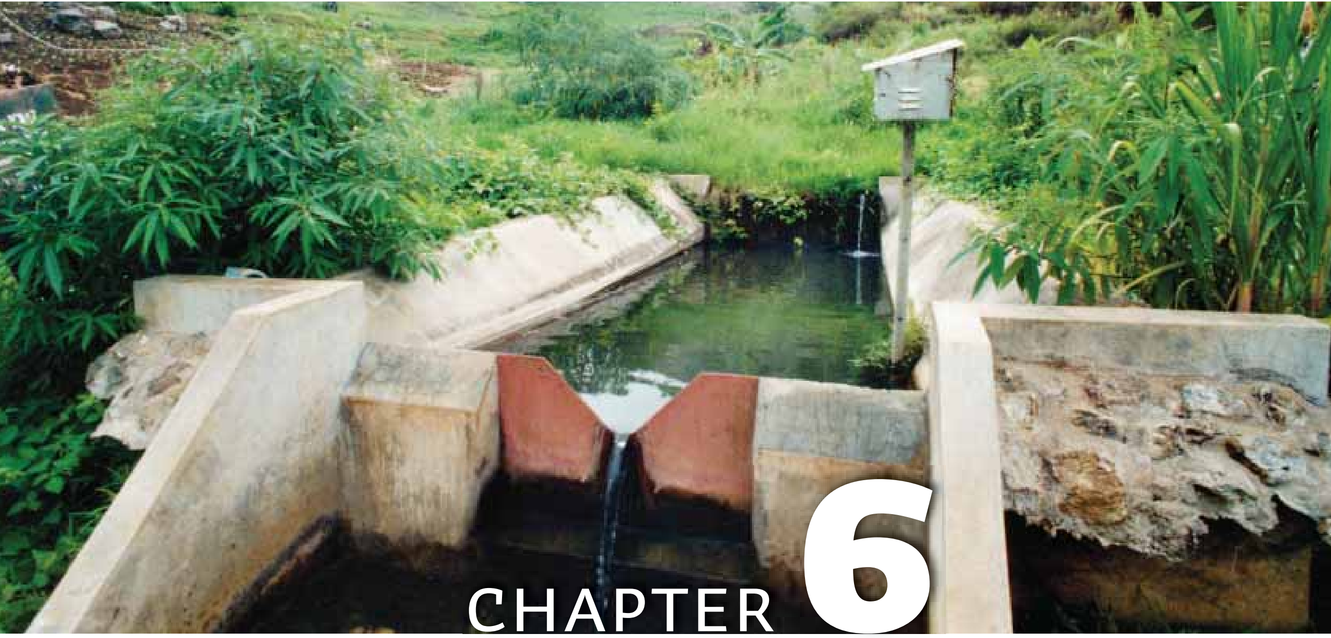

CHAPTER 6

Simulating the effects of land-cover change on streamflows

Barry Croke, Wendy Merritt, Pongsak Wittawatchutikul, Atthorn Boonsaner and Sompop Sucharit

Summary

The potential impacts of deforestation on hydrological response are of significant importance worldwide, and especially in highland regions of northern Thailand and other parts of Southeast Asia. In these regions, where climate exhibits strong seasonality, the availability of water in the dry season determines the feasibility of multiple crop rotations. This chapter presents two approaches to the prediction of hydrologic response to land use changes as well as prediction of flows in ungauged catchments. These approaches are based on the IHACRES rainfall–run-off model (applied to the Mae Chaem catchment) and the US Soil Conservation Service curve-number approach (applied to gauge P37 in the Mae Ping basin). Both of these approaches have been used within the Integrated Water Resources Assessment and Management decision support system.

The prediction of flows in ungauged catchments is a major hurdle in water resource analyses in regions like northern Thailand where there is a lack of stream gauge instrumentation, or where assessment of water availability is required at locations between gauging sites, or under conditions of changes in forest cover, as input to agricultural production models.

Figure 6.1 Small stream flowing through the Mae Chaem catchment, northern Thailand

Introduction

The type and complexity of a hydrological model used in an integrated modelling framework depends on what management decisions are to be considered, the spatial and temporal scales considered in the integrated framework, and what outputs are required by other models within the framework. For example, for a rural environment, the hydrological model may need to be sensitive to the pattern of land use, or just the relative areas of each land use. The primary role is to estimate the streamflow for a given land use pattern or management scenario. The model may also need to supply additional water-related information required by other models; for example, soil moisture variations (spatial and/or temporal) for crop modelling.

Thus, the structure of the integrated framework dictates what the inputs and outputs of the hydrological model should be. Ideally, the simplest model that fulfils these basic requirements should be used, as more-complex models will require more resources to develop, due to increased data requirements and difficulty in calibration. The two examples presented here differ mainly in the degree of spatial sensitivity included in the models and, as a result, differ slightly in their complexity, the data requirements and the difficulty in calibration.

Role of hydrological models

One of the key roles of hydrological models in an integrated modelling framework is to provide estimates of the streamflow for a particular land use/management scenario. This can be used to estimate water availability for downstream users such as irrigators, and hence determine what type of crops can be grown. Generally, the effect of a land use scenario on streamflow is limited to the effect of the vegetation cover across the catchment (divided into broad types such as evergreen forest, cropland, pasture etc.), as the effects of finer land use classifications and spatial distribution are difficult, if not impossible, to determine from gauged streamflows.

Hydrological models are also used to infer the effects of climate variability and climate change, though results become increasingly uncertain the further catchment and climatic conditions are from those used to calibrate and validate the model. For a review of the current state of knowledge on forest hydrology and related land- and water-management issues in the humid tropics see the compendium by Bonnell and Bruijnzeel (2005).

Model types

There have been many reviews of the status of catchment hydrology as a science, and our ability to make predictions (e.g. Klemeš 1986; Beven 1987; Goodrich and Woolhiser 1991; Wheater et al. 1993; Hornberger and Boyer 1995; Croke and Jakeman 2001). In this section, a brief summary of the different model types is presented.

Wheater et al. (1993) classified rainfall–run-off models into four categories: metric, conceptual, hybrid metric-conceptual and physics-based. Metric models are based primarily on observational data, and attempt to characterise system response using that data. As a result, these models do not attempt to describe the physical processes taking place. An example is the earliest unit-hydrograph methods. Conceptual model types represent the next step up in model complexity. These models attempt to represent all the important hydrological processes at the catchment scale, based on other prior knowledge. These are generally spatially lumped—e.g. MODHYDROLOG, Chiew and McMahon (1994), though distributed models also exist—e.g. LASCAM, Viney and Sivapalan (1999). While the models are based on the important processes taking place, generally the parameters cannot be measured in the field due to the lumped nature of these models (even the distributed ones). The structure of conceptual models is defined a priori, in accordance with the perception of the important processes. Hybrid models combine the metric and conceptual paradigm—e.g. IHACRES, Jakeman and Hornberger (1993)—utilising data to discriminate among many hypotheses about the appropriate model structure. All these models need to be calibrated against observed data, with limited ability to transfer parameters to other catchments.

Physics-based models use a more classical mathematical form to describe hydrological processes (such as the Richards' equation for vertical transport). Such models—e.g. TOPOG_IRM, Zhang et al. (1999) and ANSWERS, Connelly et al. (1997)—are necessarily distributed, and require that each cell be homogeneous, or at least that the heterogeneity within each cell does not significantly affect the model's accuracy, or the ability to derive the necessary parameter values from field measurements. While distributed models have the highest potential for yielding information, particularly in studies of the effect of land use change on flow volumes, they also require more extensive validation than lumped models.

Woolhiser (1996) noted that, even if the physical entities represented by the parameters vary smoothly in space and are constant in time, the parameters are actually lumped to some extent (and hence may be impossible to measure directly) due to the use of discrete time steps. To avoid this difficulty, the time step used in the model must be small enough to approximate the continuity of the system. The necessary time step depends primarily on the temporal nature of the precipitation, with storm events requiring a much finer time step. The question here is: at what temporal resolution does the discrete nature of the model affect the representation of the processes? For storm events, high spatial and temporal resolution data are needed, and so a major limiting factor for physics-based models is the availability of rainfall data.

Three of the issues related to complexity of a model are over-parameterisation, computational demands and error accumulation.

Grayson et al. (1992) discussed the merits of process-based, distributed-parameter models, arguing that the real uses of such models are research-related, including: analysis of data, testing of hypotheses in conjunction with field studies and improving our understanding of processes and their limitations. The large data requirements of such models essentially limit their use to well-instrumented test catchments. For management purposes, simpler models that require fewer data and have clearly stated assumptions may be a more realistic approach.

Hydrological data

It is becoming increasingly accepted that the complexity of hydrological models used for prediction should not exceed that warranted by the information content and accuracy of the field data (Jakeman and Hornberger 1993). However, overly complex models continue to be reported and used, and it seems that the appropriate level of complexity warranted is still being over-estimated. While more complex models can provide more information than just streamflow prediction (spatial distribution of soil moisture content, for example), they require more-extensive testing and so-called validation. Therefore, such models can be reliably tested only in well-instrumented catchments.

Hydrological information for the Integrated Water Reources Assessment and Management project

For the Integrated Water Resources Assessment and Management (IWRAM) project, the hydrological focus was on volume of streamflow. As such, the model developed addressed this issue only. Other potential issues such as water quality (including turbidity, sediment load, nutrients, heavy metals, pesticides and pathogens) and groundwater resources were not considered. Inclusion of such issues would require a more-complex hydrological model that simulated the effects of management options on these aspects of the system. For example, in areas with significant groundwater extraction, then the impact of changes in the extraction rate on the groundwater level would have to be included within the model, so that future availability of groundwater, as well as the impact of falling groundwater levels on streamflow, could be evaluated.

For an integrated model that is required for integrated assessment purposes, information on streamflow is needed by that model at locations that have no recorded streamflow. In such cases, the hydrological model component is required to estimate the flow at these sites, requiring methods for estimating the values for the model parameters. This can be done using regionalisation techniques, where the parameter values for gauged sites are related to catchment attributes, thus permitting the attributes for the ungauged catchment to be used to estimate the parameter values. An alternative approach was adopted within the IWRAM decision support system (DSS), where deep drainage and run-off estimated by the crop model were used to adjust the values of the parameters in the hydrological model.

Choosing suitable models

The requirement of the hydrologic component of the IWRAM project was a model capable of: showing sensitivity to broad-scale land-cover changes; predicting hydrologic response over a range of spatial scales from tens to thousands of square kilometres; incorporating a parsimonious approach to model parameterisation; partitioning flow between quick flow (dominant during the wet season) and slow flow (dominant during the dry season); and allowing parameter values to be related to catchment attributes in ungauged catchments. The catchments to which the procedure was applied are sparse in hydrologic and climatic data. The above factors strongly influenced the selection of an appropriate model structure. There were few data to support complex representations of the hydrologic system, let alone verify the performance of such models.

Physics-based models were deemed to be not applicable in the catchments used in this study. Despite the benefit of using such models—that is, the use of measurable properties potentially reduces the need for calibration—data limitations in the catchment studied here prevent application of these models. Conceptual models provide a much more appropriate alternative. The IHACRES metric–conceptual rainfall–run-off model (Jakeman et al. 1990; Jakeman and Hornberger 1993) is the basis for the hydrological modelling in the Mae Chaem catchment. This is a lumped model that considers the catchment as a single unit (though the parameter values were adjusted for changes in land use based on the results from a semi-distributed crop model). As an alternative, a distributed model based on the United States' Soil Conservation Service (SCS) curve-number approach was also developed and applied in the P37 catchment, giving the hydrological module greater spatial sensitivity at the cost of increased complexity.

The IHACRES model

The IHACRES rainfall–run-off model has been applied across a wide range of climates and catchment sizes. It has a parsimonious approach to model parameterisation (six parameters in the version used in this project). This parsimony facilitates regionalisation to ungauged catchments. Simple catchment attributes, such as forest cover area and catchment area, can be used to regionalise its parameters and thereby predict streamflow in ungauged catchments (e.g. Post et al. 1998; Post and Jakeman 1999).

The IHACRES model consists of a non-linear loss module that converts rainfall to rainfall excess, and a linear routing model that converts the rainfall excess to streamflow (Figure 6.2).

There are several formulations developed for the non-linear loss module—see Jakeman and Hornberger (1993), Ye et al. (1998) and Croke and Jakeman (2004). All of these formulations calculate the amount of rainfall excess based on the input rainfall and a catchment moisture indicator (s_k). Typically, the non-linear loss module has three parameters, though the model of Ye et al. (1998) has five (additional parameters needed to model ephemeral catchments in Australia). For the IWRAM project, the Jakeman and Hornberger (1993) form was used,

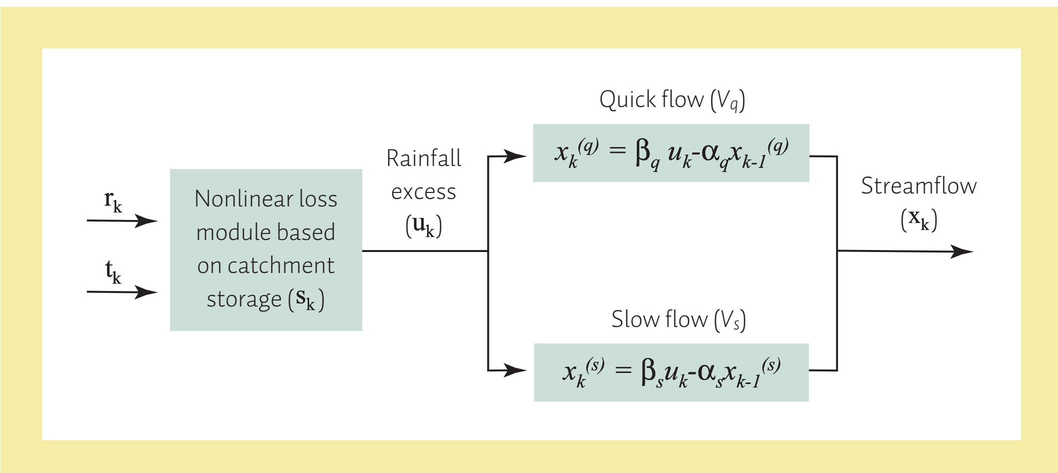

Figure 6.2 Generic structure of the IHACRES rainfall–run-off model. The climate inputs are rainfall (r_k) and temperature (t_k), though the temperature can be replaced by an estimate of the potential evaporation or potential evapotranspiration if this is available.

with the non-linear loss module comprising a storage coefficient c, a time constant for the rate of drying τ_w of the catchment at a fixed temperature (20°C), and a factor f that modulates τ_w for changes in temperature.

The linear routing module converts the rainfall excess (u_k) into modelled streamflow (x_k) using a unit-hydrograph approach. The usual structure to represent the hydrograph accurately is two exponentially decaying stores in parallel, representing quick- and slow-flow components (as shown in Figure 6.2), though for ephemeral catchments where the baseflow component is very weak or absent, a single store can be used. Each storage is characterised by a time constant (or equivalently the rate) of its unit-hydrograph recession (α_q and α_s). The proportional volume of the quick-flow (v_q) to slow-flow (v_s) storage response completes the parameterisation of the linear routing model.

The IHACRES model assumes that the partitioning of rainfall excess into quick- and slow-flow components is constant and thus does not depend on rainfall amount or intensity, or catchment condition. This assumption is inherent in any rainfall–run-off model incorporating a constant unit hydrograph approach. In order to represent the influence of land use on the strength of the slow-flow component, estimates of the run-off and deep drainage derived using the crop model (Chapter 7) were used to modify the quick- and slow-flow volumes (see Figure 6.3). The influence of land use on volume of streamflow produced was included in the integrated model by varying the catchment storage coefficient c by the variation in the combined run-off plus deep drainage calculated using the crop model. This technique was also used to estimate the model parameters for ungauged sites (sites where information on the streamflow was needed by the integrated model, but no stream gauging had been carried out).

Direct calibration and regionalisation results

The hydrologic module was developed and tested in sub-catchments of the Mae Chaem catchment in northern Thailand (Figure 6.4). In the Mae Chaem catchment, rapid agricultural intensification, rural development initiatives, and government conservation policies have created points of tension in relation to land- and water resource management. Environmental and social issues of particular relevance for the Mae Chaem catchment are the distribution of dry-season flows between upland and lowland farmers, increased rates of erosion from agricultural land and surface water quality.

Results of using the combined IHACRES model and the crop model to predict flows at ungauged sites and in response to land-cover changes are reported comprehensively in Croke et al. (2004). Procedures of direct calibration to stream-gauge data and regionalisation from any one gauge were undertaken for three sub-catchments: Kong Kan, Hai Phung and Mae Mu (Figure 6.4).

As a benchmark we undertook direct calibration for the Kong Kan site from its gauged rainfall–discharge time series for the period of available records (1985–1994). Reasonable model performance was obtained—except for the 1987–1988 hydrological year (Figure 6.5). The bias (in mm) in simulating Kong Kan ranges from 0.4% of annual rainfall in 1986 to 18% of annual rainfall in 1987.

Next we used the gauged data and IHACRES model for each of the three catchments in turn as reference catchments in order to regionalise the flows in the other two catchments. Figure 6.6 shows volumes of observed versus predicted discharge for each hydrologic year and its wet- and dry-season divisions. The procedure

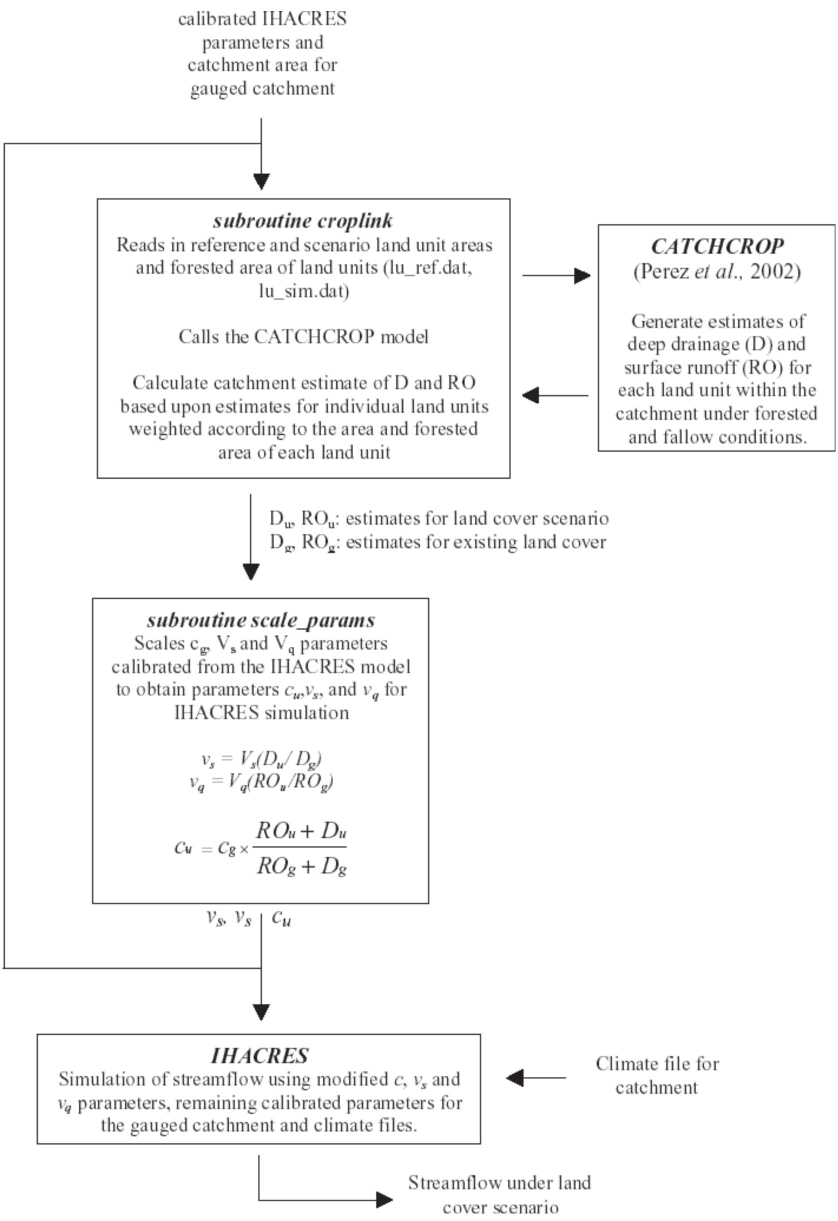

Figure 6.3 Flow diagram outlining procedure used to estimate the effect of land use change on streamflow at gauged sites, as well as estimation of the flows at an ungauged site. Source: Merritt et al. (2004)

Figure 6.4 Location of discharge gauges and the ungauged Mae Uam sub-catchment used to test the regionalisation procedure and model response to forest-cover changes. The large dot gives the location of Mae Chaem city. Source: Merritt et al. (2004).

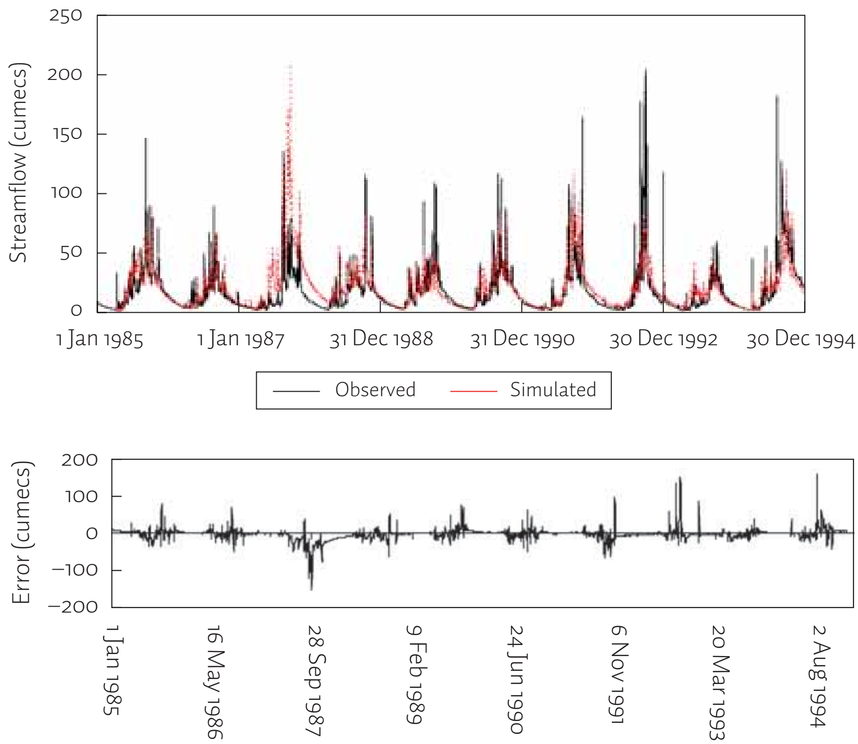

Figure 6.5 Observed and predicted streamflow for the Kong Kan sub-catchment using direct calibration, and the error in modelled flow for the simulation period (1985–1994)

seems capable of predicting the year-to-year flow pattern for all three sub-catchments. This is more evident in the estimates of wet-season and annual discharge than in the dry season. In the dry season, discharge estimates for the Kong Kan sub-catchment are between 57% and 95% of observed discharge when simulating from Huai Phung and over-estimated by between 9% and 70% when regionalising from Mae Mu (Figure 6.6a). For the Huai Phung sub-catchment (Figure 6.6b), the dry-season performance is poor. Whatever the reference catchment, neither magnitude nor relative flow pattern is being captured. In the wet season, the relative increase in discharge with increasing rainfall is much improved and the predicted magnitude of discharge is superior to that for the dry season.

Patterns in annual, wet- and dry-season discharge are captured reasonably well for simulations of Mae Mu, except for the wettest years (Figure 6.6c). The performance in the dry season where regionalising information from both Kong Kan and Huai Phung does lead to an under-estimation of dry-season flows by, on average, between 28% and 42% based on Kong Kan and Huai Phung calibrated parameters, respectively.

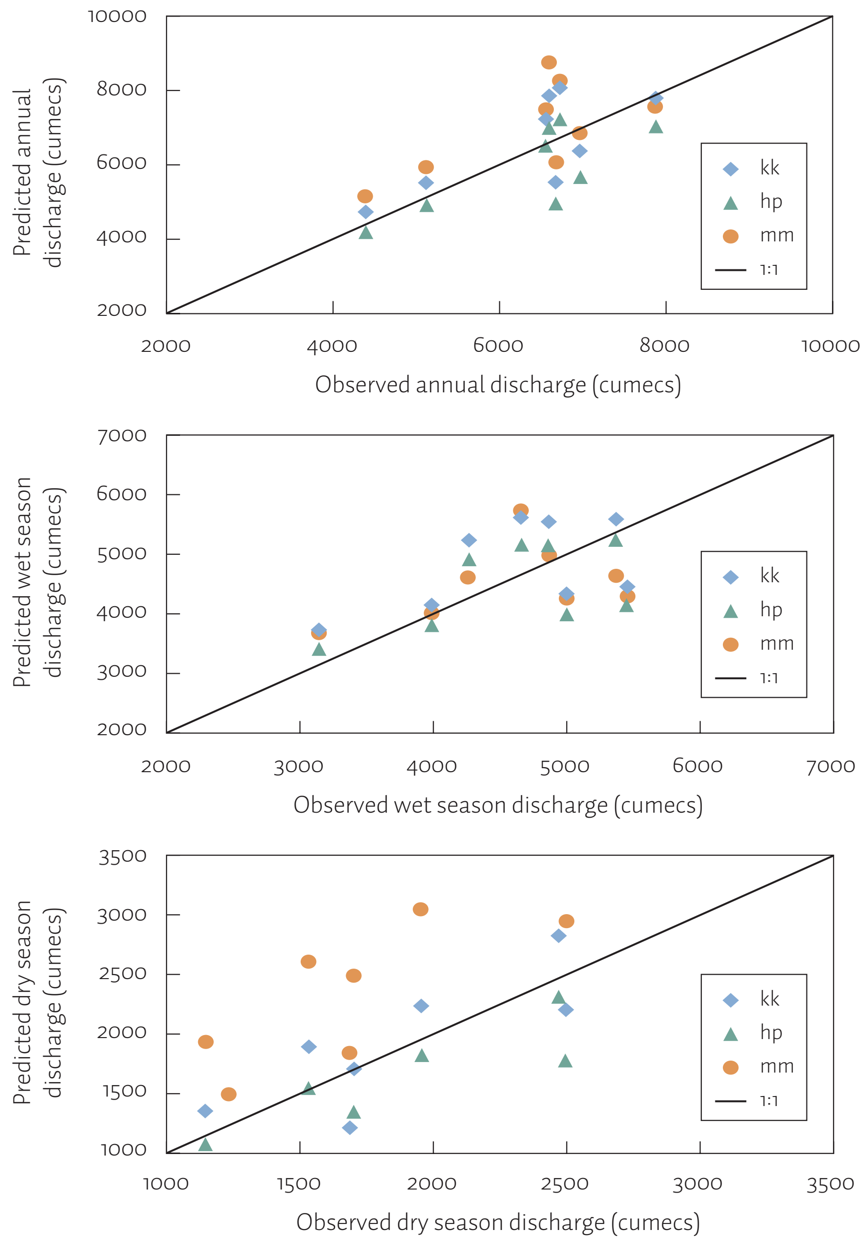

Figure 6.6a Observed versus predicted annual, wet-season and dry-season discharge for Kong Kan. Estimates are provided for all reference sub-catchments: Kong Kan (kk), Huai Phung (hp) and Mae Mu (mm).

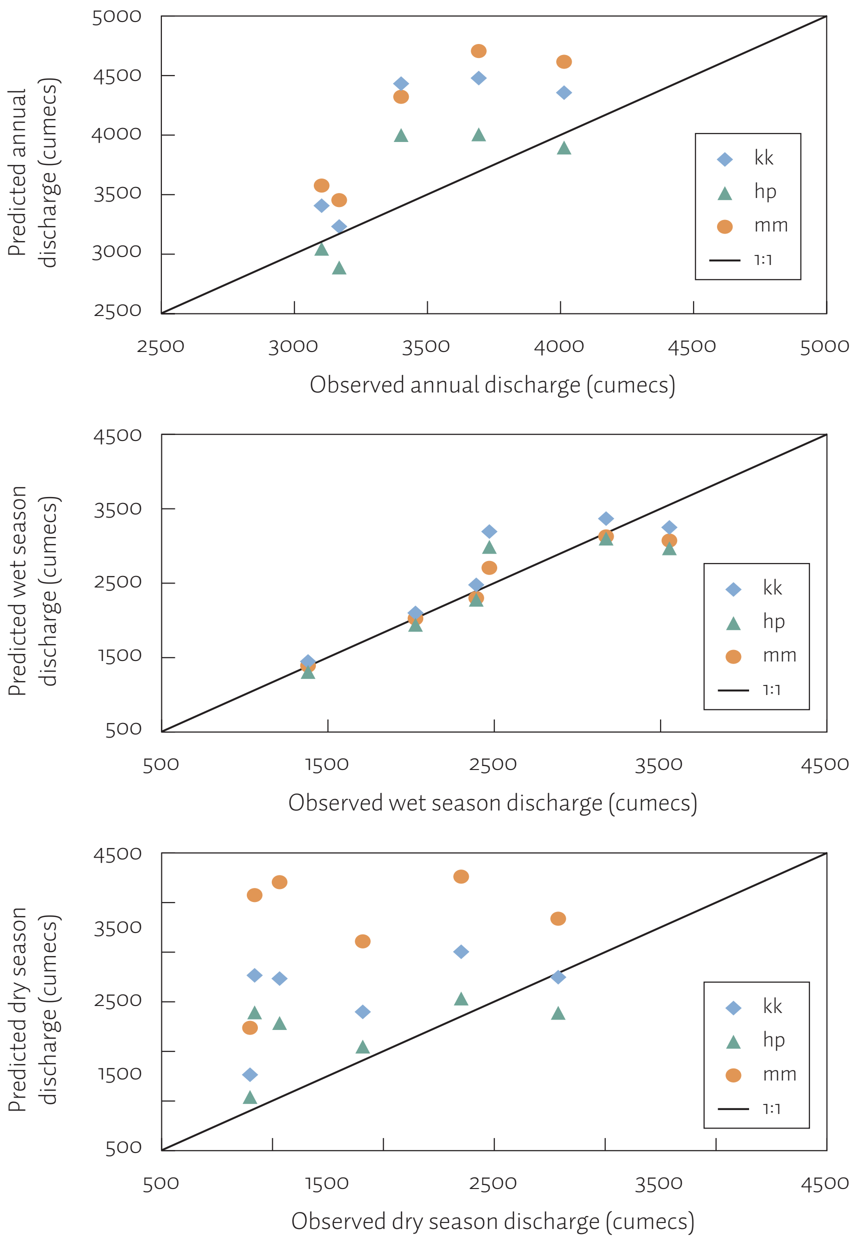

Figure 6.6b Observed versus predicted annual, wet-season and dry-season discharge for Huai Phung. Estimates are provided for all reference sub-catchments: Kong Kan (kk), Huai Phung (hp) and Mae Mu (mm).

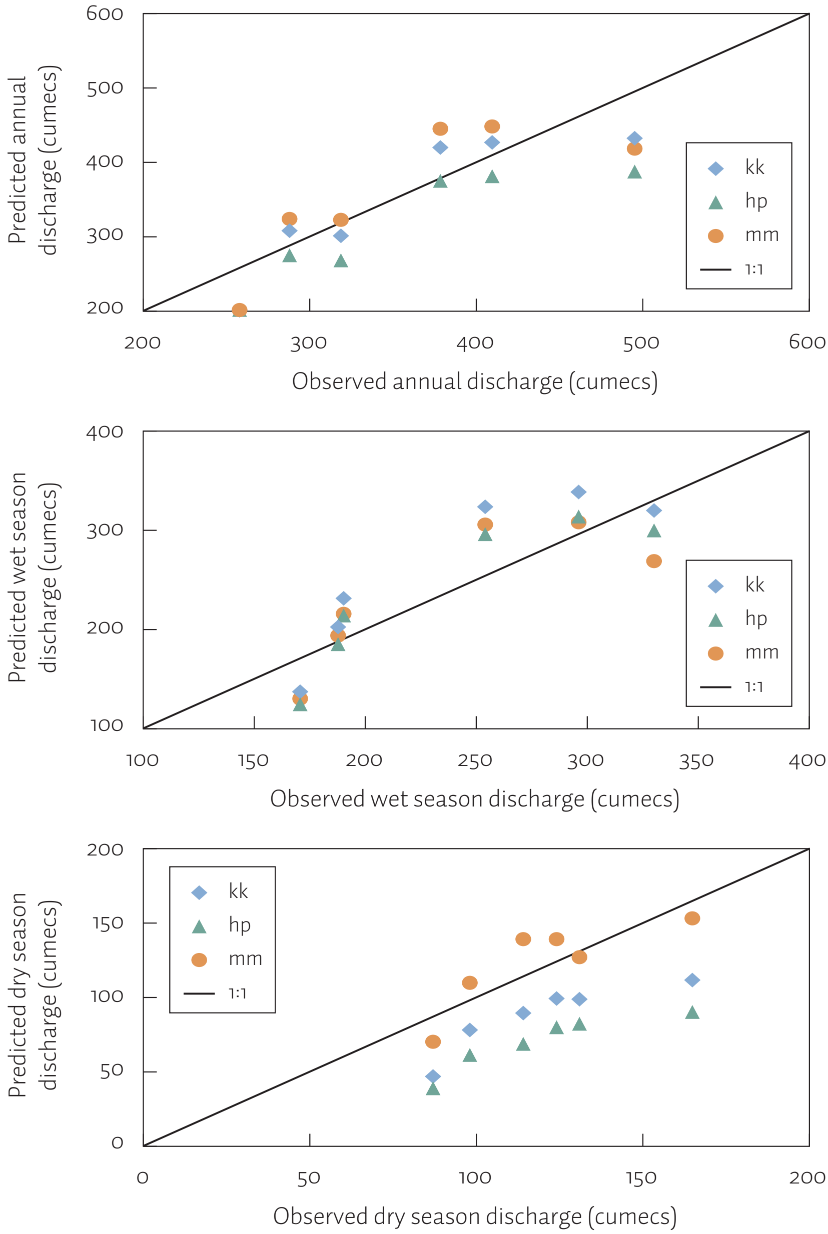

Figure 6.6c Observed versus predicted annual, wet-season and dry-season discharge for Mae Mu. Estimates are provided for all reference sub-catchments: Kong Kan (kk), Huai Phung (hp) and Mae Mu (mm).

This partial success of the regionalisation approach warrants its testing in other catchments with higher-quality rainfall and discharge data. Nevertheless, it remains useful in the current situation.

Predicting flows under forest cover changes

The influence of changes in forest cover on the quick- and slow-flow volume components of the IHACRES model were investigated for the 45.3 km^2 Mae Uam sub-catchment. The sub-catchment is ungauged, although the Land Development Department in Thailand provided land-unit information for this catchment. Thus, the regionalisation approach described above was used to model flows in the Mae Uam sub-catchment, using the gauged Mae Mu sub-catchment as the reference catchment. As none of the gauged sub-catchments had significant forest-cover change over the period of record, Mae Uam was selected to look at the model response to forest change. The catchment is largely dominated by steeply sloping, loamy soils in the upper catchment (land units 47 and 49), with gently sloping paddy land and mid-sloping gravel soils in the lower catchment.

Twelve scenarios of forest conversion were run to illustrate the effect of forest cover on the catchment estimates of drainage and run-off and hence the impact on the quick- and slow-flow volume components of the IHACRES model and predicted streamflow. The net change in forest cover in Table 6.1 is in relation to the forest cover in 1990 (sc1) where forest cover is 90.4% of the catchment. Table 6.1 illustrates the impact of forest cover on mean annual, wet season and dry season discharge under the same climatic series over the period 1985 to 1993.

Decreasing forest cover increases the catchment estimates of surface run-off while decreasing deep drainage for an average rainfall period corresponding to the 1990 hydrologic year (Figure 6.7a). Given our assumption that the slow-flow volume component of the IHACRES model, *vs*, is dominant during the dry season—where the majority of streamflow derives from water that has percolated through the soil subsurface—deforestation increases the quick-flow component, *vq*, relative to the slow-flow component, *vs* (Figure 6.7b). With decreasing forest cover, increases are seen in the total annual discharge predicted by the procedure. The increase in annual discharge from 1990 land-cover conditions (sc1) to complete deforestation (sc12) corresponds to 1214 ML in the driest hydrological year (April 1989–March 1990) and 3592 ML in the wetter hydrological year of April 1985–March 1986.

The response of the hydrologic model to forest-cover scenarios is consistent with other observations in the literature. Changes in annual, wet-season and dry-season discharge under deforestation scenarios in Mae Uam show limited response in discharge until forest removal of the order of 13%. From the literature, it appears that, at least in small catchments, a change in forest cover of approximately 20% is necessary before changes in streamflow are observed (e.g. Bruijnzeel 1990; Johnson 1998). This suggests that the hydrologic module is sensitive to forest-cover changes to a degree similar to that observed in the field. In large catchments or at basin scales, the change in forest cover required to observe changes in hydrologic response is not well established. Some literature has identified that changes in hydrologic response in large catchments may not be obvious even with large forest-cover changes (e.g. Wilk et al. 2001).

Table 6.1 Effect on discharge of land-cover scenarios and change in forest cover from 1990 (± afforestation). Also shown is percentage change from the 1990 land cover scenario (sc1) for mean-annual, wet-season and dry-season yields (– indicated as the decrease from sc1). Yields (in ML) are provided for sc1.

	Description		Discharge (ML)		
			Mean annual	Wet-season	Dry-season
sc1	1990 forest cover	––	18,271	13,433	4838
			Net change (%)		
sc2	100% forest cover on all land units	+9.6	−1.2	−2.5	2.2
sc3	0% forest cover on paddy fields (land units 88 and 99)	−3.0	0.3	0.3	0.4
sc4	70% forest cover on land units with slopes less than 16° (land units 23, 88, and 99)	+1.9	−0.2	−0.2	0.3
sc5	50% forest cover on land units with slopes less than 16° (land units 23, 88, and 99)	−0.6	0.1	0.3	−0.5
sc6	70% on land unit 49 (slopes greater 35°)	−13	2.4	4.5	−3.5
sc7	70% forest cover on land units with slopes less than 35° (land units 23, 25, 45, 47, 88, and 99)	−8.1	2.3	4.0	−2.5
sc8	50% on land unit 49 (slopes greater 35°)	−21.2	3.9	7.4	−5.7
sc9	70% on all land units	−20.4	3.2	8.4	−11.1
sc10	50% forest cover on land units with slopes less than 35° (land units 23, 25, 45, 47, 88, and 99)	−8.1	2.3	8.3	−14.4
sc11	50% on all land units	−40.4	6.2	16.0	−21.1
sc12	0% on all land units	−90.4	13.6	36.6	−50.2

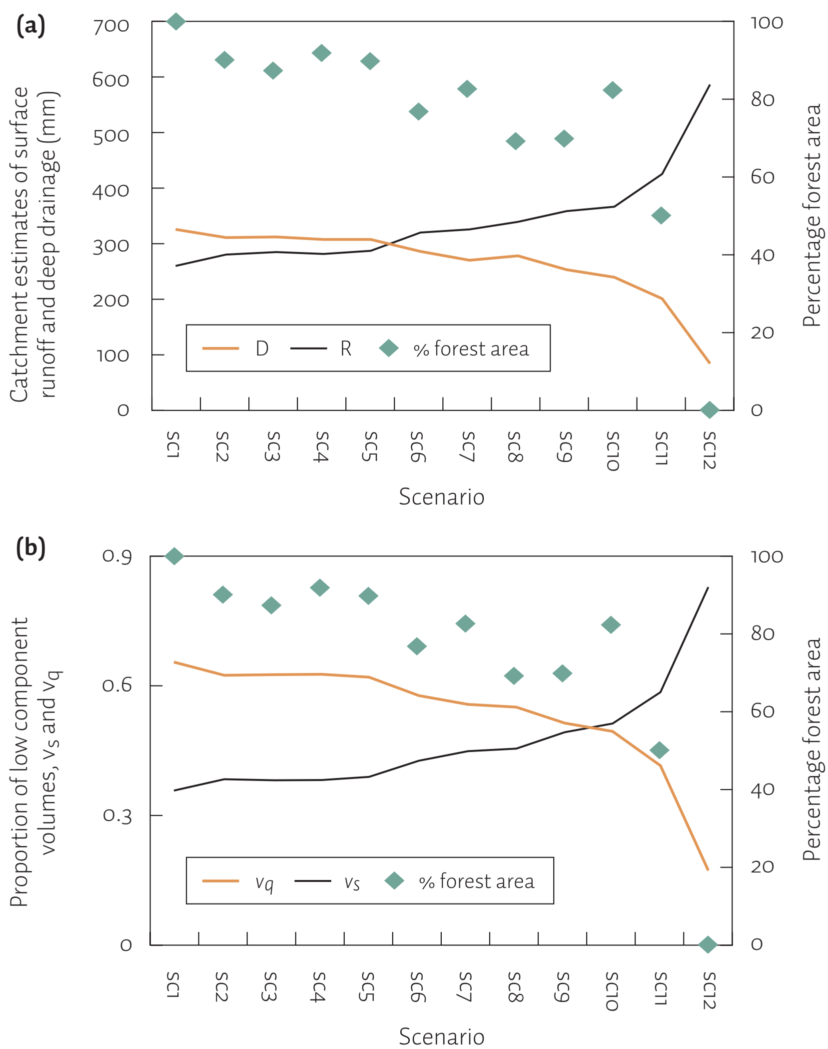

Figure 6.7 Effect of forest-cover-change scenarios on (a) deep drainage (D) and surface run-off (R) estimates for Mae Uam, and (b) the IHACRES quick (*vq*) and slow (*vs*) flow volume components. Total forest area under each scenario is also shown (◆).

US Soil Conservation Service curve-number approach

The IHACRES model is a lumped parameter rainfall–run-off model. As a result, there is no representation of the spatial variability in the catchment included within the model (though this may be included in the model parameters if these are estimated by a spatially distributed cross model). One alternative would be to divide the catchment into zones with similar hydrologic response (hydrologic response units) and run the non-linear loss module separately on each of these (e.g. Carlile et al. 2002). Another alternative is to use a hydrological model that attempts to model the spatial movement of water through a catchment in addition to the temporal movement out of the catchment. An example of such an approach is presented here using a simple flow-generation algorithm based on the SCS curve-number approach. The SCS approach uses empirically derived 'curve numbers' that can be used to estimate the run-off generated based on the combination of soil properties, topography and vegetation cover, as well as antecedent moisture, at a particular site.

While the SCS curve number approach was developed in the USA, the method has been employed across many regions of the world, though care must be taken to check that the coefficients apply in each region. Small experimental watersheds at Rayong (southeastern Thailand) were selected for construction of the model, with regression relationships for the variation in the streamflow recession rate as well as the streamflow volume being defined, based on the observed time series (Witthawatchutikul et al. 1985).

The primary purpose of this model is to simulate the influence of the pattern of land use across the catchment, rather than just the relative fractions of each land use within the catchment, and to produce a spatial map of the effective rainfall needed by the crop model. This gives both the hydrological and crop models greater sensitivity to the spatial pattern of land use.

This model is currently under development, and when completed, it will provide the IWRAM DSS with additional predictive powers and flexibility (Witthawatchutikul et al. 2005).

Integrating the hydrology model into the Integrated Water Resources Assessment and Management decision support system

The requirement of the hydrologic component of the IWRAM DSS was a model capable of showing sensitivity to broad-scale land-cover changes, and of predicting hydrologic response over a range of spatial scales from tens to thousands of square kilometres. The catchments in northern Thailand that were being studied were sparse in hydrologic and climatic data, and this prevented any complex representations of the hydrologic system from being applied.

The hydrology model is a key component of an integrated framework for water resource assessment, as it provides the volume (and timing) of water for irrigation of crops.

The IHACRES rainfall–run-off model had previously been applied across a wide range of climates and catchment sizes and requires only a small set of parameter values. Also, simple catchment attributes, such as forest-cover area and catchment area, can be used to regionalise these parameters and thereby allow the prediction of streamflow in ungauged catchments. This made the IHACRES model particularly suitable for incorporation in the IWRAM DSS. One limitation is that the model cannot easily represent spatial variability in the catchment. Therefore, another model that can represent spatial variability and uses an algorithm based on the SCS curve-number approach is also being developed. Of course, the increased complexity of this model has the drawback that it has increased data requirements. Hence, it is more difficult to apply in catchments where there is little or no monitoring taking place.

The two hydrologic models developed for the IWRAM DSS are focused on the availability of surface water in rivers and streams for crop irrigation. Other potential issues such as water quality (including turbidity, sediment load, nutrients, heavy metals, pesticides and pathogens) and groundwater resources were not considered. Inclusion of such issues would require a more-complex hydrological model that simulated the effects of management options on these aspects of the system as well. For example, in areas with significant groundwater extraction, then the impact of changes in the extraction rate on the groundwater level would have to be included within the model, so that future availability of groundwater as well as the impact of falling groundwater levels on streamflow, could be evaluated. Nevertheless, such inclusions would not change the basic framework of the DSS, just some details in the component models.

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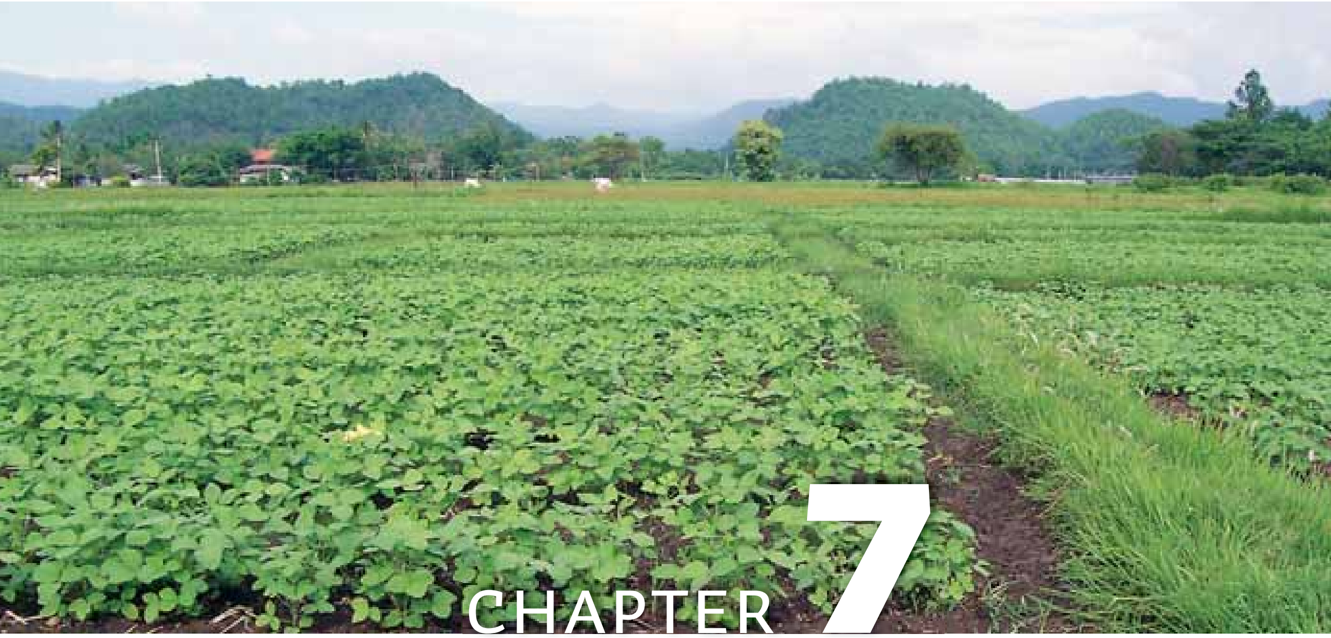

CHAPTER 7

Determining crop yield and water use

Wendy Merritt, Barry Croke and Somjate Pratummintra

Summary

A key component of an integrated model for land and water resource assessment in agricultural districts is a crop model that is capable of providing estimates of crop water-use and of seasonal crop yields. The crop model provides the link between land use, water management, economic costs and benefits, and environmental impact.

Crop models can vary from simple, empirical growth functions to more complex mechanistic models that simulate the chemical and physical processes of plant growth.

This chapter will focus on crop-modelling approaches suitable for inclusion in an integrated framework for water resources assessment. In particular, it will describe the two crop models developed for the Integrated Water Resources Assessment and Management project in northern Thailand.

Figure 7.1 Agricultural fields in the Mae Chaem catchment after harvesting

The role of crop models

In terms of their role, crop models can be split into three main groups: research; crop systems analysis at a farm level; and policy analysis at a catchment or regional level. Crop models can also be used for educational purposes.

Research models have tended to focus on linking the physical processes (such as the availability of sunlight, water and nutrients) with the more traditional discipline of plant physiology. Relative to the other roles of crop models, the function of research models, as a tool for understanding plant-physiological processes, ensures that such models are generally more complex than those developed for farm management, catchment policy analysis or educational purposes.

Crop models that are developed to assist farm management are used to assess alternative crop practices and assist decision-making, for issues such as water use, fertiliser use, erosion control, and pesticide use (Boote et al. 1996). These models have also been incorporated into decision support systems (DSS) to provide an integrated assessment tool that can be used for developing optimal farm-management strategies.

Crop models also have the potential to be used as policy analysis tools. For example, the use of crop models to develop land suitability classes may be applicable in development of land use planning policy. In particular, crop-simulation models or some form of crop yield relationships are being increasingly applied to assess yield potentials of crops at regional or greater scales. The crop model used to predict yield for each land unit can vary from simple empirical growth functions (Liengsakul et al. 1993) to the incorporation of more complex crop-simulation models (Bouman 1994; Roetter et al. 1998).

This chapter will focus on crop-modelling approaches suitable for inclusion in an integrated framework for water resources assessment at a sub-catchment or catchment scale.

Crop modelling approaches

Two distinct model classifications have been presented in the literature. Models have conventionally been classified according to the methodology by which they are developed as:

- mechanistic—processes are described with explicit biological and physical functions
- empirical—processes are described with statistical fitting functions.

As with other modelling disciplines, most crop models are neither purely mechanistic nor empirical, rather they contain a mix of both approaches.

Empirical models

Empirical approaches include simple linear, non-linear and multivariate analyses used to fit historical yield data to average temperature and precipitation records. Perhaps the greatest disadvantage with empirical approaches to crop modelling is that they tend to be site specific. That is, the relationships used to predict yield for one site may not be valid for sites with different conditions. Despite this, empirical models have the potential to remain an important tool for land evaluations and yield prediction. This is especially so in areas where it is inappropriate to apply more-complex models due to data limitations. Hence, these models are still used widely and are likely to continue serving a purpose for some considerable time to come. This is enhanced by the ease with which these models can be applied, thus increasing their attractiveness for policy or decision-makers. Care must be taken to ensure that these models are not applied outside conditions for which the model was developed.

Mechanistic models

Mechanistic models range in their complexity and specificity in representing the biological and physical processes controlling plant growth. They can be further classified into sub-groups of *crop specific* and *generic* models.

Mechanistic models tend to allow dynamic simulation on a number of time steps and in-depth consideration of the processes underlying crop growth. Consequently, these models are more complicated and computationally demanding than empirical models.

An advantage of mechanistic crop models is that the explicit relationships within the model have a physical basis. However, even the most process-oriented crop models still contain empirically determined constants or relationships.

Use of mechanistic models is potentially constrained by a lack of physical data for calibration and validation.

Intermediate approaches

In practice, most models represent a compromise between rigour and utility. In other words, crop-simulation models are generally neither purely empirical nor mechanistic. These ‘intermediate approaches’ are very useful for resource-management evaluation if correctly constructed, and provide a good compromise between empirical and mechanistic models.

Comparative analysis of crop-modelling approaches

The applicability of a modelling approach is determined by a number of factors, the two most important being:

- the intended use of the model
- data availability.

Intended use

Perhaps the most important factor in determining the appropriateness of a model is its intended use. This determines the processes to be considered and their level of detail, and the model accuracy required. For example, an emphasis on erosion–productivity requires detailed consideration of soil processes but this may not be as important for, say, pest damage studies. Intended use also determines the complexity of the model—that is, the number of processes to be included and the level of detail. For example, if an annual crop yield is all that is required, a relatively simple empirical approach may be perfectly adequate, if not more appropriate, than a more-complicated, mechanistic approach.

Data availability

In catchment- or regional-scale studies, the issue of data availability becomes of utmost importance. Mechanistic models often require a large amount of physical data, such as a variety of soil parameters, which are rarely collected during land surveys and are available at only a few experimental sites. Empirical models tend not to require such large quantities of data and are computationally simple, but have limited meaning. Therefore, intermediate approaches may represent a suitable compromise between data requirements and physical meaning. Problems of data availability are exacerbated in developing countries, where detailed information for supporting complex models is less-often collected.

Incorporating crop models within integrated modelling frameworks

Currently, there is also a need for the development of catchment-scale approaches that integrate agronomic factors (crop growth) with socioeconomic and land-degradation factors. A number of complexities must be addressed when developing such an integrated approach. These can be summarised as:

- the large number of crops that are grown within a catchment
- the different types of cropping systems within the catchment
- the different scales of analysis at which the system can be modelled
- distribution of water within the catchment
- the accentuated problem of data availability.

Realistically, it is not possible to model every single crop grown within a catchment, so some simplification of the system is necessary.

Approach for crop modelling in northern Thailand

The crop-modelling approach used in the Integrated Water Resources Assessment and Management (IWRAM) project for northern Thailand needed to be directly linked with the socioeconomic and physical models within the integrated DSS, with particular emphasis on scenario simulation. The objective of the crop modelling was to develop an understanding of both yield variability over time and water use for a range of crops typically grown in northern Thailand. It also needed to simulate yield response to water deficit and fertility depletion, both of which are important factors determining final yield in this region. The model was to be linked with an economic model, so that the relationships between farmers' decisions and variable production of different crops could be explored.

There was little need for complex models that considered large numbers of processes, primarily because of the limited amounts of data available for use within the catchment. This indicated that a relatively simple, crop-yield model, capable of simulating crop stages throughout the season and yields at the point of harvesting, would be most suitable. The outputs required were crop water-use through the season and final crop yield. Two alternative crop models were adapted for this application, the CATCHCROP model (Perez et al. 2002) and a crop model developed for the Food and Agriculture Organization of the United Nations (FAO 1978). Both of these models were tested in the IWRAM DSS, and are described in detail below.

Figure 7.2 Beans are not only a valuable cash crop but also increase fertility of the soil by fixing nitrogen. Photo by Anthony Scott, June 2004

The CATCHCROP model

The CATCHCROP model can predict crop water-use in addition to crop yield for a number of different crop types. It was developed in response to the recognition that many existing crop models required large amounts of highly specific data, such as detailed soil information (e.g. conductivities of each soil layer, and cation-exchange capacity), to drive them. These are rarely collected outside experimental stations.

CATCHCROP is a plot-based model that is applied over areas considered homogeneous in terms of soil, crop and climate properties and inputs.

The model involves a number of sub-routines (Figure 7.3) whereby:

- run-off over a 10-day time step is estimated
- water balances are constructed for the reservoirs of soil and crop available and for deep drainage
- maximum, sub-optimal and actual evapotranspiration are calculated at the current time step
- water demand for the next 10-day time step is calculated.

At the end of each season, yield is calculated according to a crop's potential yield, the water stress of the crop, and the ratio of actual and maximum evapotranspiration.

CATCHCROP is a simplified conceptual crop model that attempts to account for the effects of soil type, fertility, landform and water availability on crop yield. It does not attempt to include the radiation limits to crop growth (i.e. the model assumes that growth is limited by soil characteristics and water availability only).

Applying CATCHCROP to the Mae Chaem catchment in northern Thailand

In the IWRAM project, CATCHCROP was applied on a land-unit basis, where each land unit is considered homogeneous in terms of soil, crop, climate properties and other inputs. The Mae Chaem is a complicated agricultural catchment; over 100 crops have been grown within it (Scoccimarro et al. 1999). Not only is the number of crops large, but also the types of crops grown are varied, ranging from rainfed crops to irrigated crops such as paddy rice, and from annual crops to perennial crops. For purposes of simplification, the crops that were considered in the IWRAM DSS were generally limited to the major crops found in the catchment: upland and paddy rice; soybean; groundnut (peanut); maize for grain or forage; cabbage; potato; onion; and temperate and tropical fruit trees. Despite this simplification, the model still needed to account for a mix of irrigated, rainfed, annual and perennial crops.

An example of the outputs produced by the CATCHCROP model is shown in Table 7.1 for the Mae Uam catchment. In this example, the model was run outside the DSS, so the water available for irrigation was unknown. Thus, the amount of water used for irrigation was set to the water demand for each crop. The yield estimates for wet-season rice were very close to the average observed value, as were the run-off estimates, suggesting that the model assumptions were adequate for this purpose. The yield estimate for soybean was slightly high, possibly due to the assumption that the irrigation equalled the water demand (i.e. unrestricted water availability in the dry season).

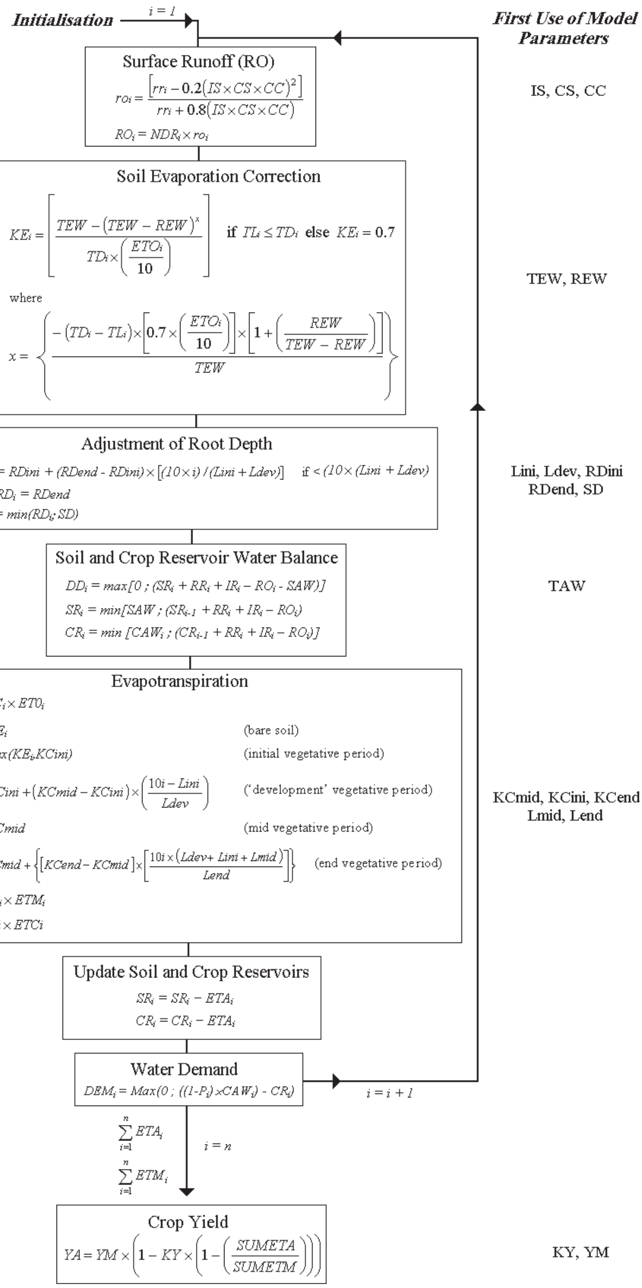

Figure 7.3 Detailed flow-chart of the CATCHCROP model. From Merritt et al. (2004)

Table 7.1 Average simulated yields and water balance derived using the CATCHCROP model for the Mae Uam catchment, 1988–1992. From Perez et al. (2002)

	Paddy field		Upland field	
	Wet-season rice	Dry-season soybean	Wet-season cabbage	Dry-season cabbage
Irrigation (mm)	150	233	0	339
Run-off (mm)	93	17	621	37
Percolation (mm)	409	12	35	0
Evapotranspiration actual (mm)	675	241	485	320
Yield (kg/ha)	3207	1419	15284	16184

The performance of the CATCHCROP model in estimating the water balance was also tested by Perez et al. (2002) for the Mae Mu catchment, which has almost 100% forest cover (see Table 7.2). Generally, the model was able to reproduce the seasonal discharge volume, though there is a tendency for the model to over-estimate the dry-season flows.

Sensitivity analyses have been performed on the CATCHCROP model by Merritt et al. (2005) for a range of different management practices

Table 7.2 Comparison of simulated and observed discharge (mm) for Mae Mu catchment. From Perez et al. (2002)

Year	Season	Run-off	Drainage	Simulated discharge	Observed discharge
1988	Wet	452	174	452	423
	Dry	26	3	203	213
1989	Wet	361	224	361	381
	Dry	0	0	224	147
1990	Wet	358	148	358	328
	Dry	75	0	223	160
1991	Wet	252	82	252	243
	Dry	73	6	161	169
1992	Wet	305	111	305	245
	Dry	61	0	172	121

(presence or absence of bunding, fertiliser levels and irrigation status) and also for different crop and soil parameters. The model behaved as expected and the analysis indicated where the model structure could be simplified.

For irrigated crops, less irrigation was required with bunds, as water was retained within the plot. As crop water demand is less than the amount of water available for irrigation, significant differences in crop yields were not observed. Non-irrigated crops generally showed increased crop water demands and decreased actual evapotranspiration and deep drainage compared with irrigated crops. The greater the fertility of the plot, the more the crop was able to transpire. Hence, crop water demand and actual evapotranspiration both increased with more fertile plots, thus increasing crop yields. Deep-drainage estimates in plots that are bordered by bunds were considerably greater than in non-bunded plots.

Radiation- and water-limited crop-production model

The CATCHCROP model considers only the water limitations for crop growth, and not the radiation limitations. In order to make the crop model more broadly applicable (both spatially and in crops included), a second model is being developed for the IWRAM DSS. The new model is based on a crop-production model originally developed for the Agro-ecological Project of the Food and Agriculture Organization of the United Nations (FAO 1978).

General description

Crop production is estimated from the product of:

- the radiation-limited growth
- the water-limited growth
- a harvesting index for each crop
- a site index accounting for topography and soil characteristics (texture, structure and fertility).

The main components of the model are as follows:

Radiation-limited growth module. Crop production under optimal conditions (production potential) is calculated using a radiation model for each of the major crops being grown in the catchment. Growth is calculated from the amount of solar radiation intercepted by the leaves. This model assumes ideal water and nutrient supply, and a disease-free crop. Crops are divided into groups (I to IV) according to their photosynthetic pathway and optimum growth temperatures, with C4 plants generally having a higher heat tolerance than C3 plants (see Table 7.3).

Water-limited growth module. Because of several limitations on ideal growth rates, it is rare for a crop to reach full production potential. One of these is the availability of water. Water-limited growth is calculated using a water balance that provides an estimation of water availability in the soil layer.

The model provides similar outputs to the CATCHCROP model, including crop yield and a soil water balance. This model is still undergoing development and validation trials, and will be a useful addition to the suite of models available in the IWRAM DSS.

Table 7.3 Crop groupings according to photosynthetic pathways and optimum growth temperature

Group	Photosynthetic pathway	Optimum temperature (°C)	Examples
I	C3	15–20	Wheat, white potato and Phaseolus bean
II	C3	25–30	Soybean, rice, cotton and sweet potato
III	C4	30–35	Sorghum, maize, pearl millet, sugarcane
IV	C4	20–30	Maize (temperate and tropical high-altitude variety)

Note; C3 plants use a photosynthetic pathway in which the first stable compound formed from carbon dioxide (CO_2) is a three-carbon compound. C4 plants are so-named because the first organic compound incorporating CO_2 is a four carbon compound.

Integrating the crop models into the Integrated Water Resources and Management project decision support system

A purpose of the IWRAM DSS is to investigate the influence of land use scenarios on water availability (both locally and for downstream users), on the economics of crop production, and on environmental impacts (such as erosion rates). Different combinations of crops, soil and topography can have significant differences on the hydrological, environmental and economic impacts. As such, the DSS needs to be sensitive to the water demands of various crops on different land units included in each scenario, as well as the crop yield. The role of the crop module is to:

- supply the hydrology module with the effects of changes in land use so that the impacts of land use change on streamflow can be estimated
- estimate the amount of water extracted for irrigation, so that the impacts of the land use scenario on downstream users can be assessed
- provide the economic module with crop yields so that the economic return can be determined.

There are two key inputs that must be supplied to the crop model. These are the distribution of crops on the different land units within the catchment, and the water that is potentially available for irrigation use. The distribution of crops is supplied as part of the input data for each land use scenario being investigated, and comprises the fraction of the area of each land unit planted with each crop. The amount of water that is available for irrigation is supplied by the hydrology module. Where there are significant water storages within a catchment, the hydrology module can be used to predict the inflow to these storages as well as the natural flow in the irrigation areas. The release of water from the dams for irrigation then has to be included separately.

Conclusion

The crop models incorporated in the IWRAM DSS must provide a compromise between complex deterministic models with large data requirements, and overly simplistic empirical relationships. The CATCHCROP model meets these requirements and has been tested for sub-catchments in northern Thailand. It behaved as expected under different management conditions and parameter values and provides a useful tool for integrated water resources assessment on a catchment scale. A second crop model has been developed, based on a crop-production model originally developed by FAO (1978). This model has the advantage that it not only takes into account water limited growth, but also considers radiation-limited growth. However, this added complexity does have the disadvantage that the data requirements for the model are greater.

The crop model is a key component of an integrated framework for land and water resource assessment in agricultural districts because it provides estimates of both crop water use (which affects the catchment water balance) and of crop yields (which directly affects the economic costs and benefits of different land use scenarios). The crop model provides the link between land use, water management, economic costs and benefits, and environmental impacts.

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CHAPTER 8

Estimating the effects of changed land use and management on soil loss

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Summary

Accelerated soil erosion in the highland regions of the world is a result of land clearing and agricultural activities, and has been recognised as a serious problem in Thailand for over 30 years. The hills of northern Thailand have steep slopes and the soils are exposed to an erosive monsoonal climate for seven months of the year. The rate of erosion depends on the timing and amount of rainfall, the slope of the hillside, soil type, land use and land-management practices. Soil erosion can cause declines in agricultural productivity, reduce water quality in nearby streams, and cause siltation problems downstream.

Erosion models can be used to estimate soil loss from agricultural catchments, which can assist with soil-conservation planning, land use planning, soil-erosion inventories, and regulation. The erosion model used in the Integrated Water Resource Assessment and Management project decision support system was based on the universal soil loss equation (USLE), modified to suit conditions in northern Thailand. The USLE is widely applied worldwide and is also used by Thailand's Land Development Department for land use planning. The USLE is capable of providing estimates of potential erosion within the catchment, for different scenarios of land-management planning.

Introduction

Soil erosion by water is a natural process involving the detachment and transport of soil particles, caused by rainfall and overland flow of water. 'Natural' soil erosion has been occurring ever since the first soils were formed, but 'accelerated' soil erosion is a much more recent problem. It is a result of the unwise actions of humankind, such as the clearing of forests on sloping lands, overgrazing by domestic stock, and unsuitable cultivation practices, which leave the land vulnerable during times of heavy rainfall.

Accelerated erosion can result in rapid loss of topsoil, and this can cause decline in agricultural productivity. Eroded soil is washed into nearby streams and rivers, reducing water quality and causing siltation problems in the lower catchment.

Increasing awareness of the impacts of erosion has stimulated a large amount of research. There are three main focuses of this research: the erosion process itself, the effects of soil loss on crop productivity, and the effect of erosion on the water quality of nearby streams. Historically, much of the research was focused on the productivity of agricultural lands (e.g. Loch and Silburn 1996), but more recently there has been an increasing interest in the off-site impacts that sediment, and associated nutrients, have on water quality.

This chapter provides a brief review of erosion processes and the different forms of erosion that can take place. More-detailed information is then presented for the humid tropics, in particular for northern Thailand. The mathematical modelling of erosion is then introduced, followed by a description of the modified universal soil loss equation (USLE) computations that were applied in the Integrated Water Resource Assessment and Management (IWRAM) project.

Accelerated erosion leads to loss of topsoil

One analysis of soil erosion on a global scale, estimates that, depending on the region, topsoil is being lost 16 to 300 times faster than it can be replaced. Soil-making processes are extremely slow, requiring from 200 to 1000 years to form 2.5 centimetres of topsoil under normal agricultural conditions.

Processes of soil erosion by water

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

Detachment of sediment from the soil surface is caused either by the impact of rain droplets, or by the shear forces of overland flow. Rainfall-induced detachment will often be the dominant process on relatively flat regions of small extent. In regions with long, steep slopes, detachment is often dominated by the very high shear-stresses induced by fast-flowing overland flow.

Transport of sediment is initiated when detached particles are washed downstream along gullies, streams and rivers. As the velocity of flow (and hence the water turbulence) increases, larger soil particles will remain suspended in the water and the capacity for sediment transport increases.

Deposition of sediment is the final process in soil erosion. When there is not enough energy (or turbulence) to transport the sediment, it gradually settles out of the water and comes to rest. Sediment sinks, or depositional areas, can be visible as newly deposited silt or sand on a flood plain, as bars and islands in a river channel, and as mudflats at the mouth of a river.

Types of soil erosion

There are six main types of soil erosion by water: sheet, rill, gully and streambank, mass movement (or landslides) and road erosion.

Sheet erosion refers to the uniform detachment and removal of soil or sediment particles from the soil surface by overland flow evenly distributed across a slope. Sheet erosion is often considered to be the most serious type of erosion from an agricultural viewpoint as it tends to strip nutrients concentrated in the surface layer of the soil. This has the potential to lead to reduced fertility and decreased productivity.

Rill erosion occurs when water moving over the soil surface starts to concentrate down preferential pathways, forming an easily recognisable channel, or rill. These rills are defined as being 'small flow channels that can be obliterated by tillage'.

Gully erosion, in contrast to rill erosion, describes channels of concentrated flow too deep to be obliterated by cultivation. Gully development is controlled by thresholds related to slope and catchment area. Two main stages in gully development can be identified:

- There is an initiation period where there is rapid erosion and massive movement of sediment as the head of the incised gully moves rapidly up hill. The gully bottom is also scoured out and becomes deeper.
- This is followed by a period during which the gully bottom remains fairly stable, with equal amounts of scouring and sedimentation, while the gully width increases due to lateral erosion and collapse of the side banks.

Gullies have been identified as potentially contributing large amounts of sediment if connected to the river network.

Streambank erosion occurs along rivers and streams, particularly when riparian vegetation has been removed. The vertical side banks are undermined by the water flow until they collapse into the river.

Mass movements, or landslides, occur on steep slopes after intense rainfall periods. The soil weight is increased dramatically by saturation with water and exceeds its restraining capability. Alternatively, a zone of weakness in the underlying material is further weakened and lubricated by infiltrating water. Disturbances to slopes that increase the weight factor (such as large buildings or stockpiles of earth or rock), or reduce the restraining capability (such as road cuttings), will greatly increase the risk of failure.

Road erosion has the potential to be a significant source of sediment in some catchments. Four features of paved and unpaved roads that can increase erosion in mountainous catchments are:

- the highly compacted road surfaces and disturbed roadside margins reduce infiltration, thereby increasing surface run-off and the associated erosive forces
- road cuttings can intercept sub-surface flow then re-route it via overland flow mechanisms toward the stream channel
- ditches and culverts capture both sub-surface flows and surface run-off and channel it more directly to streams,
- road cuttings can reduce the strength of steep slopes and increase the risk of landslides.

The degree to which each factor contributes to erosion from a segment of road differs between sites and particular circumstances.

Characteristics of erosion in the humid tropics

The overall rate of soil erosion in Asia far exceeds that of any other region of the world (Chang 1993). Froehlich and Starkel (1995) note that rains in the humid tropics are more erosive than in temperate regions due to the high rainfall intensities that commonly occur during storm events. In the humid tropics, the number of thunderstorm days exceeds 30 per year, and in Bangladesh, southern Burma, southern Thailand, Malaysia and the western part of Indonesia this increases to more than 60 (Chang 1993). In these circumstances, the potential for the generation of overland flow, when rainfall intensity exceeds the infiltration capacity of tropical soils, is extreme. This excess run-off has been identified as a dominant source of erosion in the humid tropics, particularly on steep lands (Yu and Rose 1999).

The humid tropics are also under increasing pressure from rapid population growth in rural areas, and farming on steep lands has continued to increase in recent years, especially in developing regions of Southeast Asia. Steep lands have been identified as being highly prone to erosion. Traditional shifting-cultivation practices of long-rotation systems have, in many areas, been converted to more-intensive, shorter-rotation systems, thus presenting increased problems with soil fertility and soil erosion in the steeper areas (Turkelboom et al. 1997). In traditional shifting-cultivation systems, soil loss is generally very small as the roots of the fallow vegetation bind the soil together and help limit erosion.

In summary, erosion in humid regions can largely be attributed to the timing and amount of rainfall, the importance of overland flow and slope, and changes in land uses and land-management practices arising from increasing population pressures.

Extent and types of erosion in Thailand

Accelerated soil erosion in the highland regions of Thailand, as a result of land clearing and agricultural activities, has been recognised as a serious problem for over 30 years (Lal 1975; Liengsakul et al. 1993). Lal (1975) reported that the most serious erosion problems are in the northern highland region because of the rainfall patterns and landforms. The hills of northern Thailand are rugged, with steep slopes and soils that are exposed to an erosive monsoonal climate for seven months of the year. When the forests are cleared for agriculture, these lands are highly prone to accelerated erosion.

Previously, the more traditional practice of shifting cultivation had not been identified as a significant cause of accelerated soil erosion (e.g. Lal 1975; Turkelboom et al. 1997). However, with increasing hill-tribe populations, this traditional system of cultivation has become more intense, with the length of the cultivation period increasing and the period of regeneration becoming shorter. Hussain and Doane (1995) noted that, for northeastern Thailand, the period of fallow (that is, the recovery of the land) had fallen from 10–15 years to only 3–4 years. This places a much greater pressure on soil resources and increases the risk of erosion.

What are the impacts of erosion?

On-site impacts. Loss of topsoil not only reduces the depth of soil but also its capacity to hold water and the amount of nutrients it contains. This can lead to a reduction in crop productivity. Other on-site impacts include damage to embankments, earth walls, roads, trails and fences.

Off-site impacts. These include increased sediment, nutrient and pollutant loads in rivers and streams, which degrade the quality of household water supplies downstream and reduce ecological health. Siltation of dams and irrigation channels reduces their capacity. The sediment also deposits in estuaries, smothering aquatic plants and other food supplies for fish.

Land use impacts upon erosion

The rate of erosion on a hillside depends strongly on the land use. Several projects have attempted to quantify this relationship for Thailand (see Table 8.1, for example). These have included projects run by government agencies in Thailand (e.g. the Land Development Department), as well as international agencies. The Australian Centre for International Agricultural Research (ACIAR) funded two collaborative projects with field sites in Malaysia, Thailand, the Philippines and Australia (Coughlan and Rose 1997). Table 8.1 shows the measured average annual soil losses for the sites in Thailand, indicating the effect of different cultivation practices on erosion. The complexity of the erosion process is demonstrated by a comparison of these two plot sites, which have different soil types, rainfall and slopes.

Sumrit et al. (1993) classified soil erosion in Thailand into five categories (Table 8.2) and summarised the land uses observed in each of the categories. While most of the country fits into the slight and very slight categories, a large area of the country still has considerable erosion problems. Most of this land is associated with field crops and horticultural practices. Although the authors do not split the land into lowlands and highlands, some inferences can be made from looking at the land uses. Paddy, situated on lowlands, is usually associated with low levels of erosion, while shifting cultivation generally occurs on the steeper slopes. Table 8.2 provides an indication of the land uses that pose a greater risk of erosion. However, this needs to be related to position within the landscape to be of real use for identifying erosion 'hotspots'.

With increasing land use pressure and rising populations in the highlands of Thailand, an expansion of road networks is to be expected (Ziegler and Giambelluca 1997). Hence, erosion generated from roads is likely to be of increasing importance in its contribution to the total eroded sediment leaving a catchment. There

Table 8.1 Average annual soil loss and sediment concentrations from ACIAR plots at Khon Kaen and Nan, Thailand (uses data over a three-year period for Khon Kaen and one year for Nan). Source: Coughlan and Rose (1997)

Site	Treatments	Average annual soil loss (t/ha)
Khon Kaen, loamy sand, 4% slope, average annual rainfall = 913 mm	Bare plot	48
	Cultivation up and downslope	2.8
	Cultivation across slope	1.0
Nan, clay, average slope ≈ 30%, annual average rainfall = 1886 mm	Bare plot	7.2
	Clean cultivation farmers practice	0.6
	Tephrosia hedgerows	0.4
	Natural vegetation	Trace

Table 8.2 Soil erosion in Thailand. The proportional area of each erosion category is indicated in parentheses. Source: Sumrit et al. (1993)

Categories	Soil loss (t/ha/year)	Area (ha)	Land use
Very slight	0.06–0.63	18,995,500 (0.37)	Forest, paddy
Slight	6.3–31.3	14,444,200 (0.28)	Forest, rubber, orchards, paddy
Moderate	31.3–125.1	4,146,000 (0.08)	Rubber, orchards, field crops, forest + field crops
Severe	125.1–625.1	6,819,300 (0.13)	Rubber, orchards, field crops, forest + field crops, shifting cultivation
Very severe	625.1–6042	6,265,100 (0.12)	Field crops, forest + shifting cultivation field crops
Others	––	729,900 (0.01)	Coastal area, mangrove forest, shrimp farms etc.

has been little research into the extent of road erosion in Thailand, but those studies that have considered it have indicated that it has the potential to contribute significantly to the total sediment budget of a catchment.

Soil conservation, or erosion mitigation, can be achieved by reducing the run-off rate, either by engineering structures (e.g. ditches, terraces) or by using strips of vegetation that capture water and eroded sediment (e.g. alley cropping). Any attempt to predict the rate of erosion from a particular land use needs to account for these different management practices (see Table 8.3).

Introduction to erosion models

Erosion models can be used to estimate soil loss from agricultural catchments. This can assist with soil-conservation planning, land use planning, soil-erosion inventories, and regulation. Erosion models are a necessary component of an integrated water resource management approach. Given the constraints that are commonly encountered with large-scale field measurements (e.g. money, time and resources), erosion models can provide a viable alternative for assessing erosion risks across an entire catchment or region, as well as considering likely changes in erosion as a response to land use or management changes.

The demand for erosion-assessment tools has led to the development of a wide range of models, some of which are summarised in Table 8.4. These models vary, among other things, in the erosion processes considered, and the level of detail included. Some models are based on an empirical approach using statistically fitted functions. Others use a more mechanistic approach where the physical processes of erosion are described by mathematical equations.

Most models focus on one erosion process such as overland flow (sheet and rill), gully or in-stream erosion. Rarely does a model have the capacity to deal with two or more of these erosion types. For example, the USLE (Wischmeier and Smith 1978) and WEPP (Laflen et al. 1991) models have been designed to study erosion in situations of overland flow only.

Table 8.3 Erosion rate under different erosion control measures. Source: Ongprasert and Turkelboom (1995)

Cropping packages	Median erosion rates (t/ha/year)	Median run-off rates (% of annual rain)	No. of data points
'Traditional package'	60	11	91
Alley cropping with grass strips	0.4	2	128
Alley cropping with nitrogen-fixing trees in hedgerows	4.4	2	71
Hillside ditches	13	10	12
Bench terraces	01	1	35

Table 8.4 Some erosion and sediment-transport models (classifications are based upon the main processes modelled)

Model	Type	Scale	Output	Event/ non event	Spatially distributed	Comments
CREAMS	Physical	Field size 40–400 ha	Erosion, deposition, transport (slope to 2nd order channels)	Event	No	Catchment assumed to be uniform in soils, topography and land use
EPIC	Conceptual	field size, less than 250 acres (usually approx. 1 ha)	Nutrients, sediments, run-off, pesticides, plant growth	Event	No	Weather, soils and management considered homogeneous; considers N, P, pesticides, sediment; USLE based (i.e. sheet erosion)
EUROSEM/LISEM	Physical	Catchment	Run-off, sediment yield	Event	Yes	Does not model erosion in rills and gullies
GUEST	Physical	Field	Soil-loss predictions	Event	–	Sheet and rill erosion
PERFECT	Physical	Field	Run-off, erosion, crop yield	Event	–	Incorporates a crop-growth simulation module
Reid and Dunne (1984)	Empirical	Road segment	Sediment concentration	Non-event	No	Relates concentration to discharge, road segment length, gradient, and road type
USLE/RUSLE/ MUSLE/ USLE-M	Empirical/ conceptual	Hillslope	Average annual soil loss due to rainfall	Non-event[a]	No[b]	Many modifications of the original model (MUSLE, USLE-M). Model has also been revised to include new information (RUSLE). This revised USLE has been implemented in Australia in the SOILOSS model. Does not model gully or in-stream erosion.

a modified versions may be event based

b can model spatial variation when considered in grid

Table 8.4 (continued) Some erosion and sediment-transport models (classifications are based upon the main processes modelled)

Model	Type	Scale	Output	Event/ non event	Spatially distributed	Comments
WEPP hillslope model	Physical	Hillslope	Run-off, sediment characteristics, form of sediment loss	Both	Yes	Does not account for gully erosion or mass movement
WEPP watershed model	Physical	Hillslope	Run-off, sediment characteristics, form of sediment loss	Both	Yes	The watershed model comprises hillslope model with channel erosion component, impoundment component, and irrigation component.

Although the importance of road (and trail) erosion in terms of contribution to total sediment yield has been acknowledged (e.g. Douglas et al. 1993; Wallin and Harden 1996), there is relatively little literature about the prediction and simulation of road erosion either on its own, or incorporated into catchment-scale models. One exception is the extension of the WEPP model to predict road erosion. Also, the KINEROS2 model has been applied to unpaved mountain roads in northern Thailand to simulate total discharge, sediment transport and sediment concentration on small-scale road plots (Ziegler et al. 2001).

Identifying the most appropriate model for a particular study requires consideration of catchment characteristics, data availability, model assumptions and the desired outputs of the model, including the scale at which model outputs are required.

The universal soil loss equation erosion model

One of the most widely used models for predicting soil loss in agricultural regions is the USLE, which was developed by the United States Department of Agriculture. Annual soil loss (A) is calculated in tonnes per hectare:

$$A = R.K.LS.C.P \qquad (8.1)$$

where R is rainfall erosivity, K is soil erodibility, LS is the topographic factor, C is the cropping factor and P is a management-practice factor.

Although the USLE has a number of limitations, it is easy to use and, unlike more complex models, does not require large amounts of field data. The model's main strength is that it can be used to develop indicators of potential erosion across catchments in relation to rainfall and land-cover scenarios. The USLE approach is widely used by the Land Development Department (LDD) in Thailand for land use planning.

A description of each term used in the USLE is provided below.

Rainfall erosivity (*R*). The impact of raindrops on the land surface loosens soil particles and makes them susceptible to erosion. As rainfall intensity increases, the impact of raindrops increases, leading to a greater displacement of soil particles. Heavy rainfall also leads to overland flow of water, and this can lead to sheet, rill and gully erosion. As rainfall intensity and duration increase, the rates of erosion from overland flow also increase.

Rainfall erosivity in the humid tropics is calculated using the equation developed by El-Swaify et al. (1987):

$$R = 38.5 + 0.35(p) \quad (8.2)$$

where p is annual precipitation (in mm). R is in units of tonnes per hectare per year. This equation is more suitable for tropical climates than the EI_{30} index of Wischmeier and Smith (1978) and has been successfully applied in Thailand.

Soil erodibility (*K*). Some soils are naturally more prone to soil erosion due to their physical and chemical structure. Erodibility is dependent on soil texture, organic matter content and permeability.

The LDD in Thailand provided values (Table 8.5) of the soil erodibility factor, K, for each of the land units mapped within the catchment of the Mae Chaem in northern Thailand. Within each land unit, the soil erodibility was assumed to be homogeneous.

Slope factors (*LS*). The slope of the land has a major effect on the rates of soil erosion. As slope increases, the velocity (and hence energy) of overland flow increases, thus increasing the shear stresses applied to soil particles on the surface. As slope length increases, the volume of overland flow and its velocity also steadily increase, leading to greater erosive forces applied to the soil surface.

Slope in the Mae Chaem catchment ranges from 0° to 78°. For slopes less than or equal to 8%, the topographic factor (LS) is calculated using the Wischmeier and Smith (1978) equation:

Table 8.5 *K* (soil erodibility) factors for the Mae Chaem catchment. Source: provided by the Land Development Department, Thailand, May 2000

Land unit(s)	Soil texture	K factor
6, 8, 10	Loam + gravel	0.25
12	Loam	0.25
23, 25	Loam	0.27
27	Loam	0.27
45, 47, 49	Clay	0.24
46, 48, 50, 55, 35, 37	Clay + gravel	0.22
88, 99	Clay	0.17

$$LS = [(length(m))/22.13)^{0.5}] \times (0.065 + 0.0456(slope)) + 0.0065(slope)^2 \quad (8.3)$$

and for slopes greater than 8% the Hellden (1987) equation is used:

$$LS = (0.799 + 0.0101(length(m))) \times (0.344 + 0.0798(slope)) \quad (8.4)$$

where slope length is defined as the distance from the point of origin of overland flow to the point where either the slope gradient falls enough that deposition begins, or run-off water enters a well-defined channel.

Cropping factor (*C*). The vegetation cover, or type of crop planted, plays a critical role in determining the rate of erosion. The leaves of plants protect the soil from raindrop impact, and the roots hold the soil together. Plants also tend to increase infiltration of water, thus reducing the volume of overland flow running down the slope.

Crop-management factors (*C*) have been provided by the LDD in Thailand for a large number of individual crops in addition to mixed-farming systems. Table 8.6 shows the crop-management factors for selected crops in the Mae Chaem catchment. The value for *C* was set to 0.001 for bunded plots (Saifuk, pers. comm.).

Management-practice factor (*P*). A number of land-management practices have been developed that can significantly lower the rates of soil erosion. This is generally achieved by reducing the run-off rate, either by engineering structures (e.g. ditches, terraces, contour banks), or by using strips of vegetation that capture water and eroded sediment (e.g. strip cropping).

Values of *P* were provided by the LDD for a number of management practices on different slope classes (Table 8.7). The value of the *P* factor has been set to 0.1 for bunded plots (Saifuk, pers. comm.).

Applying the universal soil loss equation to northern Thailand

In this project, only sheet erosion from agricultural fields and forested areas was modelled, using the USLE-based approach modified to suit conditions typical of northern Thailand highlands. Anecdotal evidence and personal field surveying in the case-study sub-catchments of the Mae Chaem suggested that gully erosion was not a major source of sediment in this region. Erosion along roads and trails can also contribute to sediment loads, but a lack

Table 8.6 *C* (crop-management factors) for the Mae Chaem catchment. Source: provided by the Land Development Department, Thailand, November 1999

Crop	Crop-management factor (C)
Paddy rice	0.28
Upland rice	0.7
Soybean	0.421
Groundnut (peanut)	0.406
Maize (grain)	0.28
Maize (forage)	0.1
Cabbage	0.6
Potato	0.6
Onion	0.34
Temperate fruit trees	0.3
Tropical fruit trees	0.15
Fallow	0.09
Forest	0.001

Table 8.7 *P* (management-practices factors) for Thailand. Source: provided by the Land Development Department, Thailand, November 2000

Slope (%)	*P* factor				
	None	Contour cultivation	Strip cropping around contours	'Arable' land terrace	Bench terrace
0–2	1.0	0.6	0.3	0.4	0.12
2–7	1.0	0.5	0.25	0.5	0.1
7–12	1.0	0.6	0.3	0.6	0.12
12–18	1.0	0.8	0.4	0.8	0.16
18–24	1.0	0.9	0.45	0.9	0.18
24–100	1.0	1.0	0.5	1.0	0.19

of field data prevented inclusion of this type of erosion. However, it would be relatively easy to incorporate an additional component capable of predicting sediment sources from roads if sufficient data were collected in the future.

As a departure from the standard, annualised application of USLE, in northern Thailand it was applied separately for both the wet (April–November) and dry (December–March) seasons. This was done to allow for the running of scenarios affecting cropping patterns during the wet and/or dry season.

Over large scales, the area to which the model is applied is broken into segments in which the USLE factors are assumed to be uniform. For the case studies in northern Thailand, the USLE was applied to each land-unit type within the catchment.

Results for northern Thailand case study

Figure 8.1 shows plots of erosion rates predicted by the erosion model for a range of crops and land covers for the wet season of 1990 for land unit 88 and Figure 8.2 for land units 47 and 49 in the Mae Chaem catchment. The predicted erosion rates for most crops and management types on land units 88 and 99 are within the LDD-prescribed threshold of 31.25 t/ha. In comparison, for upland fields in land units 47 and 49, most crops are prone to extreme rates of erosion, with only maize, fallow and forest types yielding less than the LDD-prescribed threshold. In practice, policy designates land unit 49 for forest cover only. No differences in erosion rates are distinguished between land units 47 and 49, despite land unit 49 being generally much steeper, as the land units fall into the same slope category for defining the *P* factors for the USLE. In reality, it would be expected that considerably more erosion would occur on land unit 49 than on land unit 47.

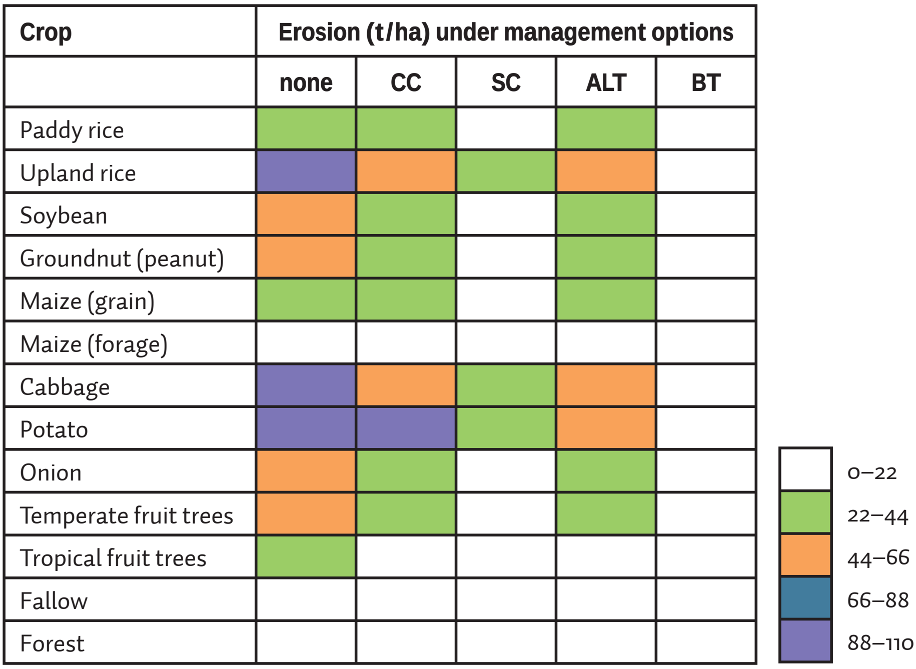

Figure 8.1 Erosion rates (t/ha) on land unit 88 under available management options for 13 crop or land-cover types on low-sloping land units suitable for paddy agriculture (BT: bench terrace, ALT: 'arable' land terrace, SC: strip cropping around contours, CC: contour cultivation)

Integrating the erosion model with the Integrated Water Resources and Management project decision support system

The erosion model, based on USLE, was easy to implement and could be readily integrated into the DSS framework. It was also widely used by staff of Thai Government agencies, so increasing the likelihood of the DSS being adopted by these agencies. Since the erosion model is calculating only local soil loss (and not downstream sediment movement), the integration of the erosion model with the DSS involved only the crop and land-management options being passed from the land use decision tool. Interaction with the hydrology model would be needed only if water quality impacts on downstream users were being considered, as the flow volume would determine the capacity of the channel to transport suspended sediment.

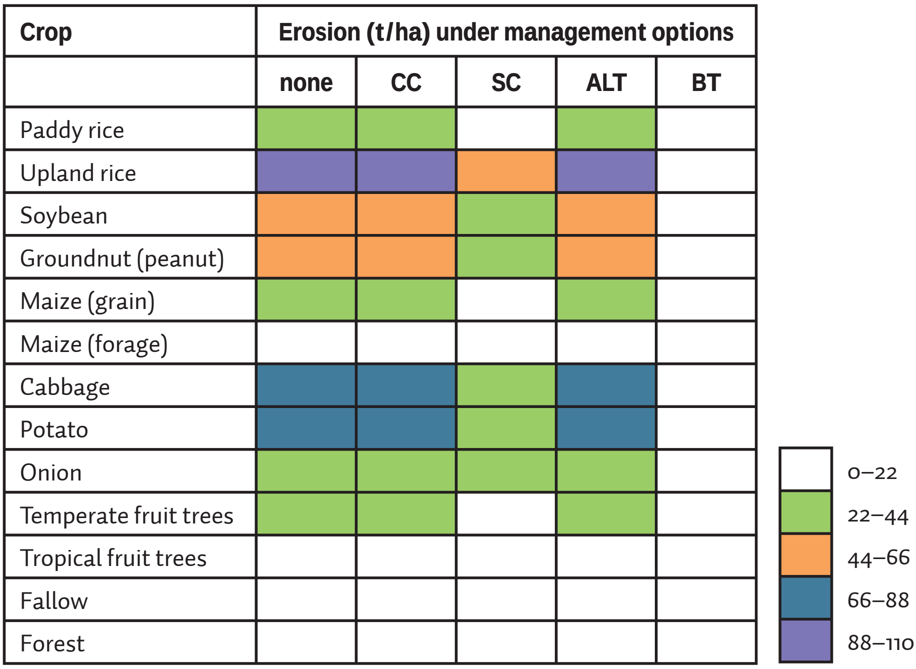

Figure 8.2 Erosion rates (t/ha) on land units 47 and 49 under available management options for 13 crop or land-cover types on low-sloping land units suitable for paddy agriculture (BT: bench terrace, ALT: 'arable' land terrace, SC: strip cropping around contours, CC: contour cultivation)

The USLE can be used to provide spatial estimates of annual erosion and is of low complexity. Another major advantage of the technique is that explicit consideration is given to crop type and management practices (within the *C* and *P* factors)—a requirement for scenarios of land and water management. The USLE has been used over a range of scales from small plots, from which the original equations were developed, to large-scale projects to determine soil erosion hazard within a catchment.

Although the USLE has a number of limitations, the paucity of data for the sub-catchments used in this study made it inappropriate to use more data-intensive erosion models. The model's main strength was that it could be used to develop indicators of potential erosion across entire catchments in relation to rainfall and land-cover scenarios. The USLE approach is also widely used by the LDD in Thailand for land use planning. The LDD was a primary target user for the IWRAM DSS as a whole, so using the USLE approach increased the likelihood that the IWRAM DSS would be adopted by LDD for investigating the impacts of management options on catchment-scale erosion.

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