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Assessing the Ecosystem Services for Water Supply: Case Study of the Forested Water Intake Areas of the City of Vienna

Master's Thesis (Double Degree Progamme)
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Abstract

This thesis has the objective to evaluate the ecosystem services of forests in the intake area of the Viennese water works. Especially the benefits from water storage, erosion reduction and water purification of forest stands are analyzed. Vienna's water supply is provided by spring water which is extracted from a forested water protection area situated in the North-Eastern limestone Alps. This evaluation of alternative land use options and their impacts on water supply should render the conversion of the value of clean water in monetary units possible using suitable economic valuation techniques. Therefore, the effects of water protection forestry versus those resulting out of a strategy of forest management seeking to maximise profits are investigated. The hydrological processes in a forested catchment are simulated using the *Water Erosion Prediction Project* resulting in a clear sign of increases of sediment yield due to intensive forest management. This analysis forms the foundation for the economic assessment comparing three different alternatives to cope with an increase of turbidity after intensive forestry. Thus, the assessment of the benefits is based on an *alternative cost approach* in which the losses of intensive forest management are compared with increased purification costs of technical systems. As a result, the costs arising due to filtration of turbid water, a combination of water provided by the water work Moosbrunn with filtration and the diversion of water of decreased quality are displayed. The replacement cost for the ecosystem service of clean water provision arrives at a value of 265 €.ha^{-1}.y^{-1}. Additionally, continuous cover forestry shows the highest cost-effectiveness of 0.082 €.m^{-3} compared to 0.195 €.m^{-3} of the combined approach and 0.251 €.m^{-3} of pure filtration. In conclusion, these results only represent a frame of the value of the ecosystem services in focus, even though they are based on a broad methodological foundation.

Kurzfassung

Das Ziel der vorliegenden Masterarbeit liegt in der monetären Bewertung der hydrologischen Ökosystemdienstleistungen des Einzugsgebiets der Wiener Hochquellenleitungen. Dabei liegt das Hauptaugenmerk auf der Analyse der positiven Auswirkungen des Waldes auf Wasserspeicherung, Erosionsrate und Filtration des Wassers. Das Wiener Wasser entstammt dem bewaldeten Wasserschutzgebiet der Wiener Wasserwerke in den Nordöstlichen Kalkalpen. Der Vergleich der Auswirkungen einer intensiven auf größtmögliche Gewinne ausgerichteten Forstwirtschaft mit jenen einer Dauerwaldwirtschaft auf Abfluss und Sedimentfracht stellt die Grundlage für eine ökonomische Bewertung dieser Dienstleistungen dar. Für die Modellierung der vorherrschenden hydrologischen Prozesse im Einzugsgebiet wird hierfür das *Water Erosion Prediction Project* herangezogen. Basierend auf den Ergebnissen der Modellierung, welche eine Erhöhung der Sedimentfracht begründet durch intensive Forstwirtschaft belegt, werden durch die Alternativkostenmethode die Kosten einer schonenden Waldbewirtschaftung den anfallenden technischen Aufbereitungskosten im Falle erhöhter Trübungswerte gegenübergestellt. Hierfür kommen einerseits Filtrierung des Wassers oder Filtrierung kombiniert mit der Versorgung durch das Wasserwerk Moosbrunn sowie andererseits eine Ausleitung kontaminierten Wassers in Frage. Aufbauend auf den Opportunitätskosten einer Dauerwaldwirtschaft können so die Alternativkosten durch einen Wert von 265 $€.ha^{-1}.J^{-1}$ geschätzt werden. Darüber hinaus weist die rein auf eine Maximierung des Wasserschutzes ausgerichtete Forstwirtschaft die höchste Kosteneffektivität mit einem Wert von 0.082 $€.m^{-3}$ auf. Ein kombinierter Ansatz, welcher eine Bereitstellung des Trinkwassers vom Wasserwerk Moosbrunn mit einer Filtration kontaminierten Wassers ergänzt erreicht eine Kosteneffektivität von 0.195 $€.m^{-3}$, was einer höheren Effektivität als jener der reinen Filtration mit 0.251 $€.m^{-3}$ entspricht. Diese Werte basieren grundsätzlich auf einem methodisch breit angelegten Fundament und bieten somit einen wissenschaftlich fundierten Richtwert. Trotzdem dürfen die generierten weder verallgemeinert noch als Absolutwerte als Kalkulationsbasis herangezogen werden und stellen daher einen Werterahmen für die Integration der Wertigkeit der Dienstleistungen, welche vom Ökosystem bereitgestellt werden in die Entscheidungsfindung auf operationaler Ebene dar.

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1 Introduction

This thesis aims at the estimation of the benefits from landscape protection and “soft forestry” in the intake area of the Viennese water works with respect to water supply. An evaluation of alternative land use options and their impacts on water supply renders the conversion of the value of clean water in monetary units possible using suitable economic valuation techniques. A fundamental understanding of the forest hydrological processes represents the foundation for the monetary valuation of this ecosystem service. Therefore the estimation of potential impacts on water supply caused by a change in land use is based on the modelling application of the Water Erosion Prediction Project.
In addition, the integration of the value of the hydrological services provided by the ecosystem into decision making represents one of the main goals of this analysis. The basic understanding of the ecosystem processes which are directly correlated to land use management is the basis for an appropriate monetary valuation of the services provided.

Like many of the biggest cities of the world including New York, Los Angeles or Tokyo, Vienna is provided with high quality drinking water by spring water being extracted from a forested catchment which is also subjected to strict landscape production. The catchment of the Viennese water is situated at the border of Styria and Lower Austria in the mountainous region of Hochschwab and Rax/Schneeberg/Schneealpe, both being part of the North-Eastern limestone Alps. 97.5% of the Viennese drinking water is extracted from these catchments where the major part is protected by water protection areas. Best practice forestry is applied in this protection zone in order to secure a constant high quality of water (Köck, 2008).

The Viennese water works provide drinking water of high quality extracted from the catchments of the spring water mains. The hydrological ecosystem services provided by the water protection forest include the regulation of flows due to water storage and positive effects on water quality including a decrease of sediment yield and other pollutants resulting out of buffering and filtering processes. Despite the fact, that this whole ecosystem provides a large quantity of interlinked ecosystem services such as air quality, carbon sequestration or soil generation, this work focuses exclusively on the hydrologic services of the region. This approach assists the assessment of the impacts accruing due to alternative land use actions on water supply concentrating on the economic consequences of an increase in turbidity due to a change in the strategy of forest management.

Forestry represents the only possible form of land use in this region with a wide range of management practices. This analysis evaluates the impacts of the following set of alternatives:

- Continuous cover forestry aiming at a maximum level of water protection
- Clear-cut forest management seeking to maximise forest revenues

The economic assessment includes on the one hand the *replacement cost technique* based on the opportunity costs of water protection forestry versus a *cost-effectiveness-analysis* providing information about the costs of various alternatives. As a result, this analysis compares the opportunity costs of continuous cover forest management to the costs occuring due to other alternatives. These alternatives are applied in order to react to a deterioration of water quality after intensive forestry. The monetary analysis of three alternatives includes:

1. filtration of water,
2. a combination strategy extracting drinking water from the water work of Moosbrunn complemented by filtration of the residual amount water of decreased quality
3. diversion of turbid water.

The result is an economic estimate of the value of the water protection forest for the region. As it derives a common metric for making comparisons, this guideline value of the worth of the ecosystem services in focus thus can provide a rational and objective basis for the decision making of the Viennese water works.

This thesis is divided into various main parts including state of the art in forest hydrology, the methodological setup and its application for the case study.

In chapter 1 a general introduction into the assessment of hydrologic ecosystem services for Vienna is presented. Chapter 2 describes the basic theoretical foundation concerning state of the art in ecosystem valuation. Secondly, chapter 3 provides the methodological background which is represented by a basic theoretical foundation concerning forest hydrology in section 3.1 and the economic assessment in section 3.2. The forest hydrological chapter consists of two parts, where 3.1.1 deals with state of the art in forest hydrology and 3.1.2 describes suitable approaches for the modelling of the hydrologic processes occurring in forested catchments. The theoretical background provided by chapter 2 and 3 is then applied for the case study of Vienna in chapter 4. Starting with a description of the headwaters of the city of Vienna and suitable land use alternatives in the area (section 4.1 and 4.2), the section 4.3 provides the modelling of the hydrological processes in the catchment area of Vienna. Subsequently, the economic assessment of the ecosystem services of drinking water for the case study area is covered in section 4.4 including the assessment of forest revenues and 4.5 dealing with the application of the replacement cost technique and the cost-effectiveness analysis. Finally, section 5 presents the discussion and conclusions of this evaluation.

2 State of the Art in Ecosystem Valuation

The following chapter presents an overview of the theoretical background of the state of the art in ecosystem valuation. Therefore it provides basic definitions of an ecosystem, ecosystem services and the valuation of ecosystem services. Beyond that, the two concepts of economic theory of value and the basic steps of ecosystem valuation will be discussed.

Ecosystem services are characterised as the benefits people obtain from ecosystems. The current trend of intensifying human impacts on natural capital has resulted in a decline of two-thirds of the world's ecosystem services (MA, 2003). Due to this fact, there is a strong urge to foster more sustainable land use management based on the fundamental understanding of ecosystem processes and the value ecosystem services offer to people. The awareness of humans being beneficiaries of the services nature provides to humankind represents the essential precondition for appropriate management decisions for ecosystems.

The Millennium Ecosystem Assessment (MA) represented a focal point of this field of study, as it published the first comprehensive global assessment of the status of the world's ecosystem services in 2003. This assessment was planned to assist stakeholders such as the business community, the health sector and nongovernmental organisations in management decisions (MA, 2003). The MA published a series of reports assessing the present state as well as future trends of the world's major ecosystems and the services provided by them. 1300 international experts participated in this cooperation in order to find available options for restoration, conservation and enhancement of sustainable use of ecosystem goods and services. The alerting statement that 60% of the ecosystem services which are presently being degraded or used unsustainably can be seen as major finding of these studies (Haines-Young and Potschin, 2010).

In fact, a long time before the MA started, the benefits humans gain of the world's ecosystems were in focus of attention in the 1970s, when the term "environmental services" was used for the first time by the Study of Critical Environmental Problems (SCEP, 1970). As a result, the ecosystem services approach is now integrated into strategic planning all over the world.

Nevertheless, the implementation of studies of ecosystem services moving from a conceptual to an operational framework still requires intensive research in the future. Due to the neglect of ecosystem services on the world's commodity markets, the integration of the value of ecosystem services into daily decision making is rendered difficult.

Ecosystem services significantly contribute to human welfare representing a high portion of global total economic value. Nevertheless, they are not adequately quantified in economic terms hampering the comparison of the ecosystem's services with manufactured capital. An estimate of total economic value of ecosystem services needs to be infinite owing to the fact that the world's economies would collapse in absence of these life supporting systems (Constanza, 1997).
The following chapters provide a description of the state of the art of valuation of ecosystem services. It includes important definitions of ecosystem, ecosystem services and values. In addition, the Ecosystem Services Approach based on the MA and state of the art valuation methods for ecosystem services are described.

2.1 Definition of Ecosystems

The following chapter will give basic insight into the development of the concept of ecosystems and their definition.

The ecosystem concept can be seen as the foundation for understanding the basic principles of life on earth representing an emerging field of research and management (MA, 2003). According to Tansley (1935, p. 299), an ecosystem is 'the whole complex of physical factors forming what we call the environment'. Beyond that definition, the MA (2003) defines the ecosystem approach as a strategically developed management of land, water and living resources. This integrated approach should aim at the conservation and sustainable use of natural resources, of which each component needs to be seen as being equally important for human beings. It implies the use of scientific methodologies including the essential structure, processes and functions of ecosystems. Therefore, also human beings have to be regarded as an integral part of the ecosystems of the earth (MA, 2003).

2.2 Definition of Ecosystem Services

According to Constanza et al. (1997) *ecosystem functions* represent the habitat, biological or system properties or processes of ecosystems. *Ecosystem goods* (e.g. timber) and *services* (e.g. water filtration) stand for direct and indirect contributions to human well-being derived from ecosystem functions. Daily (1997) considers *ecosystem services* as the conditions and processes being crucial for the conservation and fulfilment of human life. Another definition provided by the Ecosystem Services Project (2001) speaks of *ecosystem services* as the flow from natural assets, which are capable of the provision of ecological, financial and cultural benefits to humans. In addition, the MA (2003, p. 53) conceives *ecosystem services* as ‘the benefits people obtain from ecosystems’, which represents the most common characterisation. Figure 1 shows the linkages of ecosystem services and components of human well-being according to the MA. Furthermore, it presents reference levels showing to which extent socio-economic factors are able to compensate these linkages and the intensity of the relationship between ecosystem services and human well-being (MA 2005).

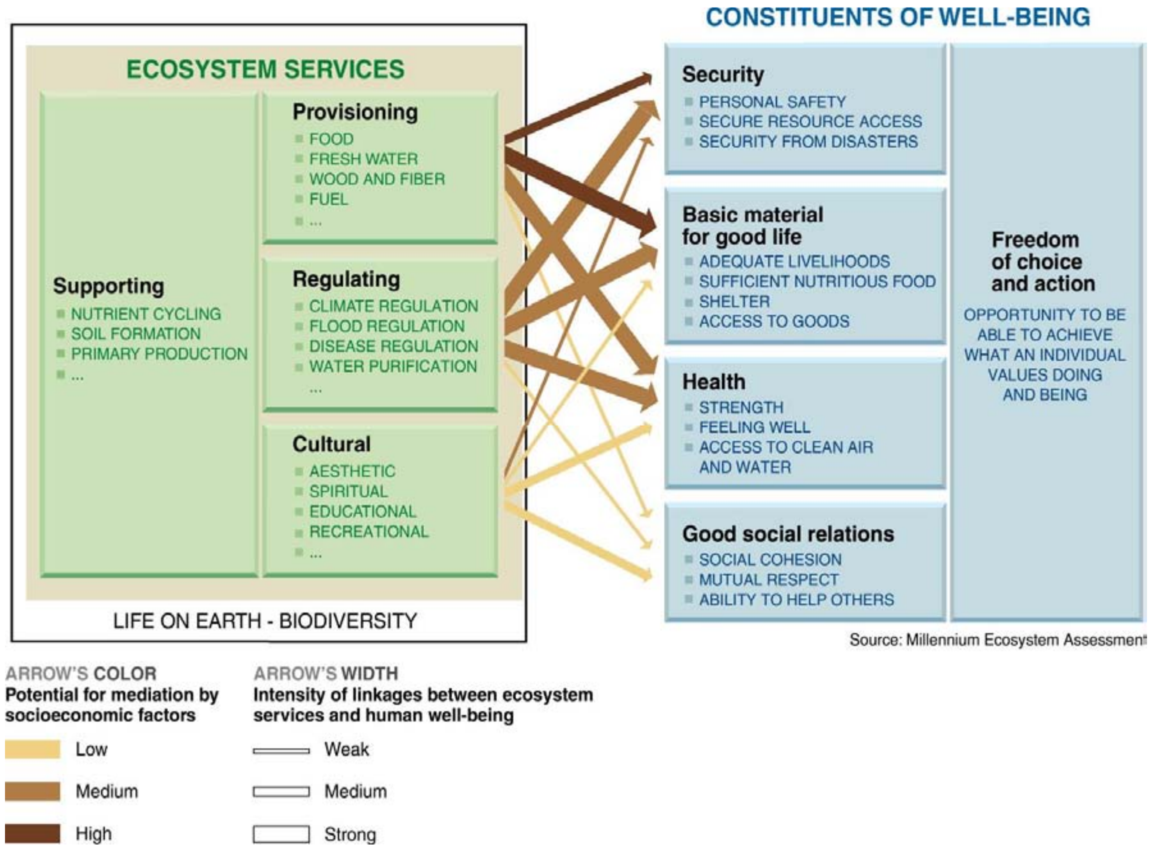


Figure 1 Links between ecosystem services and human well-being (MA 2005)

Ecosystem services contribute to human welfare in many diverse ways. Due to this high diversity, benefits have been categorized in a number of different approaches (Groot et al. (2002) and Pushpam et al. (2010); Norberg (1999); Constanza (2008); Moberg and Folke (1999)). This thesis uses the categorization of the Millennium Ecosystem Assessment which represents a *functional classification* approach with the four categories of provisioning, regulating, supporting and cultural services. Figure 2 presents an overview of this classification scheme.

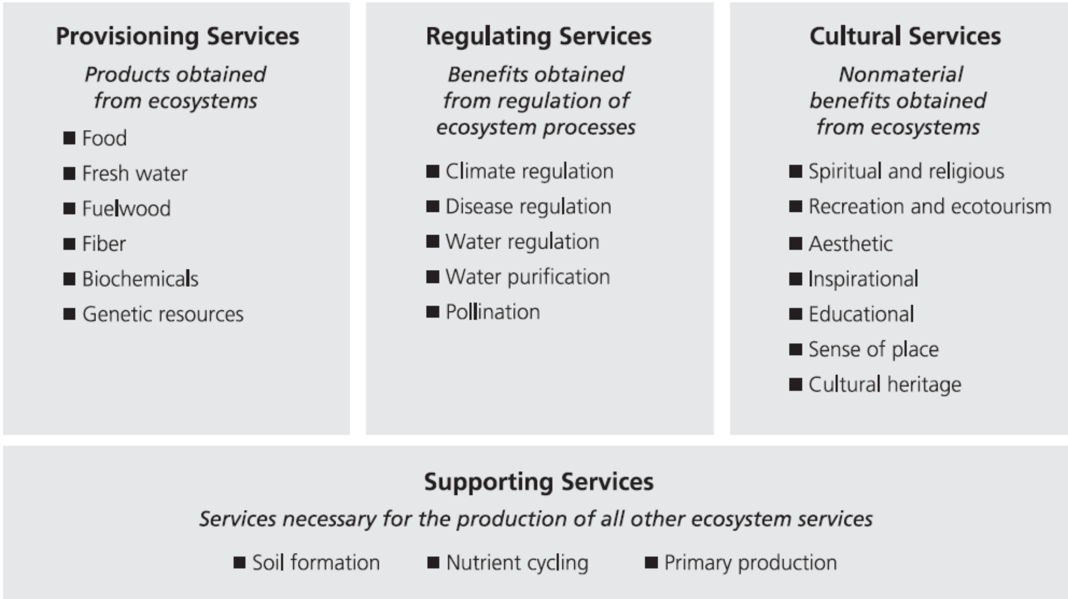


Figure 2 Classification of ecosystem services according to the MA (2003)

Provisioning services are products being directly obtained by ecosystems such as fresh water. The example of fresh water illustrates the interlinked properties of the two relevant categories of provisioning and regulating services (MA, 2003).

Regulating services are the beneficial effects arising out of the regulation of ecosystem processes. These services include e.g. water regulation with respect to timing and quantity of runoff, floodings or changes in aquifer recharge (MA, 2003).
Water supply is one of the regulating services and includes filtering, retention and storage of fresh water. It represents the provision of water for the consumptive use for drinking, irrigation and industrial purposes. The ecosystem function of water supply strongly depends on the implications of the ecosystem in the hydrologic cycle (de Groot et al., 2002).

Cultural services represent non-materialistic enrichment for humans such as spiritual experience, recreation and enrichment in an aesthetic way. Due to the subjective perception of individuals, the valuation of cultural services strongly differs among different social, economic or political groups (MA, 2003).

Supporting services are the foundation for the provision of all other ecosystem services. Impacts of these services on people are revealed rather indirect or over long periods. In contrast to that, changes in provisioning, regulating or cultural services show direct impacts occurring on a short time horizon (MA, 2003).

Boyd and Banzhaf (2006) define ecosystem services as ‘components of nature, directly enjoyed, consumed or used to yield human well-being’. This definition mainly corresponds to that of the Millennium Ecosystem Assessment.

From the standpoint of welfare measurement, the *direct use* as essential part of the definition implies that ecosystem services are seen as end-products of nature. In welfare accounting, the distinction of intermediate from final goods prevents double counting of their value. Boyd and Banzhaf (2006) regard ecosystem services as contribution to the production of the *final good* or *services*. Thus, ecosystem processes and functions turn out to be no end-products, as they are biological, chemical and physical interactions between components of the ecosystem. Owing to that definition, ecosystem services are defined through the final good or service consumed by households. This definition can be consistently used in combination with conventional income accounting.

This thesis applies the well known categories of ecosystem services developed by the Millennium Ecosystem Assessment due to the fact, that this classification is most widely used. Moreover, the application of the categorisation provided by the MA is reasonable, as this thesis solely focuses on the ecosystem service of water supply as regulating service. If it was an assessment of the value of the entire ecosystem, there would be a risk of double counting of various ecosystem services in the usage of the MA classification scheme, as it is using a rather generic definition of services (Fisher, 2009).

The ecosystem service of forests for water supply includes regulating, provisioning, supporting as well as cultural services related to water. Most obvious examples of water related benefits to humans include the provision of drinking water, irrigation water, hydropower, aquatic fauna, recreation opportunities and flood mitigation. These ecosystem services are dependent on basic hydrologic processes determining water retention, water yield, natural water filtration and sediment regulation, which can be listed as supporting services for themselves (Vigerstol and Aukema, 2011). Additionally, some of these services are competitive of nature. Looking at the ecosystem service of water supply, increases in timber yields lead to decreases of the water protective functions of forests. Thus, the social functions provided by forests represent economic goods as soon as they imply additional costs for producers (Pabst, 1971).

2.3 Economic Theory of Value

The term *value* represents the ‘contribution of an action or object to user-specified goals, objectives or conditions’. Moreover, *valuation* can be seen as an expression of the value of a specific action or subject (Farber et al., 2002).

In applying the term *value* in economics, it refers to the appropriate measure of an equivalent in money equally expressing a change in individual utility. The goal of economics can be characterised as an increase of human well-being (Freeman, 1994). Ecosystem services can be expressed by an economic value as soon as they contribute to the satisfaction of human needs and wants and the increase of individual’s well being. This anthropogenic focus of economic valuation does not include any value of the survival or well-being of other species. Only in case of e.g. altruistic concerns of individuals, non-use values would be valued (Freeman, 1994).

The classical economic definition of *value* goes back to the labour theory of value. Adam Smith (1723-1790) introduced the value of any commodity, not to use, but to exchange, as an equivalent to the quantity of labour which it enables an individual to purchase. Thus, labour represents the real measure of the value in exchange for all commodities according to Smith. In contrast, Jeremy Bentham (1748-1832) argued that utility is the measure of an individual’s happiness. In this context, welfare is stated as the sum of individual utility. The neoclassical view of value introduced the distinction between total and marginal utility as well as the theory of demand using measures of welfare such as consumer and producer surpluses (Schmid, 2011). Neoclassical economics define the purpose of economic activity in increasing human well-being, including that each individual knows best how well off he or she is in a given situation (Freeman et al., 1994).

In environmental economics, total economic value of the change in quantity or quality of a resource includes use and non-use values (Bockstael and Freeman, 2005). Figure 3 shows the components of total economic value.

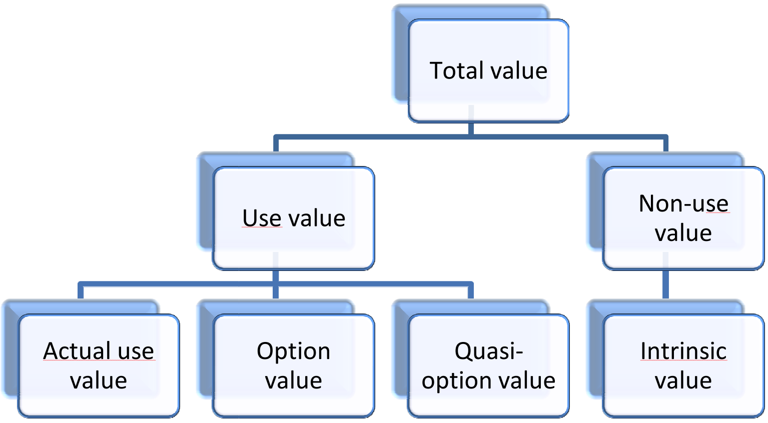


Figure 3 Components of total economic value

Use value, also called instrumental value, can be defined as direct or indirect (so called intrinsic) value, depending on how it is linked to human interests or actions (Echavarria, 2000). It is formed as a result of its contribution to some anthropogenic goal or purpose (Constanza and Folke, 1997). Option value includes the value in potential use by oneself, the value in potential use by other individuals or the value in potential use by future generations. Quasi-option value stands for the value of avoiding irreversibilities with regard to expected future findings. The intrinsic value, also called existence value, is not related to any actual or potential use (Schmid, 2011). An ecosystem service having intrinsic value means, that it is valuable for itself, as its value is not dependent on its utility (Freeman, 1994). It may be brought in context with sympathy or stewardship, which does not fit into the concept of neo-classical economics. Instead, existence value can also be seen as spiritual consumption, without being right-based (Schmid, 2011).

Natural capital is essential for human welfare. At a level of zero natural capital, the level of human welfare also arrives at zero as there is no feasible substitute. Thus, an accurate measure of total value of natural capital to human welfare is not possible. Instead, the preferable question is how changes in the quality or quantity of natural capital and ecosystem services may alter human welfare (Constanza et al., 1997).

In applying the economic value concept, ecosystem or non-human species do not have any value system, as they do not follow any conscious aims. Thus, it is not possible to define value as the amount of which a component contributes to meet a particular goal (Farber et al., 2002).

2.4 Valuation of Ecosystem Services

According to Farber et al. (2002) *ecosystem valuation* is characterised by the process of expressing a value for ecosystem services. This process represents the foundation for scientific observations and measurements.
The main goal of the valuation of ecosystem services is the assessment of impacts accruing due to differing management practices. Valuation of these impacts should provide a tool that facilitates the evaluation of trade-offs between alternative ecosystem management regimes for decision-makers. These strategies should include the impacts which each management option provides for the services of the investigated ecosystem (MA, 2003).
In general, ecosystem valuation is critically discussed, as it attempts to value intangibles such as human life or long-term ecological benefits. From another point of view, valuation of human life is done in daily life. Ecosystem valuation is a challenge to value marginal changes in natural capital. This value is represented by the differences of human welfare due to small changes in environmental services, which either alter the benefits or the costs of human activities (Constanza et al., 1997). This anthropogenic view implies that the value of a species or form of nature to individuals is entirely determined by the ability to satisfy the wants and needs of humans (Kareiva et al., 2011).

In general, it is not possible to estimate values which are placed on certain services by each individual in an accurate way. Thus, ecosystem valuation addresses deep lying problems of economics, such as revelation and aggregation of preferences and related uncertainty with this process. When dealing with market prices in ecosystem valuation, they often do not include full costs of production for the society. In absence of market prices, contingent valuation surveys provide rather unreliable results, especially when applied on topics with which the public is not familiar. Additionally, preferences revealed by individuals, may cause valuation problems as they are strongly depending on institutional contexts and the level of knowledge of the people addressed (Daily et al., 2000).

The purpose of valuation of ecosystem services is regarded as the organisation of information in order to help decision makers. Valuation methods combined with financial instruments and institutional arrangements represent tools for politics to capture the value of natural capital. In fact, due to the construction of crude lower bound estimates valuing natural water purification services, communities are able to determine whether it is preferable to preserve natural services or to construct a water treatment plant (Daily et al., 2000). Basic understanding of the biophysical processes forms the foundation needed to support such decisions. Therefore, valuation of ecosystem services has to point out appropriate methods to perform the measurement of value as well as the strengths and weaknesses of various approaches (Kareiva et al., 2011).

2.5 Steps of Ecosystem Valuation

In moving from theory to implementation, the framework presented in figure 4 shows the role that ecosystems can play in decision-making (Daily et al., 2009).

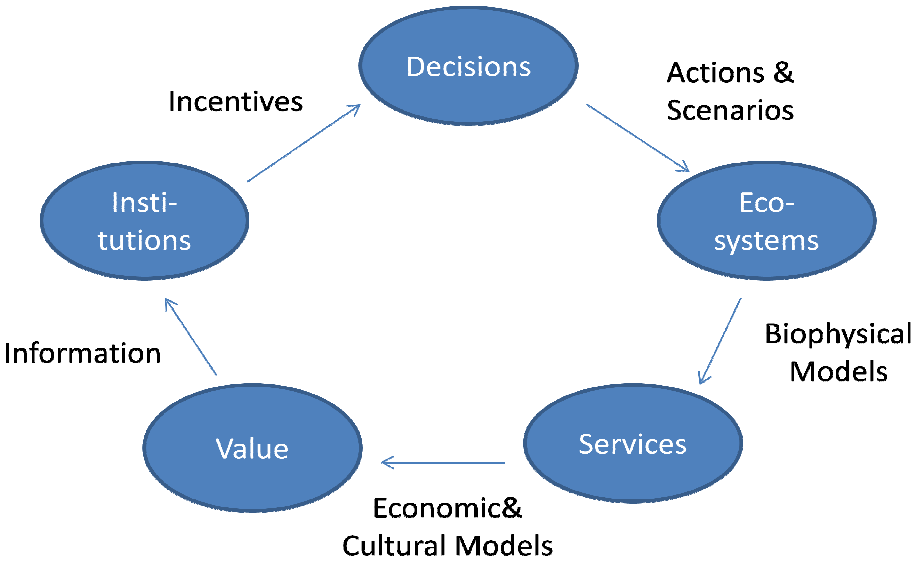


Figure 4 A framework illustrating how ecosystem services can be integrated into decision-making. All of the items could be interlinked here (adaptation of Kareiva et al., 2011)

The given framework connects science of quantifying services with valuation as well as policies to payment schemes and management actions taking account of the services provided by ecosystems. The understanding and valuing of ecosystem services represents an important tool in order to provoke an improvement of decisions dealing with natural capital. As a result, activities concerning use of land, water and other natural resources depend on these decisions.

Biophysical sciences form the basis for understanding the linkages of decisions and ecosystems. In addition, biophysical science needs to be combined with economics and social science in order to link ecosystems and services.

The next step includes social sciences fostering the understanding of the value of services to people, using economic and cultural models. Therefore, economic valuation methods are frequently applied, placing monetary values on natural capital (Kareiva et al., 2011). In this context, value may include also non-monetary dimensions such as health, livelihood

support and cultural significance, which should not be left out (Dasgupta, 2001). Thus, the inclusion of various types of benefits and people into valuation and broader decision-making approaches should be ensured (Heal, 2000).
As a result, institutions guiding resource management and policies are provided with helpful information derived from the valuation of ecosystem services. This idealised way of framing the operationalisation of ecosystem services into decisions should help to clarify the frontiers of research and implementation (Carpenter et al., 2009). This information should also be provided by this analysis.

Daily et al. (2009) propose three steps of assessment also representing the basis of this analysis:

1. Identification of alternatives, including the identification of ecosystem services of interest in combination with comprehension of its characteristics. Description of relevant scales, differing ecosystem services and social and economic aspects.
2. Modelling and Mapping Phase including an impact analysis: Understanding of relevant ecosystem processes as well as biophysical characteristics and their variations in time and space. Identification of all impacts, uncertainties and risks. This step represents the foundation for economic valuation techniques.
3. Valuation including a translation of consequences of various alternatives into comparable units.

These steps of moving from ecosystems to monetary values are illustrated in form of an environmental change scenario analysis of the water supply provided by the Viennese spring water mains in this thesis.

3 Methodology

Approaching the topic of ecosystem service valuation, there is a huge database of methodologies available for a huge variety of applications being highly discussed. This chapter should provide an overview of available techniques, facilitate the identification of advantages and disadvantages of individual methods and illustrate the selective process of the methodological approach building the foundation for this analysis.

In order to capture the complexity of ecosystem valuation accordingly, the state space approach provides a suitable technique to arrange the different pieces making up this analysis in a structured way. As illustrated in figure 5 the ecosystem in focus which is the forested catchment area of the water supply of Vienna is characterized by various inputs, its state and the output.

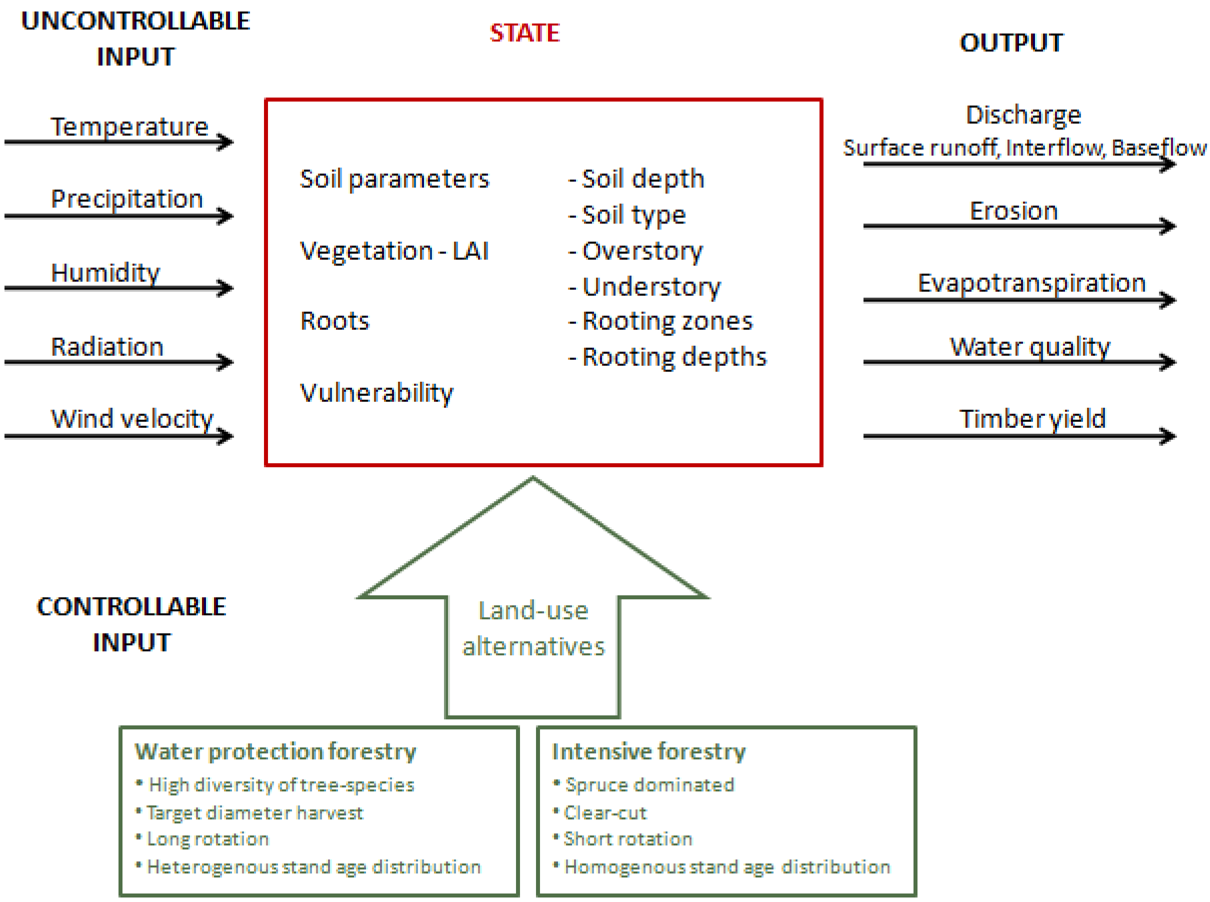


Figure 5 Representation of the state space approach illustrating the structuring mechanisms of the forested catchment area

Uncontrollable input into the system is mainly provided by climatic factors including temperature, precipitation, humidity, radiation and wind velocity. These factors have significant impacts on the state of the system being represented by soil parameters, vegetation and its vulnerability. Aside from the climatic factors, the state of the system is strongly dependent on the controllable input of land-use mainly representing anthropogenic impacts. In this case, there are two land-use alternatives having the power to change the state of the system. The first alternative is aimed at a maximum level of water protection including a high diversity of tree-species, target diameter harvest, long rotation periods and a heterogeneous stand age distribution. On the contrary, there is intensive forestry seeking to maximize profits using spruce dominated stand, a harvest strategy based on clear-cut, short rotation periods and a homogenous stand age distribution.

As a result of input and impacts of land use the state of the system and consequently its outputs may be altered. Thus, discharge, erosion, evapotranspiration, water quality as well as timber yield can be affected.
These interrelations build the foundation for understanding the processes going on in the system, the impacts various management strategies assert on the services provided by it and the importance of the ecosystem for human well-being.

3.1 Forest Hydrology

The following section discusses the impacts of forested areas on water supply (3.1.1), forest management and its impacts on forest hydrology (3.1.2) and present approaches for the modelling of hydrologic processes in forested catchments (3.1.3).

In the last decades, intensive studies in the field of forest hydrology including Bosch and Hewlett (1982), Robinson et al. (2003), Calder (2007) and Van Dijk and Keenan, (2007) have brought some insight into the ecosystem processes occurring in forested catchments.

The benefits well managed forests assert to humankind in terms of high quality water supply are generally accepted throughout the world. In contrast to the well proven qualitative services of forests to water supply such as high quality water, decreasing sediment yields and fewer pollutants, quantitative services are more complex. Owing to the strong interactions of variables such as tree species composition, stand age, soil characteristics, climate and management which are poorly understood in many situations, generalisations cannot accurately represent the processes going on in the ecosystem. Thus, the understanding of the hydrological processes occurring in forests provides the foundation for integrated management of water resources (Dudley and Stolton, 2003).

3.1.1 Impacts of Forested Areas on Water Supply

Forest cover directly affects rates of transpiration, evaporation, soil freezing, snow accumulation and snowmelt (Hetherington, 1987). Thus, they provide a series of water related ecosystem services, which can be attributed on the one hand to over-ground processes, such as overland flow and soil erosion and on the other hand to below-ground hydrological processes. Above ground biomass, soil surface interface and the vadose zone as well as groundwater impact significantly the hydrological regime. In general, the water cycle is affected by the following processes being either directly or indirectly influenced by humans (Grottker, 1999):

- Overland processes such as surface properties, exposition, slope
- Atmospheric factors such as precipitation, radiation, wind, temperature, humidity
- Vegetation represented by plant species, age of plants, plant density, development of roots, stems and leave residues
- Soil properties such as pore volume, soil humidity, soil temperature, soil organisms, soil treatment/tillage practices and soil compaction

3.1.1.1 Overland Processes

It is well known that effects of forest cover on catchment hydrology mainly include significant alterations of surface runoff, overland flow and soil erosion (Rothacher, 1970; Troendle, 1983; Hornbeck et al., 1993; McCulloch and Robinson, 1993; Hager and Holzmann, 1997; Zhang et al., 2001; Farley et al., 2005; Debene, 2006; Sidle et al., 2006). These effects are strongly dependent on vegetation, soil and bedrock characteristics as well as climatic conditions (Farley et al., 2005). The catchment terrain can either be formed of deep (>1.5m) or shallow (<1.5m) soil over the bedrock. Additionally, the bedrock can also be directly exposed to the atmosphere, having a very low or high (extensive cracks, as e.g. in karstic areas) conductivity (Ben-Hur et al., 2011).
In general, precipitation arriving at the forest terrain penetrates the surface area and is absorbed into deeper soil layers. This water either supplies the groundwater reservoir or flows laterally as subsurface runoff if it meets an impermeable layer. As soon as the rainfall intensity exceeds the rate of infiltration as well as the surface water holding capacity, precipitation ponds at the surface or forms surface runoff (Ben-Hur et al., 2011).

The water balance as it is used in the WEPP model is therefore calculated as:

$$\theta = \theta_{in} + (P - I) \pm S - Q - ET - D - Q_d \quad (1)$$

Where
Where	Θ_{in}	initial soil water in the root zone
	Θ	soil water content in the root zone in any given day
	P	cumulative precipitation
	I	precipitation interception by vegetation
	S	snow water content: (+) snowmelt; (-) snow accumulation
	Q	cumulative amount of surface runoff
	ET	cumulative amount of evapotranspiration
	D	cumulative amount of percolation loss below the root zone
	Q_d	subsurface lateral flow or flow to drain tiles

The forest cover affects overland flow by three main mechanisms (Ben-Hur et al., 2011):

- Water use by vegetation through interception and transpiration, where high consumption of water by the vegetation decreases the available amount of water to contribute to overland flow. In addition, the infiltration rate of the terrain could be increased as a result of impacts by the vegetation.
- Barriers on the forest floor such as vegetation or litter may reduce surface runoff flow velocity. As a result, higher infiltration and lower runoff arise.
- Plant canopy and litter act as a protection for the soil surface against the impacts of raindrops. A resulting increase in aggregate stability then lowers seal formation which again enhances infiltration.

According to Wang et al. (2011) the effects of forests on catchment water yield are mainly constituted by the partitioning of precipitation into evapotranspiration and runoff in a catchment. Also Debene (2006) points out the decrease of runoff peaks in forested catchments, where higher amounts of base flow could be observed. Interception represents a significant factor in the water balance of forested areas, even though its absolute values are rather low in comparison to the volume of precipitation events which initiates floods. Debene (2006) differentiates between interception of the crown and of litter. Crown interception mainly depends on the duration and intensity of the precipitation event as well as the phenological stage and characteristics of the stand (Debene, 2006). Field measurements by Hager and Holzmann (1977) showed an interception of litter in beech stands of at least 30% of the total precipitation amount. Representative values for the interception capacity of different forest stands were published by Brechtel (1977). In addition, Brooks et al. (2003) published absolute values for maximum infiltration rates of different vegetation types.

Soil erosion in a forested catchment is represented by two main processes (Ben-Hur et al., 2011):

- Detachment of soil materials from the soil surface by raindrop splash and shear of surface runoff
- Transport of the detached sediments by raindrop impacts and overland flow

Rill and interrill erosion form the two main components of erosive processes (Foster et al., 1982). Surface runoff concentrating into small erodible channels forms rills. In these rills, soil loss occurs due to the detachment of soil particles which are then transported by the water. On the contrary, interrill erosion transports the soil as a result of raindrop splash and runoff flow (Watson and Laflen, 1986).

In forests, soil erosion can be manipulated by the following four processes:

- Again, a decrease of the amount and flow velocity of surface runoff by litter and forest floor. Thus, the erosive and transport capacity of surface runoff is decreased. Due to this fact soil loss is hampered.
- Plant canopy and litter prevents the soil surface from raindrop impacts reducing soil detachment during heavy precipitation.
- Eroded sediments are trapped from vegetation and litter on the forest floor.
- Increased aggregate stability and decrease of soil detachment as a result of the stabilising impact of plant-root activities and decomposition of the forest floor litter on soil properties.

3.1.1.2 Below-Ground Processes

In contrast to the rather homogeneous soils of arable land, forest soils are characterised by a high heterogeneity and stoniness resulting in preferential flow paths of water. In addition, forests intercept a substantial part of rain accounting for a low soil water content in forest stands (Novak et al., 2011).

For some decades soil water transport was described by the Richard's equation assuming a homogeneous soil matrix without preferential flow paths. Early preferential flow observations were declared as an exception resulting out of a lack of experimental evidence. Recent findings clearly declared preferential flow as an important hydrologic mechanism in soils. These unstable flows are even more distinct in forested soils, due to their inhomogeneous properties (Novak et al., 2011).

Matrix flow as well as preferential flow can have both impacts on various ecosystem services provided by forests. These impacts include immission load buffering, water purification, degradation of pesticides used in forestry, groundwater recharge, runoff moderation, carbon sequestration, etc. (Novak et al., 2011). In order to account for preferential flow several models such as the kinematic wave model (Germann and Beven, 1985), the two domain model (Skopp et al., 1981), the stochastic-convective flow model (Jury and Scotter, 1994) and the dual-porosity approach (Gerke and van Genuchten, 1993) or combinations of them were developed. These distinctive approaches show the high complexity of capturing water and solute transport variability in soils accruing owing to forest management activities. In order to assess the effects of various management practices on forest below-ground hydrological processes three main factors need to be in focus of analyses:

- High soil skeleton content
- Spatio-temporal variability of the vegetation cover
- Presence of surface humus

3.1.1.3 Above-Ground Biomass

Forest management includes the control of tree species composition, forest type, thinning and biomass harvesting. These activities have strong influences on below-ground hydrologic processes. Approaches for the modelling of hydrological processes mainly refer to three basic canopy characteristics, including leaf area index (LAI), albedo of evaporating surface and the roughness of the evaporating surface (Novak et al., 2011).

Some part of the precipitation falling on a stand is intercepted by canopies and evaporated back to the atmosphere. This process consumes latent heat representing a significant part of the net radiation in the soil-plant-atmosphere system. In forests, interception can amount up to 40% of the net radiation on an annual basis. This factor significantly affects the soil water regime. Alterations of canopy interception can be easily caused by forest management, resulting from a change in canopy closure and forest density (Novak et al., 2011).

Due to high LAI values and particular properties of leaves, interception capacities of forests are rather high in comparison to grass or crops. Interception losses are strongly dependent on meteorological influences including air temperature, wind velocity and rain intensity. In addition, canopy properties such as species composition, stand density and canopy closure provide for a significant amount of interception loss. In Slovakia, 37 % of total precipitation was intercepted by a temperate 80 year old Norway spruce forest canopy in comparison to 18 % of interception by a deciduous forest at the same place over a 10 year monitoring period. Additionally, considerable differences in annual interception of a mountain spruce (212 mm) and annual interception of a beech forest (87 mm) were found by Kantor (1984). These findings underline the work of Mitscherlich (1981) attributing higher interception efficiency to conifer canopies.

Looking at differences in evaporation rates between tree species, similar rates can be attributed to both, conifers and deciduous trees. With decreasing forest density and LAI transpiration rates get lower (Holst et al., 2004). On the contrary, well-structured, even-sized forests with distinct LAIs show similar levels of evapotranspiration.

Additionally, tree species composition and forest structure highly affect albedo. Beyond that, roughness of canopies influences a stand's evapotranspiration rate and energy balance. Thus, in case of low energy consumption by evaporation of intercepted water higher transpiration rates can occur.

In contrast to the impacts of tree species, forest densities and stand structures on the flow of water which were intensively studied of several tree species, impacts on soil water content (SWC) are not fully known. Generally, there are differences of up to 5-10 % of SWC in shelterwood stands compared to a natural stand. Especially forest density asserts significant effects on SWC (Novak et al., 2011). In highly competitive stands, suppressed trees are able to sustain transpiration at cloudy weather, e.g. following intense precipitation when main canopy trees do not respond. According to Black and Kelliher (1989) the amount of stand evapotranspiration stemming from the understory gets higher with an increase of water vapour pressure deficit and drier soils.

In addition, redistribution of precipitation creating distinct patterns of throughfall is strongly determined by forest canopies. In this context, stemflow plays a vital role in the production of spatial precipitation patterns (Novak et al., 2011). Beech shows great differences to spruce concerning throughfall and stemflow resulting in a distinction in stand interception and variable distributions of stand precipitation. Due to a funnel like branch architecture and the smooth bark of beech, an effective stemflow as well as concentrated infiltration of water at the stem base arises. In contrast, spruce conducts the water to the peripheries of the canopy. Thus, the drops reach the ground dispersedly (Schume et al., 2004). A substantial bigger amount of water carrying particles and compounds of dissolved chemicals arrives in areas of beech stemflow as found in zones being only exposed to throughfall (Jochheim and Schäfer, 1988).

3.1.1.4 Soil Surface Interface

Infiltrating water, originating from throughfall, canopy drip and stemflow, partly recharges the groundwater whereas the other part is stored in the soil. Subsequently, the accumulated water is redistributed and thus available as a source of water for vegetational transpiration via root extraction. A minor part of the water stored in the soil is evaporated by the surface. Infiltration of water into the soil decreases with time until it arrives at saturated soil hydraulic conductivity. Owing to the high infiltration capacity of forest soils, surface runoff rarely accrues. The surface of forested soils is prevented from soil erosion by a layer of undecomposed organic material enabling water to infiltrate into the soil. Even though the retention capacity of this surface layer is low, it represents a highly conductive layer for water entering the soil. Further redistribution of stand net precipitation (precipitation - interception) occurs in the surface humus layer determining the initial stage of infiltration (Novak et al., 2011). According to Albers et al. (2004) the influence of the leaf litter layer on this stage of infiltration and on transport pathways of solutes has not been studied intensively. Novak et al. (2011) differentiate between two flow types being mainly determined by the role of surface humus. Surface controlled flow encompasses macropore flow and heterogeneous infiltration. On the other hand matrix controlled flow includes homogeneous and heterogeneous matrix flows as well as flow fingers. Thus, different humus forms, arising due to different site characteristics, tree species compositions, forest structures and development phases significantly determine the prevailing flow types.

3.1.1.5 Vadose Zone and Groundwater

In general, forests include shallow root systems leading to a distinct pattern of depletion of soil water storage (Novak et al., 2011; Schume et al., 2004). The root density in the subsoil increases with the cultivation of deep-rooting trees. As a result, preferential infiltration into deeper horizons enlarges water-storage capacity (Novak et al., 2011). Spruce typically establishes a sinker root system than beech. Beech develops a dense, heart-shaped root system facilitating this tree species to penetrate denser soils and cope with poor aeration (Schume et al., 2004). According to Schume et al., (2004) evapotranspiration is disproportionately higher in a mixed stand of Norway spruce (*Picea abies* Karst.) and European beech (*Fagus sylvatica* L.) than in single species stands. The measured increase in evapotranspiration was fully attributable to beech and its intensive rooting system (Schume et al., 2004). Forests arrive at their final rooting depths 20 – 40 years after planting. In the following years, infiltration canals develop due to root regeneration and decomposition (Grottker, 1999).

In the determination of flow paths, roots which are not proximal to the tree stem play a minor role. In contrast, surface humus exerts high impacts on the flow of water through a minimisation of preferential flow during infiltration and a more homogeneous distribution of water over the surface. This effect can be enhanced or dampened by mineral soil properties such as biomacropores created by earthworms (Novak et al., 2011).

Low soil bulk densities and high soil porosities represent typical characteristics of the matrix of forest soils. Thus, they arrive at high hydraulic conductivities (181 – 1,290 cm day^{-1}) and high infiltration capacities. Retention capacity in forested soils is rather low, due to a high content of rock fragments from slope deposits and their shallowness.

In case of sufficient water availability for the soil, infiltration water partly reaches the groundwater table. Quantitative studies dealing with the impacts on groundwater resources of mountainous forest management are scarcely conducted (Novak et al., 2011).

3.1.2 Forest Management and its Impacts on Forest Hydrology

As the hydrological role of forests has gained substantial public attention in the last two centuries, the paired catchment approach arose as one of the earliest attempts to accurately measure hydrological responses on management practices in forested catchments.

The first systematic measurements were done by Belgrand (1853) performing a comparison of three forested catchments followed by experiments of Jeandel et al. (1862), Engler (1919), Burger (1943) etc. Based on these studies, the paired catchment approach developed with its first application at Wagon Wheel Gap, Colorado Mountains, USA (Bates and Henry, 1928). Since then, it has been frequently applied for the assessment of changes in the water cycle resulting out of different land use activities (Stednick, 1996). Paired catchment studies select two catchments being located closely to each other, offering similar characteristics regarding slope, aspect, soils, area, climate and vegetation. In the first experimental years, parallel monitoring of both catchments takes place in order to account for climatic variability. Following this calibration period, a single factor is changed in one of the catchments (treatment), whereas the second one remains unchanged (control). As a result, changes in the water balance can be attributed to alterations in one factor such as e.g. vegetational fluctuations (Brown et al., 2005; Schleppi, 2011). Andréassian (2004) points out, that most of the small catchment studies dealing with impacts of vegetation changes on the water cycle have been derived from paired catchment studies so far. In contrast, the application of hydrological models appears to be inevitable when dealing with large catchments, due to the near impossibility to find paired catchments offering suitable characteristics (Siriwardena et al., 2006). In addition, it is hardly possible to isolate variables in natural systems limiting the determination of cause and effect (Waichler et al., 2002).

Owing to this scientific database, the findings of previous studies reported in literature being mainly based on paired catchments are presented in this section. The next section describes the Water Erosion Prediction Model (WEPP). This model represents a physically based, spatially distributed model and is applied in this analysis. The application of a model should emphasize the understanding of the hydrological processes going on in the catchment area in order to allow a direct interpretation of cause-effect-relationships.

3.1.2.1 Effects of Forest Management on Water Quantity

Forest management practices (e.g. clear cutting / site preparation) can decrease forest cover and biomass of tree stands as well as alter soil structure. As a result, total evapotranspiration can be lowered, exposure of the soil to rainfall-driven erosion increased and sedimentation within the catchment enhanced. These factors may augment overland flow and soil erosion. In general, these negative effects of forestry on water quantity and quality can be limited by various water-protection techniques. One most promising preventive measure represents the avoidance of intensive forestry in areas with steep slopes or sensitive soils (Ben-Hur et al., 2011).

Based on his careful study of the effects of forest management on water yield reported in literature until 1867, Hibbert (1967) concluded the following frequently cited generalisations concerning the processes going on in the ecosystem:

- Reduction of forest cover increases water yield
- Establishment of forest cover on sparsely vegetated land decreases water yield
- Response to treatment is highly variable and mostly unpredictable

As illustrated in figure 6 Hibbert (1967) detected a linear relationship between the percent reduction of forest cover and the yield increase of the first year after intervention in 30 water-yield experiments. In these experiments, climatic conditions asserted strong effects on the magnitude of water yield increases.

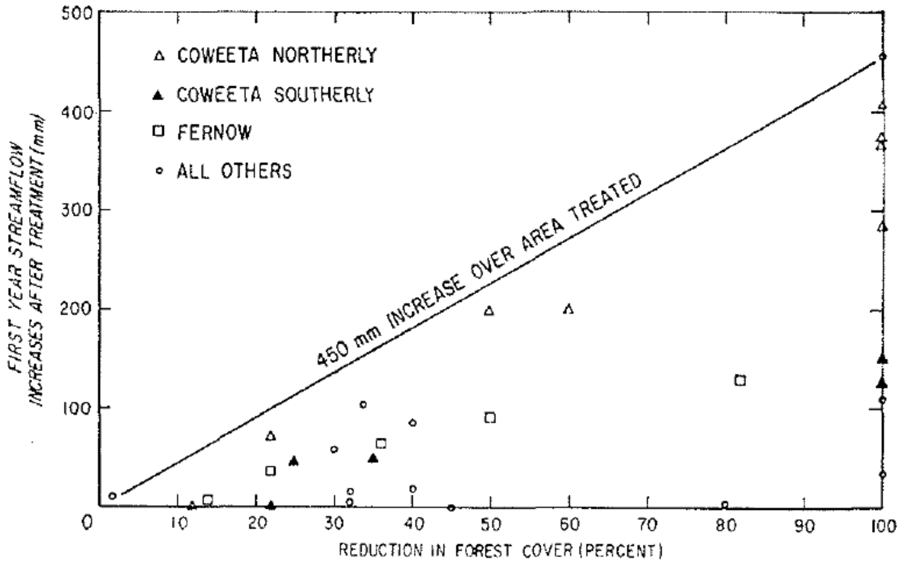


Figure 6 First-year streamflow increases after treatment in contrast to the reduction of forest cover (line representing 450 mm of increased flow) (Hibbert, 1967)

In most cases, increases of less than 300 mm were observed. A detectable limit of increases of runoff quantity and sedimentation loss is generally expected at a reduction of vegetation cover of more than 20 % in combination with small expectations concerning yield increases. Changes of water yield resulting out of a treatment of less than 20 % of the catchment area are not statistically significant (Hibbert, 1967; Bosch and Hewlett, 1982; Vilhar and Fajon, 2007). Nonetheless, Serengil et al. (2007) observed an alteration of stream nutrient discharge including sedimentation as a result of a thinning representing an 11 % reduction in biomass of a deciduous forest. Thus, the rate of the influence of frequency and intensity of thinning on forest hydrology is mainly determined by local conditions such as climate, soil and vegetation (Serengil et al., 2007). Largest increases in water yield occur in harvested areas where the moisture content of the soil is actually high during the growing season (Novak et al., 2011).

Impacts of forest management can still be measured 80 years after the disturbance took place (Chang, 2006). Based on the review of paired catchment studies, Hornbeck et al. (1993) found a variety of water yield responses on forest treatment:

- Initial increases occur promptly after forest clearing
- Resulting increases could be prolonged by controlling of regrowth, whereas increases in streamflow diminished in about 3-10 years when regeneration was permitted
- Small increases or decreases in water yield may persist for at least 10 years

Bosch and Hewlett (1982), Stednick (1996) and Scherer and Pike (2003) estimated the return of water yield to pre-treatment levels in approximately 10-30 years after the cuttings. Effects of treatment may also appear rather short-lived owing to the non-stationarity of forests.

According to the findings of Hibbert (1967) and Swank et al. (2001) impacts of forest treatment arrive at a level close to zero after a period of 7-25 years. It even decreased to a negative value as a result of changing tree species during regrowth including bigger amounts of trees with lower stomatal resistance (Hornbeck et al., 1997). These findings illustrate the importance of detailed stand analysis as a watershed description solely based on forest surface area does not include any information about alterations through seasonal variations.

Additionally, ground vegetation is of major importance for the hydrological regime, especially concerning nitrogen concentrations in the groundwater as well as drainage amounts. Small-scale forest interventions avoid soil degradation and erosion. Beyond that, a mixed tree species composition adapted to natural growth conditions exerts positive effects on forest hydrology in terms of soil and stand stability due to diverse rooting systems (Pilas et al., 2011).

Table 1 presents eight forest management measures asserting significant impacts on drinking water quantity and quality.

Table 1 Impact of forest management measures on drinking water quality and quantity indicators (Pilas et al., 2011)

Indicator *Forest management measure*	Concentration of pollutants in the water	Nitrogen content in the water	Sediment loss (erosion)	Runoff
Clear cut area	+	+++	+++	+++
Frequency, intensity, technique of harvesting	+++	+	++	++
Tree species composition	++	+++	++	+++
Crown density, cover percentage	+	+++	+++	+++
Distribution of growth classes	+	++	++	++
Vertical and horizontal stand structure	+	++	++	++
Forest regeneration, ground vegetation	+	++	++	++

+ low impact ++ medium impact +++ high impact

All forest management practices presented in Table 1 have medium or high impacts on sediment loss and runoff. In addition, frequency, intensity and technique of harvest as well as tree species composition highly influence the concentration of pollutants in the water. Nitrogen contents in water are mainly associated to clear cut area, tree species composition, crown density and cover percentage. Furthermore, the magnitude of these effects depends on soil and climatic characteristics (Pilas et al., 2011). Pilas et al. (2011) do not state any details about the backgrounds of the investigation of these tendencies. As a result, e.g. a low medium impact of clear-cut area on the concentrations of the pollutants in the water being highly dependent on tree species composition of a stand, regeneration practice and stand age cannot be generalised. The following sections provide more detailed information dealing with the interrelationships of various measures of forest management and their impacts on water quantity as well as quality.

In order to predict impacts of permanent vegetation changes on evapotranspiration and runoff Zhang et al. (2001) developed the 'Zhang curves'. As illustrated in figure 7, this two parameter model is a result of a combination of paired catchment and time-trend studies on the catchment scale predicting the difference in evapotranspiration of different vegetation types along a rainfall gradient (Brown et al., 2005).

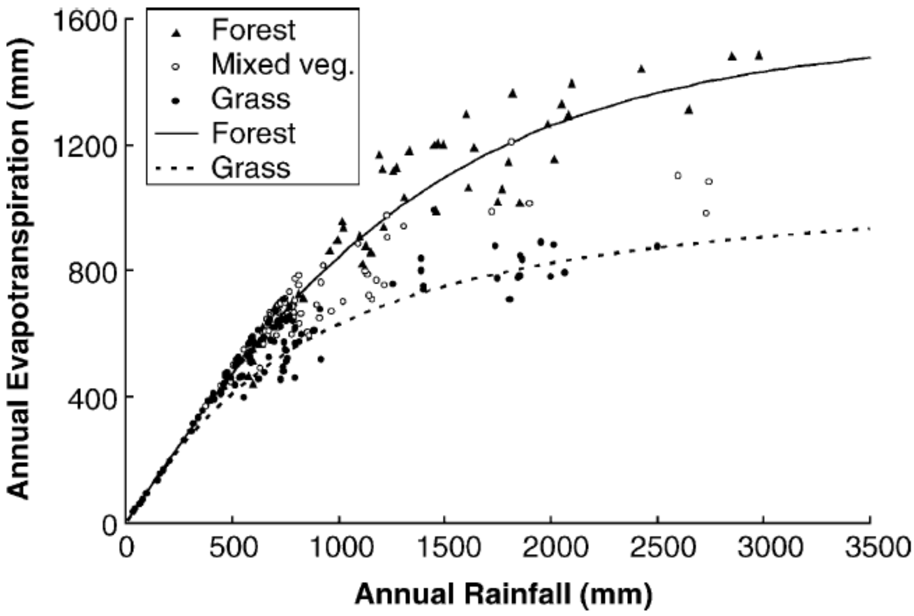


Figure 7 Relationship between land cover, mean annual rainfall and mean annual evapotranspiration derived by Zhang et al. (2001)

His analysis of 166 catchment studies basically brought up the following generalisations:

- A reduction in forest cover causes increases in water yield
- Increasing forest cover causes a decrease in water yield
- Coniferous trees provide a change in annual water yield of 40 mm per 10 % change in forest cover
- Deciduous hardwoods cause a 25 mm change in annual water yield per 10 % change in cover
- Brush and grasslands are associated with 10 mm changes in annual water yield per 10 % change in cover
- Changes in stream flow are determined by mean annual precipitation in the catchment area and precipitation as well as temperature patterns of the year under investigation

By trend, the results of the Zhang curves overestimate the mean annual water yield change in comparison to those observed by studies of paired catchments (Brown et al., 2005). Despite to expected increases of water yield, according to Brown et al. (2005) dry seasons flow can result as a result of forest removal. If the forest treatment decreases infiltration rates to the extent, that the quick surface flow leaving the area exceeds the gain in baseflow due to decreased evapotranspiration, dry season flows will occur. Zhang et al. (2001), Holmes and Sinclair (1986) as well as Turner (1991) see evaporation as the main driver of alterations in water yield resulting out of vegetation change.

Looking at the *intensity of a treatment*, water yield increases as a result of clear cutting substantially more than after a partial harvest or thinning, as presented in figure 8.

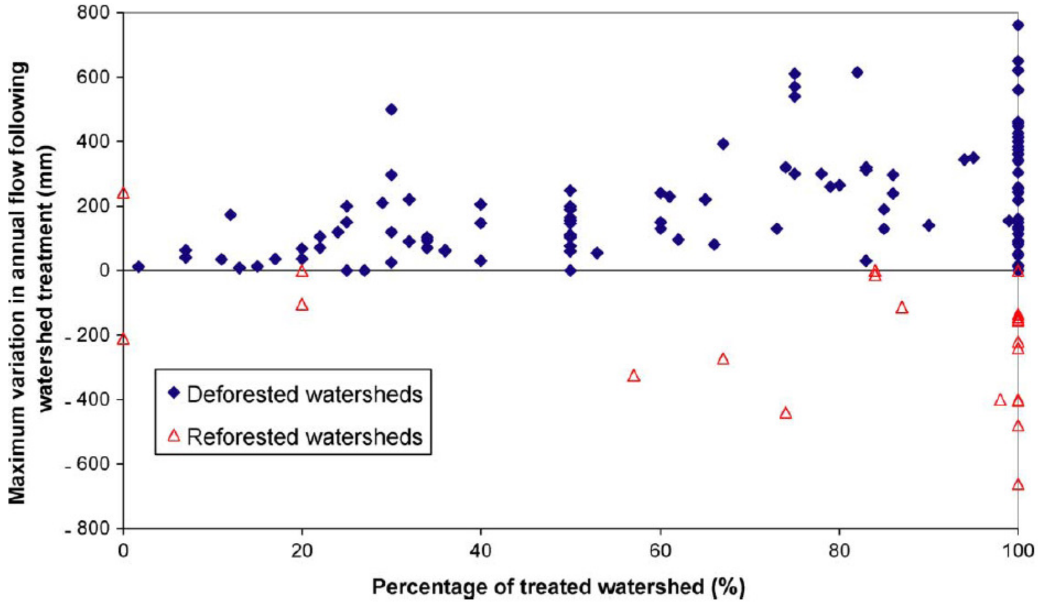


Figure 8 Maximum variations in annual flow after watershed treatment illustrated as a function of percentage of the catchment subjected to treatment (Andreassian, 2004)

According to Andreassian (2004), maximum variations in annual flow increases significantly with higher treatment area. Investigation results of Ben-Hur et al. (2011) published an increase of overland flow resulting out of a decrease in water use by trees, the formation of a denser network of water channels at the surface of the deforested area and the destruction of the soil structure in the clear-cut area by land drainage and site preparation.

Moreover, evergreen forests consume more water than deciduous. In stands of evergreen trees a rough estimate of 30-40 % of the precipitation arrives on free areas (Grottker, 1999). In contrast, a forest stand being dominated by deciduous trees only transpires during the growing season (Pilas et al., 2011). Deciduous species transpire about 10–20% of the precipitation on free areas in the non-growing season. During the growing season, evapotranspiration does not differ considerably between evergreen and deciduous species (Grottker, 1999). Due to the mostly darker foliage of evergreens a difference in albedo offering more energy available for evaporation is given (Pilas et al., 2011). Generally, evergreen conifers account for a lower maximum stomatal conductance than deciduous broadleaves (Bond et al., 2007). In practice, actual transpiration depends on stomatal closure and leaf area index (LAI). Water yield resulting out of the removal of vegetation strongly depends on the actual evapotranspiration rates of the previous vegetation being rather high in forests compared to grasslands (Pilas et al., 2011).

Additionally, vegetation density and thinning represent important components influencing the water budget of managed forests (Pilas et al., 2011).

According to deforestation experiments in the US conducted by Chang (2006), a minimum *soil depth* of 1 m is critical to observe effects of the removal of vegetation on annual water flow. Owing to the impediment of developing deep roots in shallow soils, trees cannot access deeper water reserves than herbaceous species do (Pilas et al., 2011). In the latter case, different impacts between forests and grass on water yields will be reduced to their different interception capacities (Andreassian, 2004).

The amount of *precipitation* available in a region may also assert significant effects on the amount of water becoming available due to a change in vegetation (Pilas et al., 2011). Chang (2006) demonstrated that there is no measurable effect of vegetation removal on water yield below 400 mm annual precipitation.
In forested catchments, the modelling of *snow water equivalents* represents a more challenging task than in open land. A comparison of snow accumulation patterns in the US revealed larger tendencies of accumulating snow in deciduous than in coniferous forests. On meadows, the biggest amount of snow accumulated. Total discharge patterns during snowmelt reacted according to these findings, as illustrated in figure 9.

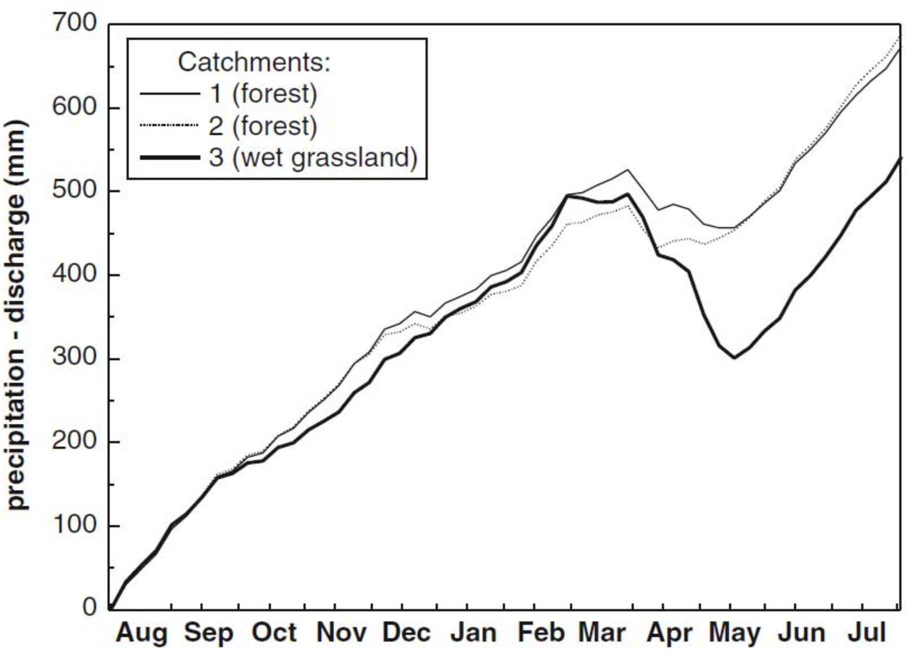
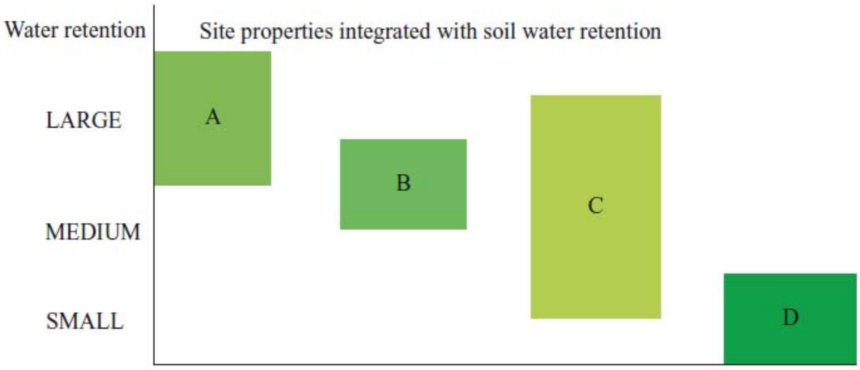


Figure 9 Water balance of three experimental catchments in Switzerland, where larger water losses due to snowmelt accrue from catchments covered by grasslands (Schleppi, 2011)

Schleppi et al. (2011) observed reduced durations of snow melt in forested catchments.

As a result, forest are earlier snow free than meadows. This difference gets bigger with increasing altitude until close to the timberline. As there were no significant differences in sublimation rates from the snow pack, the reduction of snow pack in forests stems from a combination of interception and sublimation from the tree crowns over winter (Pilas et al., 2011).
Seasonal differences are not only determined by snow and weather but also by the seasonality of foliage of deciduous tree species (Pilas et al., 2011). In Hubbard Brook (New Hampshire, USA), impacts of deforestation were much higher during the vegetation period than during the dormant season (Hornbeck et al., 1997).
In addition, transitional features of forested stands need to be taken into account due to changes in the hydrological behaviour of stands as a result of aging of trees (Andreassian, 2004). With rising tree stand age transpiration gets lower (Yoder et al., 1994) and water yield increases. This can be traced back to a decrease in leaf are index in aging forests combined with a reduced hydraulic conductance of old trees (Bond et al., 2007).
Looking at the seasonal changes in water yield resulting out of a change in vegetation cover, soil moisture content represents a factor affecting the water balance, but which has a rather negligible contribution to mean annual changes. Nevertheless, the effect of reforesting reflects not only a change in the balance of infiltration and evapotranspiration but also is affected by available soil water storage capacity. Changes in water yield can then result in changes of surface runoff, baseflow or both of them (Brown et al., 2005).

Forest mechanisation leads to soil compaction resulting in decreasing water retention capacities of soils (Croke et al., 2001). Due to the compaction of the soil, infiltration is hampered and surface soil erosion occurs. As soils are only capable of storing water if the water saturation of soils is not reached, forest condition influences the water retention capacity of the soil significantly being illustrated by figure 10 (Pilas et al., 2011).



A – large water retention capacity of soils, irrespective of forest condition. Sites on deep, well drained soils.
B – medium water retention capacity of soils, irrespective of forest condition. Sites on medium deep, well drained soils.
C – large water retention capacity of soils in case of good forest condition / low water retention capacity of soils in case of bad forest condition. Sites on deep, slowly drained soils.
D – small water retention capacity of soils, irrespective of forest condition. Sites on extremely wet, shallow or permeable soils.

Figure 10 Implications of forest condition on soil water retention capacity with regard to large precipitation amounts (Frehner et al., 2005)

The impacts on water retention capacity arising as a result of forest condition are largest in deep, slowly drained soils (C). Soils providing a low water retention capacity show low responses to changes in forest management, including extremely shallow, extremely permeable or extremely wet soils (D). Maximal amounts of soil water retention capacity are given in deep, well drained soils, being also independent on forest management practices. Changes in forest management in such regions may not improve soil water retention capacity but can decrease the development of surface runoff and erosion. Therefore continuous cover forestry should be priority in forest management (Pilas et al., 2011).

In general, decrease of forest cover in a catchment can lead to sediment loss owing to erosion processes (Binkley and Macdonald, 1994; Prybolotna, 2006), increase of high waters after storms and snow melting (Von Burger, 1954).

3.1.2.2 Effects of Forest Management on Water Quality

Water quality can be highly affected by different forest management practices including type of cutting, harvesting intensity, tree species composition, regeneration method or fertilisation. In general, major issues related to water quality in forested catchments represents the acidification of water resources, eutrophication of the aquatic system, nitrate contamination of the groundwater soil erosion and export of suspended particles into surface waters and the remobilisation of heavy metals and organic pollutants (Novak et al, 2011).

Looking at forest management and its impacts on the water cycle, studies of clear-cutting experiments at Hubbard Brook clearly revealed the applicability of paired catchment studies for this issue (Likens et al., 1970). Among the physical chemical and biological parameters describing water quality, this analysis focuses on concentrations of suspended particles and turbidity. Both of them are directly related to erosive processes (Schleppi, 2011).

On the one hand, contamination sources can be deposited in forests due to precipitation, soil release through weathering, mineralisation of organic matter or leaching from canopies with throughfall or stemflow. On the other hand, elements leave the system by biological consumption, volatilization, evaporation, percolation to the aquifer, leaching out of the watershed through surface and subsurface runoff, storage in soils by ion exchange, fixation or mineralisation (Chang, 2006).

Trees are able to assert various effects on water quality directly and indirectly. One of these effects is related to an increase in dry deposition, due to the size and aerodynamic roughness of the tree canopies (Schleppi, 2011). In addition, evaporation of water causes an augmentation of the concentration of solutes (Swanson et al., 2000). These impacts are more severe in stands being are covered by evergreen species than in stands covered by deciduous trees (Schleppi, 2011). According to Frijns and Tietema (2002) forest clear-cut could even increase water quality resulting out of a reduction of pollutants in the water reaching the soil in areas with high deposition rates. As trees develop deeper rooting systems in conjunction with associations to mycorrhizae, they are able to better access water and nutrients of larger soil volumes than other plants. High transpiration and low water yields of forests concentrate the solutes which may have a negative impact on water quality. In contrast, even small reductions of crown cover can lead to significant increases in the concentrations of ions such as NO^{3-}, NH^{4-}, Ca^{2+}, K^{+} in the stream water according to Wang et al. (2006). Also the soil water can show significant increases of ion-concentrations after felling of small groups of trees (v. Wilpert et al., 2000).

On the other hand, deep roots (re)capture nutrients from deeper soil layers resulting in a closed element cycle and improved water quality (Schleppi, 2011). As forests reduce the amount of runoff in a catchment, they hamper the transportation of sediments and elements to the streams. Due to a reduction in runoff, shielding and shading of canopies, binding effects of root systems and the screening effect of the forest floor, forested catchments provide a decrease in sediment, dissolved elements, cooler temperature and increases of dissolved oxygen (Chang, 2006).

Forest practices may lead to soil disturbance due to the removal of the vegetation causing a high exposition of the ground to environmental impacts. Thus, the hydrologic functions of forests are impaired leading to decreased water quality. Negative effects on forest management can be minimised through the application of best management practices (BMP) in order to control nonpoint sources of water pollution (Chang, 2006).

Clear-cutting represents the most intensive form of forest harvesting having the greatest potential to negatively affect the water quality of streams. Stream sediment, temperature, concentrations of water chemicals, dissolved oxygen, biochemical oxygen demand, specific conductance, pH, fecal colifom, fecal streptococcus and the aquatic environment can be influenced as a result of clear cuts (Chang, 2006).

Previous studies (e.g. Hubbard Brook experimental forest – Hornbeck et al., 1993) revealed rapid decomposition of organic matter and extensive leaching of major ions except sulphate, carbonate and ammonium in the discharge after clear-cut. This was especially observed with the major plants nutrients nitrate and potassium, showing an increase of NO_3-concentrations from 0.015 to 0.7 mg (Likens et al., 1970; Chang, 2006). Clear-cuts in North Carolina showed significant augmentations of suspended solids, total N, total P and fecal coliformes (Ensign and Mallin, 2001).

Additionally, the mobilisation of humus layers of the Ah horizons in combination with nitrate exports resulted in a contamination of soil water in the calcareous Alps. Clear-cut experiments in Upper Austria also resulted in strong leaching of nitrogen and potassium (Köck, 2008). As exports of N and K are directly related to forest management, increases of phosphorous concentrations in streams are rather related to erosion due to its presence in suspended matter (Stednick, 2000).

According to Novak et al. (2011) the increase of nitrate concentrations in seepage water after interferences such as clear-cut, windthrow or insect outbreak has been shown in various studies such as Huber et al. (2004), Weis et al. (2006) or Legout et al. (2009). High nitrogen output after clear cuts result from higher infiltration into the soil in combination with elevated nitrate concentration owing to enhanced nitrification. The highest outputs occur in the first years after cut and could lead to negative effects on drinking water supplies in fast draining soils depending on the size of the harvested area. Beyond that, the increase in nitrogen concentration is dependent on the nitrogen saturation of a specific site, where region with high nitrogen concentrations show much higher values of increase of nitrogen. Additionally, stand age influences the nitrogen uptake rate of forests leading to increasing nitrogen losses with declining forest growth rates. Thus, the combination of Norway spruce with European larch in a shelterwood system representing an example for continuous cover forestry could prevent soil losses of nitrogen, magnesium and calcium especially in the beginning of stand regeneration (Novak et al., 2011).

Figure 11 shows the response of average annual concentrations of suspended solids, total nitrogen and organic carbon in the overland flow of four Finnish catchments to clear cutting and land drainage. In figure (a) the clear-cut in combination with land drainage leads to a significant increase of concentrations of suspended solids, nitrogen and carbon in the overland flow. Increased values of suspended solids could be observed for 14 years, elevated nitrogen for 4 years and total organic carbon was increased for 4 years (Ben Hur et al., 2011).

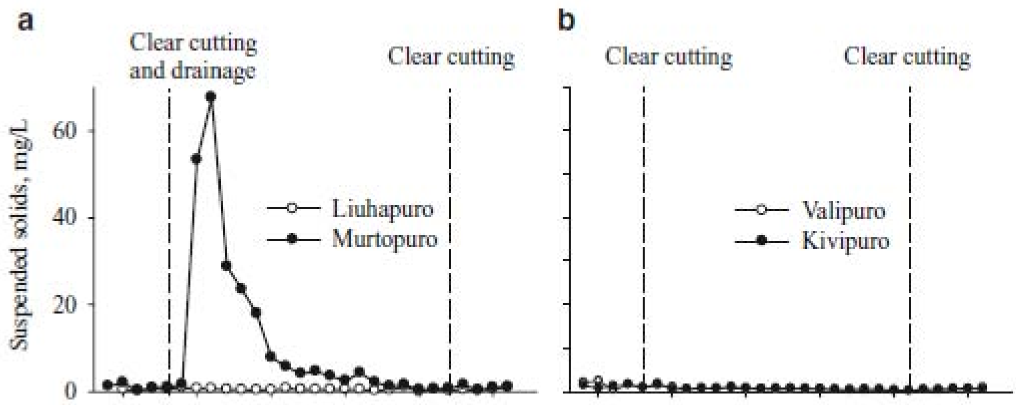


Figure 11 Average annual concentrations of suspended solids, total N and organic C in surface runoff after implementation of clear cutting and land drainage at four catchments in Finland (Ben Hur et al., 2011)

Comparing figure (a) with figure (b), figure (b) shows the reaction of the catchments to a disturbance of the forest floor by limited clear cutting. This interference was probably not sufficient to lead to clear changes of the parameter concentrations in overland flow (Ben Hur et al., 2011).

The limitation of augmented nutrient losses in time was confirmed by various other experiments. In a study conducted by Neal et al. (1992), losses of elements occurred for 5 years resulting out of decreased plant uptake and fast decomposition rates. Limitations of nutrient losses can arise in case of fast-growing early-succession species. In fact, even a *small scale biomass extraction* such as forest thinning or single tree extraction can cause a significant increase in fluxes of sediments, base cations and chloride (Serengil et al., 2007; Wang et al., 2006). Warm and dry summers as well as insect calamities or application of fertilizers may lead to intensive nitrification (Köck, 2008).

Furthermore, *forest roads* represent an essential precondition for the accessibility of forest stands for forest management activities, protection and recreation. Various studies dealing with the effect of forest roads on water quality revealed high road-induced erosion rates. In fact, as much as 90 % of all sediment produced on forested areas is estimated to originate from forest roads. Therefore, erosion rates in forestlands can be substantially reduced with adequate forest road design (Chang, 2006). Clear-cuts for forests roads lead to an increase of evaporation. In addition, forest roads increase the dotation via road passages, which leads to an increase of surface runoff and evapotranspiration of areas lying downslope of the road (Debene, 2006).

Additionally, impacts of forest management on water quality include those of *mechanical site preparation*. Mechanical site preparation includes follow-up works of forest harvesting such as site preparation for planting and seeding. Therefore, heavy machinery is used in order to treat standing trees, debris, stumps and the ground. As a consequence, changes in soil structure may occur including increases in bulk density and percent of silt and clay at the surface horizon (Chang, 2006). Additionally, the distribution and composition of species of the ground vegetation is altered in combination with an exposition of the soil surface to erosion (Koivusalo et al., 2011).

Furthermore, tree species composition can play a vital role influencing contaminant concentrations on soil water of forest stands. In this context, proportional amounts of beech in a stand promise substantial decreases of deposition rates. Spruce represents a tree species offering a high filtering capacity of air pollutants. As a result, spruce dominated stands account for highest deposition rates of nitrate. Reported deposition rates of inorganic nitrogen under beech stands lie 45–85 % under that measured under spruce stands (Köck, 2008). Moreover, groundwater under areas covered with coniferous trees exhibits higher susceptibility to nitrate contamination than under broadleaved trees. Thus, quality of percolating water is strongly dependent on tree species composition of a stand, but also regeneration practice and stand age may contribute to a change in quality (Ahrends et al., 2005). Figure 12 underlines these results showing highest rates of N deposition in the spruce stand (a). Beyond that, also the values of NO_3 leaching are highest on spruce dominated stands (b).

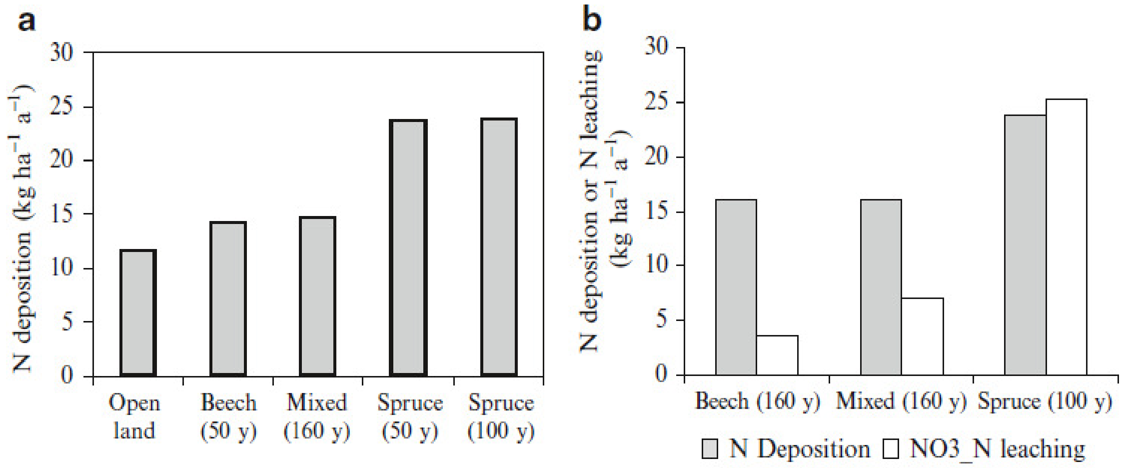


Figure 12 Mean nitrogen deposition [kg N ha $^{-1}$ a^{-1}) between 1992 and 2002 in open land, beech stand, mixed stand (beech-spruce-fir) in a spruce stand of different ages (a). (b) shows N deposition, nitrate nitrogen leaching [kg N ha^{-1} a^{-1}) in an old beech, a mixed and a spruce stand for the period 1995-1997at a depth of 120 cm (Raftoyannis et al., 2011; modification of v. Wilpert, 2007)

Despite to the fact that nitrate concentrations under spruce dominated stands tend to be higher, concrete contamination risks for water quality are dependent on regional concentrations of airborne pollutants.

Owing to a relative low exposure of the catchment area of the Viennese drinking water supplies to airborne pollutants, the next section concentrates on the contaminating power of increased sedimentation rates. According to Ripl (1995) and Chang (2006) sedimentation represents the most common form of water pollution arising as a result of forestry activities.

Generally, increasing forest areas come along with reduced erosion and sediment production. Thus, erosion rates are low due to soil stabilisation by the roots and the protective effects of leaves and the humus layer in undisturbed forests. Deforested areas result in a higher runoff and sediment production (Pichler, 2011). Forest management practices such as road construction and harvesting significantly increase concentrations of suspendend sediments in streams. As natural variations of sediment concentrations can be very large (annual average concentrations of undisturbed draining forests < 5 mg/L), increases due to management practices can also be highly variable (Binkley and Brown, 1993). Timber harvest units can be stated as the largest areas of anthropogenic disturbance contributing to increases of erosion of one to five times in comparison to undisturbed areas (Motha et al., 2003).

Looking at turbidity levels, Ensign and Mallin (2001) could not report a significant response before or after clear-cutting in a blackwater swamp. Anyway, during the first month of clear-cutting a large peak in turbidity of values twice as high as the state standard was recognised. Additionally, various studies confirm a significant increase in suspended solids after logging (Riekerk, 1983; Fisher, 1981). Beasley and Granillo (1988) observed mean annual sediment losses significantly higher in the logged than in the control watershed for two years following a clear-cut. On the other hand, Messina (1997) as well as Blackburn and Wood (1990) did not find any response in turbidity due to clear-cutting.

Binkley and Brown (1993) conducted a study in a hardwood/pine forest in Mississippi, in which harvesting was followed by a series of site preparation treatments. In the first year after treatment annual average sediment concentrations rose from 2127 mg/l (control) to over 2800 mg/l at the intensive site preparation treatments. In the second year, annual average concentrations of suspended sediments decreased to 393 mg/l in the control, whereas sediment concentrations in the treated catchment remained over 2300 mg/l. The findings of 40 experiments in forested areas in the USA resulted in the overall conclusion of highly site-specific responses of sediment concentrations dependent on forest treatment intensity and technique (Binkley and Brown, 1993).

Jordan (2006) observed a peak in runoff and sediment yield during the snowmelt period in British Columbia arising due to a rain-on-snow event. Other peaks were mainly caused by radiation driven snowmelt. In Jordan’s study, highest turbidity values accrued on the first few peaks of the hydrograph with declining values during the main snowmelt runoff peak. This hysteresis effect can be traced back to the first rise of water level in spring washing away sediment and organic material accumulated along channel margins and storage sites during the previous period of summer and fall. In addition, small-scale erosion contributes to this hysteresis as the snowline rises and new erosion sites are exposed every day (Jordan, 2006).

Furthermore, Jordan revealed a correlation between runoff and sediment yield. His results point out significant effects of varying precipitation intensities on sedimentation accruing due to logging or forest roads. Also Litschert and MacDonald (2009) observed a high correlation of mean annual precipitation and post-harvest years. This leads to the conclusion that bedrock geology represents next to precipitation an essential factor in determining the susceptibility of watersheds to impacts of sediments from forest management practices. Impermeable geologic properties cause runoff routing through surface or subsurface flows resulting in a high risk of contamination (Jordan, 2006).

Dealing with karst aquifers, the most significant contamination can be expected as a result of clear-cutting or wide spread windthrow. Thus, humus substances can act as contaminants showing increased mobility. After a harvest interference including a large reduction of forest cover soil carbon contents decline because litter fall decreases and the decomposition rate in the ecto-humus layers and the upper soil horizons increases. Main reasons for this behaviour can be derived from higher soil temperatures, changes in the soil moisture conditions and the breakdown of litter fall due to clear-cutting activities including large openings of the forest cover due to shelterwood cuts. As a result, a limitation of the quantity of timber yield provides the highest potential to maximise the protection of drinking water resources (Köck et al., 2007).

3.1.3 Modelling of the Hydrological Processes

Hydrologic models aim at a simplification of reality based on mathematical equations. The model input should allow the simulation of hydrologic processes by the application of parameters and variables representing characteristic features of a system. These variables usually exhibit a spatio-temporal variability, such as precipitation, evaporation or temperature (Singh, 1988). Hydrologic models seek to integrate state of the art knowledge about hydrologic processes into computerized tools including new technologies such as geographic information systems (Pike, 1995). They can be classified according to their process description discriminating between metric models, conceptual models and physically based models (Pechlivanidis et al., 2011).

Metric models represent empirical models which are mainly based on observations such as the unit hydrograph theory. Their advantage lies in the simplicity of application due to the establishment of a relation between model properties and physical and climatic properties of a catchment. Nevertheless, the results generated by metric models are dependent on the availability of data and are usually lacking in the specification of confidence limits (Pechlivanidis et al., 2011).

In contrast, the structure of *conceptual models* is specified prior to any modelling and not all of the parameters used in the model show a direct physical interpretation. As a result, some of the parameters need to undergo a process of calibration with observed data sets. These models normally include all important hydrological processes of a catchment using input-output-relationships. Thus, there is a high variety of conceptual models of differing complexity. The structure of the model is mainly based on extensive use of schematic storages. Very complex models are highly dependent on sufficient availability of information, whereas simple models do not reflect the rainfall-runoff response fully satisfactory. The complexity of a model can be reduced by applying identification statistics or sensitivity analysis (Pechlivanidis et al., 2011).

Physically-based models use equations of motion based on continuum mechanics in order to model the hydrological processes of catchments. These equations are solved numerically with spatial discretisation procedures. Theoretically, it should be possible to use these models without calibration as the parameters are fully measurable. Nevertheless, the physics behind the model rely on laboratory or small-scale in-situ field experiments, which make them sensitive to errors stemming from the nature of the experiments themselves. Also the extrapolation to larger catchment scales as well as the application of simplified physics/mechanics which are used in order to reduce computational time can raise the uncertainty of their results. What's more, the measurement of the parameters used in physically-based models can show a number of problems. Due to the high heterogeneity of catchments, it is very difficult to observe them with point measurements. Owing to this fact, the models are fed with averaged variables and parameters which represent a bigger scale than the variation of the processes shows (Pechlivanidis et al., 2011).

3.1.3.1 Model Selection

As pointed out in various publications (Alila and Beckers, 2001; Pike et al., 2007; Beckers et al., 2009) the prediction of the effects of forest management on the hydrologic processes represents a challenging task. In general, it is not straight forward to get valid conclusions about the impacts of forest activities on catchment hydrology derived from data alone as field experiments have large practical limitations including high cost, long duration, sparse monitoring points and restrictions in the transferability of results (Dunne, 2001; Beckers et al., 2009). Beyond that, cause-effect relationships are shaded by climatic variability and many other factors including silvicultural methods having strong implications on the effects forest management asserts on watershed hydrology (Alila and Beckers, 2001). Differing silvicultural treatments such as technique of timber extraction, logging, road management and the location of harvesting in a catchment may have cumulative or overlapping effects (Beckers et al., 2009). Therefore Beckers et al. (2009), Ziemer et al. (1991), Dunne (2001) and Alila and Beckers (2001) strongly recommend the supplementation of field measurements with numerical modelling.

There is a multitude of tools for the modelling of ecosystem services available. The selection of the best suitable model was mainly based on six criteria describing the most important requirements for the suitability of various alternatives for this analysis:

- Representation of hydrological processes
- Feasibility of the integration of forest management practices

- Representation of stand characteristics
- Model output
- Model availability
- Usability

Therefore, publications dealing with state of the art hydrological models provided a high volume of information for the model selection (e.g. Singh and Frevert, 2002), where in particular the review of hydrologic models for forest management applications published by Beckers et al. (2009) delivered a well-structured basis for evaluation. In addition, it is inevitable to partially integrate the personal bias of the user in the choice of model. Table 2 presents an exemplary enumeration of models, which were considered in the selective process of choosing the most suitable tool.

In general, there is no optimal model fitting to every specific forest management case study. As a result, model selection needs to be based on site-specific factors. This selective process represents a complex procedure in practice as it should take into account a variety of considerations (Beckers et al., 2009).

Table 2 Available tools which were considered in the selection process in order to model hydrologic ecosystem services in combination with erosive processes (model accuracy and complexity mainly based on Beckers et al. (2009))

Model	Type	Developing group	Process representation	Integration of forest management	Model output	Complexity	Availability	Usability
COSERO	cont. semi-distributed	IWHW	high accuracy	principally possible	Runoff	high	IWHW, free	user-friendly
WEPP	cont. distributed	USDA	medium accuracy, higher than SWAT	not satisfactory	runoff, erosion	high	web, free	very user-friendly
SWAT	cont. distributed	USDA	medium accuracy (erosion based on USLE)	not satisfactory	runoff, erosion	high	web, free	user-friendly
TOPOG	deterministic	CSIRO	high accuracy	principally possible	runoff, erosion	high	web, free	no experience
InVEST	black-box	Natural Capital Project	strong simplifications	principally possible	runoff, erosion	low	web, free	no experience
DHSVM	cont. distributed	University of Washington	high accuracy	very good integration	runoff, erosion	high	web, free	not executable for new users
PRMS/ MMS	determ. distributed	US Geol. Survey	medium accuracy (erosion based on USLE)	medium integration	runoff, erosion	high	web, free	no experience
Mike-She	cont. distributed	DHI Water & Environment	high accuracy	medium integration	runoff, water quality	high	costly	no experience
Mod-HMS	cont. distributed	HydroGeoLogic Software Systems	high accuracy	medium integration	runoff, erosion	high	costly	no experience
PREVAH	semi-distributed	ETH (Suisse)	high accuracy	good integration	Runoff	high	web, free	no experience
RHESSys	semi-distributed	University of California	high accuracy	good integration	runoff, nutrient fluxes	high	web, free	no experience
WaSIM – ETH	fully-distributed	ETH (Suisse)	high accuracy	good integration (but only LAI)	runoff, sediment in combination with AGNPS	high	web, free	no experience

The search for a model integrating basic hydrologic processes into the calculations and which produces accurate measures of runoff and erosion of a forested catchment resulted in the selection of the DHSVM[1] model. This model figured out to be most suitable for this purpose as it was explicitly developed for applications studying impacts of silvicultural treatment in mountainous forested catchments. According to Beckers et al. (2009) DHSVM should be preferred in mountainous terrain having the greatest forest management functionality of the studied models. On the other hand, only limited effort has been paid to make the model user friendly. Additionally, high amounts of input data are needed including a digital elevation model determining the size of the computational grid elements and the spatial resolution, a soil map, a map of the soil thickness, a vegetation map, an input file for the stream network, a meteorological file including precipitation, air temperature, air humidity, short-wave and long-wave radiation and wind speed. The meteorological parameters can then be distributed in the model using a temperature lapse rate and a precipitation lapse rate making it suitable for complex terrain. DHSVM divides the vegetation into two layers dividing forest canopy into overstory and understory. It also uses various soil layers. As a result, soil evaporation, plant transpiration, evaporation of rainfall intercepted by vegetation, canopy snow interception and ablation and snowmelt are calculated by physically based methods. The calculation of vertical unsaturated water movement is based on the one-dimensional form of Darcy's law recharging the grid cell water table. Additionally, a transient, three dimensional representation table is used to describe subsurface lateral flow (Wigmosta et al., 1994). It also allows the user to simulate erosive processes using a sediment model being set up of four components: mass wasting (stochastic), hillslope erosion, erosion form forest roads and a channel routing algorithm (Doten et al., 2006). In locations where the water table of a grid cell intersects the ground surface, return flow and saturation overland flow are generated. What's more, open channel routing is based on explicit information on the location of the channels of streams (Wigmosta et al., 2002). In general, DHSVM represents a complex model requiring substantial data collection, pre-processing and GIS analysis. Beyond that, no technical support, despite of a mailing list, is available. Due to this fact, it was not possible to apply the DHSVM in this thesis. After an intensive period of data collection and processing, various errors in running the source code erupted which unfortunately rendered the application of the model in this context impossible.

As the application of the DHSVM was not feasible, the process of model selection had to be repeated. Therefore user friendliness developed to be the most important factor in the selection procedure.

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[1] Distributed-Hydrology-Soil-Vegetation-Model

The COSERO model represents a continuous, semi-distributed tool for the modelling of rainfall-runoff processes. This model was developed by the Institute for Water Management, Hydrology and Hydraulic Engineering of the University of Natural Resources and Applied Life Sciences Vienna based on the ENNS-model. The processes of runoff formation and runoff concentration are simulated spatially distributed in various modules. Additionally, sub-catchments can be divided into hydrological response units (Debene, 2006). Nevertheless, the modelling of the impacts of land use change on sediment delivery has not yet been implemented in the model.

In contrast, WEPP represents a model being clearly aimed at predicting water erosion resulting out of land use changes in the agricultural sector. It was extended to enable the integration of forest management practices into the model in 2004. Additionally, it does not use the USLE[2] as basis for modelling erosion as it is done by SWAT. The USLE was the first widely used erosion model (Wischmeier and Smith, 1978) and is still applied all over the world. More recently, it was replaced by the Revised Universal Soil Loss Equation (Renard et al., 1997). This empirical model establishes coefficients for various factors affecting erosion such as runoff, precipitation, soil, etc. by measurements and observations. Thus, it does not simulate the erosion process as a set of physical processes. Physically based models, such as WEPP or TOPOG, are simulating individual components in the process of erosion by solving the corresponding equations. Thus, they can be used to a much wider range of applications than the empirical parameters of the USLE which have to be adapted to the conditions of each new field of investigation. To sum up, the physically based models use a spatial and temporal variable estimates for erosion processes (Shen et al., 2009). The analysis of Shen et al. (2009) showed that WEPP provided better results in the estimation of sediment yield than SWAT. Bhuyan et al. (2002) published similar results in the comparison of WEPP and EPIC being a parent model of SWAT and thus again based on a modified version of the USLE. Additionally, the USLE approach is not suitable for steep slopes, which are typically found in forests (Elliot, 2004). In contrast, estimates of WEPP in forest areas of runoff and erosion, in particular on disturbed land, have been used accurately in the USA (Elliot et al., 1995).

According to Croke and Nethery (2006), WEPP and TOPOG generated good results of runoff and soil loss in their application at a hillslope scale on disturbed surfaces. The performance of both models underestimated sediment yield when they were used on less disturbed surfaces such as general harvesting areas. Beyond that, high complexity and huge data requirements of these physically based models reduce their usability (Croke and Nethery, 2006). The InVEST model is aimed at the mapping of a series of ecosystem services across

[2] Universal Soil Loss Equation

a landscape. This is done in a general way, using a simple kind of model (Tier 1) or a more complex version (Tier 2) (Tallis and Polasky, 2009). It is ideally applied to simulate impacts of land cover changes on multiple ecosystem services over a relatively large basin. Hydrological properties are included in the model on an average annual basis and based on simplifications (using USLE) representing its major limitations (Vigerstol and Aukema, 2011).

Due to the strengths and weaknesses of each of the models presented in this chapter, this thesis applies WEPP showing an exceedingly high user-friendliness, the capability to model runoff and erosion and a good performance in contrast to other models. The WEPP model is presented in the following section more in detail.

3.1.3.2 The Water Erosion Prediction Project

The Water Erosion Prediction Project is a continuous, physically-based erosion prediction model, which was developed by the Department of Agricultural Research, the Natural Resources Conservation Service, the Forest Service, the US Department of Interior's Bureau of Land Management and the US Geological Survey (Elliot, 2004). Originally, this model was meant to replace the formerly used USLE using the description of physical processes instead of empirical values and therefore was initiated in 1985 (Foster and Lane, 1987). WEPP includes the process of infiltration and runoff, soil detachment, transport and deposition, plant growth, senescence and residue decomposition. It calculates soil water content for every single layer, plant growth and residue composition on a daily basis. In addition, effects of tillage through which also forest management practices can be represented and soil consolidation is integrated in the model. The soil cover is not directly implemented in the model and thus depends on plant growth parameters, such as the biomass energy conversion ratio, daily temperatures and the availability of soil water (Elliot, 2004). It is calculated every day by running decomposition and increase of surface cover from plant senescence (Stott et al., 1995). The WEPP model includes a run for hillslopes or for watersheds integrating unique combinations of uniform soil properties and vegetation conditions as an overland flow element (OFE). An exemplary representation of a slope characterised by three overland flow elements can be found in figure 13.

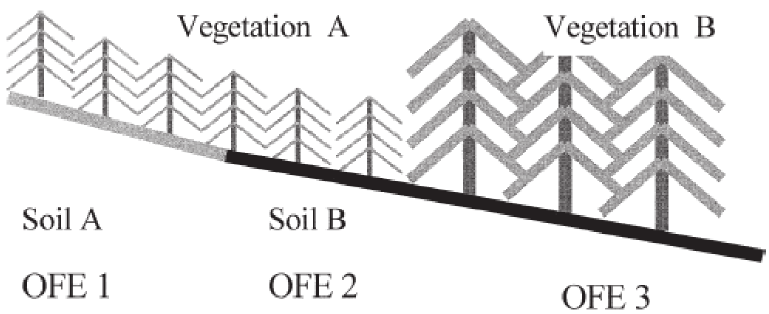

Figure 13 Examples of possible overland flow elements which can be implemented in WEPP (Elliot, 2005)

Additionally, the watershed option provides a link of hillslope elements with channel and impoundment elements (Elliot, 2004). WEPP characterises a watershed as one or more hillslopes draining into one or more channels/impoundments. The smallest possible unit can be comprised of one hillslope and one channel. For watershed routing, WEPP firstly calculates runoff characteristics, soil loss and deposition on each hillslope for the entire simulation period (Flanagan and Livingston, 1995). Each hillslope element is therefore represented as a rectangle (Shen et al., 2009). Next, results generated for each hillslope are combined and runoff and sediment routing is performed though the channels and impoundments each time runoff is produced on one of the elements. The channel and impoundment parameters are therefore updated on a daily basis (Flanagan and Livingston, 1995).

The WEPP model simulates soil loss and sediment deposition from overland flow on hillslopes, soil loss and sediment deposition from concentrated flow in small channels as well as sediment deposition in impoundments.

These processes are based on an integrated climate component using a stochastic weather generator, hydrologic components being are generated using a modified Green-Ampt-infiltration and kinematic wave equations, as well as a water balance, a plant growth and a residue decomposition component. As a result, soil loss and deposition are computed spatially and temporally distributed in order to allow the user to decide on which management practice fits best according to site specific properties (Flanagan and Nearing, 1995). The WEPP model is suitable for climates with annual precipitation values from under 250 to over 2,500 mm, to slopes in the range of research plots of 0.5 m length to hillslopes longer than 500 m and to any soil including cropland, rangeland, forest, road and construction sites (Elliot, 2004). It was developed for the application of predicting long-term average annual values at various sites without the necessity of model calibration (Flanagan et al., 2007).

3.1.3.2.1 Model Description

Figure 14 illustrates an outline of the elements of the WEPP model. It shows required inputs, implemented processes and outputs of WEPP.

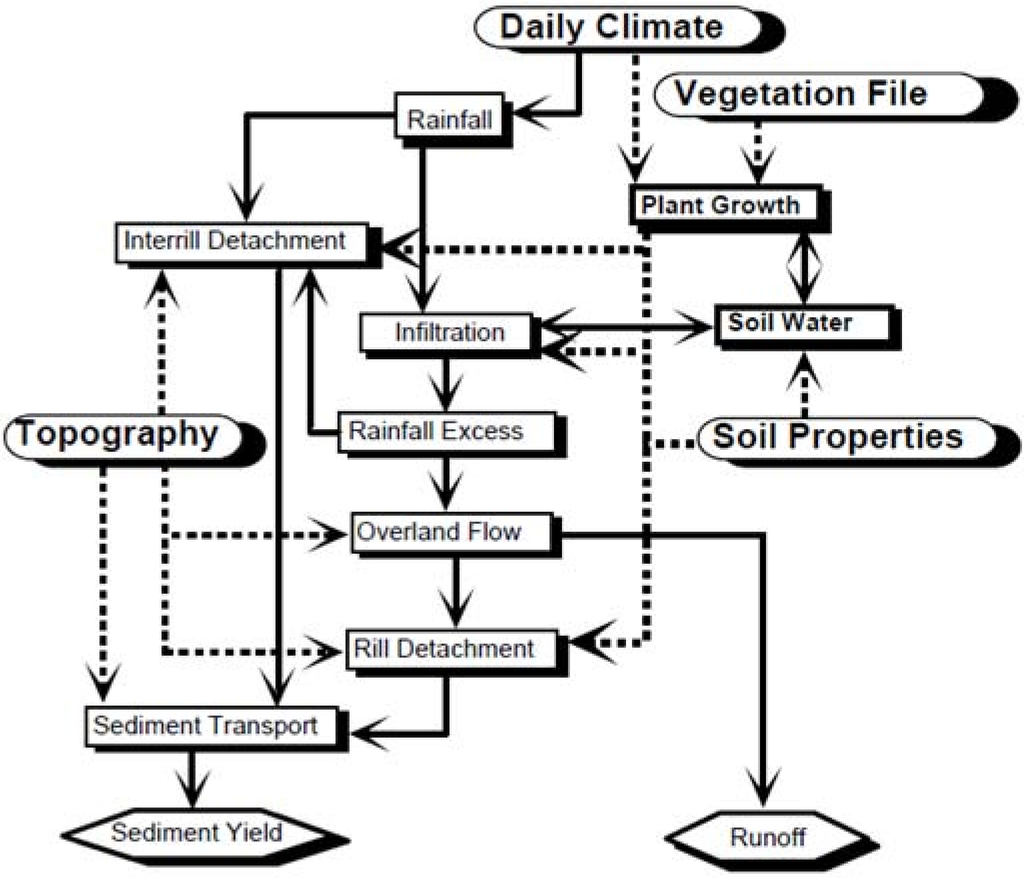

Figure 14 Outline of the WEPP model (Elliot et al., 1995)

The hillslope elements consist of the components weather generation, winter processes, irrigation, surface hydrology and water balance, subsurface hydrology, soils, plant growth, residue decomposition, overland-flow hydraulics and erosion. For weather generation WEPP offers an auxiliary climate generator, CLINGEN generating daily or single storm climate. The calculation of surface hydrology and water balance maintains a continuous balance for soil water content on a daily basis. These components are based on information concerning weather, vegetation and management practice (Pieri et al., 2007).

In the WEPP model, a hillslope is conceptualised as a rectangular strip with a slope profile and multiple OFEs (Cochrane and Flanagan, 2003).

Infiltration

Infiltration is estimated using a modification of the Green-Ampt-Mein-Larson equation (Mein und Larson, 1973) being adapted to unsteady rainfall by a ponding time calculation (Chu, 1978); (Shen et al., 2009):

$$f_{inf,t} = K_e \left(1 + \frac{\Psi_{wf} \Delta\theta_v}{F_{inf,t}}\right) \tag{2}$$

$f_{inf,t}$	(mm h^{-1})	infiltration rate at time t
K_e	(mm h^{-1})	effective hydraulic conductivity
Ψ_{wf}	(mm)	wetting front matric potential
$\Delta\theta_v$		change in volumetric moisture content across the wetting front
$F_{inf,t}$	(mm)	cumulative infiltration at time t

The process of infiltration is divided into a stage in which the ground surface is ponded and a stage without ponding. During an unsteady rainfall, infiltration can switch from one stage to another and the other way around. In case of surface ponding, the process of infiltration is not dependent on the temporal distribution of rainfall. At this point maximum infiltration capacity is reached. Thus, rainfall excess is computed as the difference between rainfall rate and infiltration capacity in this stage. WEPP also accounts for depression storage. Without surface ponding rainfall can infiltrate into the soil where infiltration rate equals rainfall intensity. As rainfall intensity is less than infiltration capacity, rainfall excess is zero (Stone et al., 1995).

Rainfall Excess

Rainfall excess is implemented in the WEPP model as the difference between rainfall rate and infiltration rate (Dun et al., 2009). The volume of rainfall excess is decreased in order to account for depression storage. In addition, runoff is expected to begin only after satisfaction of depression storage. In case of runoff events producing a partial equilibrium hydrograph, rainfall excess is reduced in order to allow for water infiltrating during the recession of an event. Next, the kinematic wave model or the value of the peak discharge derived from the approximate method transforms the time intensity distribution of rainfall excess into a time intensity distribution of runoff. Before the computation of the rate of rainfall excess its volume is adjusted for soil saturated conditions and depression storage (Stone et al., 1995). Thus, the rainfall excess volume is computed in combination with the calculation of infiltration as

$$V_i = R_i - F_i \quad for\ r_i > f_i \quad and\ F_i < S_p \tag{3}$$

$$V_i = V_{i-1} \quad for\ r_i \leq f_i \quad and\ F_i < S_p$$

$$V_i = R_i \quad for\ F_i \geq S_p$$

F_i	m	cumulative infiltration depth
f_i	$m.s^{-1}$	infiltration rate
V_i	m	cumulative rainfall excess depth
R_i	m	cumulative rainfall depth
r_i	$m.s^{-1}$	rainfall rate at time t
S_p	m	upper limit of water storage in the top two soil layers

In addition, the storage in the upper limit is computed as (Stone et al., 1995)

$$S_p = K_{\min D_r} + max\{0, \sum_{j=1}^{2} UL_j - ST_j\} \tag{4}$$

K_{min}	$m.s^{-1}$	minimum saturated hydraulic conductivity of the two layers
D_r	s	duration of the rainfall
UL_j	m	upper limit of soil moisture storage
ST_j	m	current soil moisture storage

The remainder of the rainfall becomes rainfall excess as soon as cumulative infiltration during a rainfall event exceeds the upper limit of water storage in the top two soil layers (S_p). The volume of rainfall excess as computed in equation 2 is decreased by the amount of depression storage. This amount is assumed as the portion of rainfall excess being held by micro-variations in the topography and the other portion being able to infiltrate into the soil. Dependant on the microrelief of the surface, the impact of depression storage on runoff amounts and rates can be significantly (Stone et al., 1995). Therefore, the adjustment for depression storage is calculated after infiltration as (Stone et al., 1995)

$$\mathrm{V_i} = 0 \quad \text{for } \mathrm{V_i} \leq \mathrm{S_d} \tag{5}$$

$$\mathrm{V_i} = \mathrm{V_i} - \mathrm{S_d} \quad \text{for } \mathrm{V_i} > \mathrm{S_d}$$

S_d	m	maximum depression storage

Additionally, there are three basic conditions being considered by the depression storage component. They depend on the number of discrete rainfall excess bursts, the satisfaction of potential depression storage and on the case of infiltration of the volume of water retained by depression storage before the next burst of rainfall excess (Stone et al., 1995).

After adjustment of the rainfall excess volume the average rainfall excess rate, v_i (m.s^{-1}) for an interval is calculated by (Stone et al., 1995)

$$v_i = \frac{V_i - V_{i-1}}{t_i - t_{i-1}} \tag{6}$$

Peak Discharge

Runoff is calculated in the model using kinematic wave equations and an approximation to the kinematic wave solutions (Shen et al., 2009). A semi-analytical solution of the kinematic wave model computes the runoff hydrograph in case of single storm mode for a single OFE. For most events run in the continuous simulation mode, it applies an approximation based on the kinematic wave model for peak discharge. The dynamic infiltration-hydrograph models for overland flow include two functions:

- Infiltration function calculating the infiltration rate varying with time in case of unsteady rainfall
- Routing function transforming rainfall excess into flow depths on the flow surface

According to Stone et al. (1995) the infiltration function is chosen arbitrarily. In contrast, the routing function is a form of the St. Vennant shallow water equations. Therefore the kinematic wave model represents a valid approximation for most overland flow cases (Woolhiser and Liggett, 1967).

In the WEPP model the volume of rainfall excess is adjusted to consider interception in the canopy and residue as well as surface depression storage (Dun et al., 2009; Shen et al., 2009).

The kinematic wave model calculates flow on a plane with the continuity equation

$$\frac{\partial h}{\partial t} + \frac{\partial q}{\partial x} = v \tag{7}$$

And the following depth-discharge relationship

$$q = \alpha\, h^m \tag{8}$$

h	m	depth of flow
q	m³.m^{-1}.s^{-1}	discharge per unit width of the plane
α	m^{0,5}.s^{-1}	depth-discharge coefficient
m	1,5	depth-discharge exponent
x	m	distance from top of plane

WEPP uses the Chezy relationship for overland flow routing which is

$$\alpha = C\, S_o^{0.5} \tag{9}$$

C	$m^{0.5}.s^{-1}$	Chezy coefficient
S_o	$m.m^{-1}$	slope of the flow surface

The initial and boundary conditions are represented by

$$h(x,0) = h(0,t) = 0 \tag{10}$$

and solved using the method of characteristics (Stone et al., 1995).

The approximation of the kinematic wave model was developed in order to decrease the execution time of the program. It is based on the relationship among the time to kinematic equilibrium, the duration of rainfall excess, the peak rainfall excess rate and the average rainfall excess rate differentiating between constant and variable rainfall excess (Stone et al., 1995).

In order to account for a significant lower amount of runoff compared to the rainfall excess volume in case of partial equilibrium, WEPP computes recession infiltration. As there is no interaction between the infiltration and routing equations, rainfall excess is routed as if the flow surface was not permeable. As a result, WEPP applies a relationship between the final infiltration rate, total rainfall excess volume and average rainfall excess rate in order to consider infiltration during the recession of the hydrograph by defining the following quantities

$$Q^* = \frac{Q_v}{V_t}$$
$$f^* = \frac{f_f}{v_a} \tag{11}$$

Q_v	m	adjusted runoff depth
f_f	$m.s^{-1}$	final infiltration rate

The reduction of volume is computed as

$$Q^* = \frac{1}{m+1}\frac{f_x+1}{f_x}t^{*-m} \quad \text{for} \quad t^* \geq \left(\frac{f^*+1}{f^*}\right)^{1/m} \tag{12}$$

and

$$Q^* = 1 - \frac{m}{m+1}\left(\frac{f^*}{f^*+1}\right)^{1/m} t^* \quad \text{for} \quad t^* < \left(\frac{f^*+1}{f^*}\right)^{1/m} \tag{13}$$

at the last time where the rainfall excess rate is not zero (Stone et al., 1995).

As a basis for erosion computations, the WEPP model uses the steady-state sediment continuity equation. Thus, the steady-state runoff discharge rate equals the peak discharge rate which was computed by the kinematic routing or the approximate method. Using this assumption, the computed runoff duration would not sustain continuity between runoff volume and peak discharge rate. As a result, the WEPP surface hydrology component uses an effective duration, *De (s)* being applied in the rill erosion computations:

$$D_e = \frac{Q_v}{q_p} \tag{14}$$

q_p m.s^{-1} peak discharge

The overland flow rate on an OFE is estimated from the average rainfall excess of the upstream OFES weighted by their lengths, as they contribute directly to this flow. Additionally, redistributed infiltration, including evapotranspiration (ET), percolation, subsurface lateral flow of each OFE are taken into consideration on a daily time step. The transmission of subsurface flow is simulated within two adjacent OFEs (Dun et al., 2009).

Water Balance Model Documentation

The WEPP water balance and percolation modules estimate soil water content in the root zone as well as evapotranspiration losses during the simulation period by resorting to the climate, infiltration and crop growth components. WEPP predicts evapotranspiration and percolation in a time step of 24 hours. The water balance is therefore calculated using many algorithms which were developed for the Simulator for Water Resources in Rural Basins (SWRRB) model published by Williams et al. (1985). In addition, these algorithms were modified in order to improve the estimation of rainfall interception, percolation and soil evaporation parameters (Savabi and Williams, 1995).

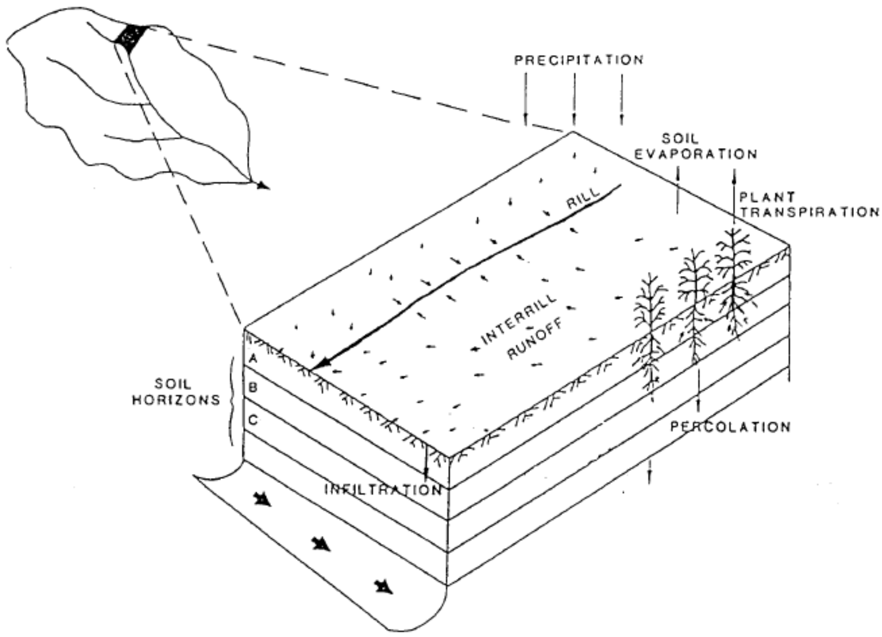

Figure 15 Relevant processes on which the WEPP hillslope hydrology depends: precipitation (rain/snow), infiltration, runoff, soil evaporation, percolation and plant transpiration (Savabi and Williams, 1995)

Figure 15 illustrates the hydrologic processes integrated in the WEPP model. It maintains a continuous water balance which can be expressed as

$$\theta = \theta_{in} + (P - I) \pm S - Q - ET - D - Q_d \qquad (15)$$

θ	m	soil water content in the root zone in any given day	
θ_{in}	m	initial soil water in the root zone	
P	m	cumulative precipitation	
I	m	precipitation interception by vegetation	
S	m	soil water content:	(+) snow melt, equals daily snowmelt
			(-) snow accumulation
Q	m	cumulative amount of surface runoff	
ET	m	cumulative amount of evapotranspiration	
D	m	cumulative amount of percolation loss below the root zone	
Q_d	m	subsurface lateral flow or flow to drain tiles	

(Savabi and Williams, 1995)

The interception of precipitation by the vegetation is computed by applying the method of Savabi and Stott (1994):

$$I = .001\,(3.7\,VE - 1.10\,x\,10^{-4} VE^2) \qquad (16)$$

VE	$kg.m^{-2}$	above ground biomass
I	m	interception

WEPP separates precipitation into rainfall and snowfall using air temperature. For days having a maximum temperature of below 0 °C, all precipitation is considered as snow, whereas for days with a minimum temperature of above 0 °C, precipitation is assumed to be rain. For days on which the maximum and minimum temperature is above 0 °C a constructed diurnal temperature function determines whether in a particular period snow or rain falls by using a randomly prediction of the time of precipitation occurrence.

An accumulated snowpack experiences evaporation and melt. Thus, soil evaporation is assumed to first occur on the snowpack in case of its presence. Afterwards it is considered to come directly from the soil. When the maximum temperature of a day exceeds 0 °C, snow is melted being treated in the water balance as rainfall in order to estimate percolation and runoff (Savabi and Williams, 1995).

Evapotranspiration

Generally spoken, the evapotranspiration (ET) component of WEPP represents a modification of the Ritchie's model (Ritchie, 1972). Beyond that, two options are provided by the model to estimate the potential ET dependant on availability of meteorological data.

The WEPP model uses the Penman equation including the original wind function method (Penman, 1963) if daily radiation, temperature, wind and dew point temperature or relative humidity are accessible.

$$E_u = \frac{\delta}{\delta+\gamma}(R_n - G) + \frac{\gamma}{\gamma-\delta} 6.43\,(1.0 + 0.53\,u_z)(e_z^0 - e_z) \qquad (17)$$

E_u	$MJ.m^{-2}.d^{-1}$	daily potential evapotranspiration
δ		slope of the saturated vapour pressure
γ		psychrometric constant
G	$MJ.m^{-2}.d^{-1}$	soil heat flux
R_n	$MJ.m^{-2}.d^{-1}$	net radiation
u_z	$m.s^{-1}$	wind speed
e_z^0	kPa	saturated vapour pressure
e_z	kPa	vapour pressure

If only solar radiation and temperature data are present, WEPP applies the Priestley and Tailor (1972) method.

$$E_u = 0.00128\,\frac{R_n l}{58.3}\frac{\delta}{\delta+\gamma} \tag{18}$$

E_u	$MJ.m^{-2}.d^{-1}$	daily potential evapotranspiration
$R_n l$	ly	daily net solar radiation
δ		slope of the saturated vapour pressure
γ		psychrometric constant

Net radiation in equation 16 and 17 are calculated by a multiplication of the incoming daily solar radiation by (1 - A), where A is the albedo lying between 0 and 1.0 (Savabi and Williams, 1995).

Soil water is extracted if residue and plant rainfall interception do not arrive at the potential evapotranspiration. Therefore the potential soil evaporation is a part of the potential ET dependent on the fraction of uncovered soil (Pieri et al., 2007). Potential soil evaporation, E_{sp}, and plant transpiration, E_{tp}, are calculated using the following equations (Savabi and Williams, 1995):

$$E_{sp} = E_u e^{(-0.4\,L)}$$

$$E_{tp} = \left(1 - \frac{E_{sp}}{E_u}\right) * E_u \tag{19}$$

L		leaf area index
E_u	$MJ.m^{-2}.d^{-1}$	daily potential evapotranspiration
E_{tp}	$m.d^{-1}$	potential plant transpiration

Figure 16 shows the sequence which WEPP uses to calculate evapotranspiration and soil water redistribution.

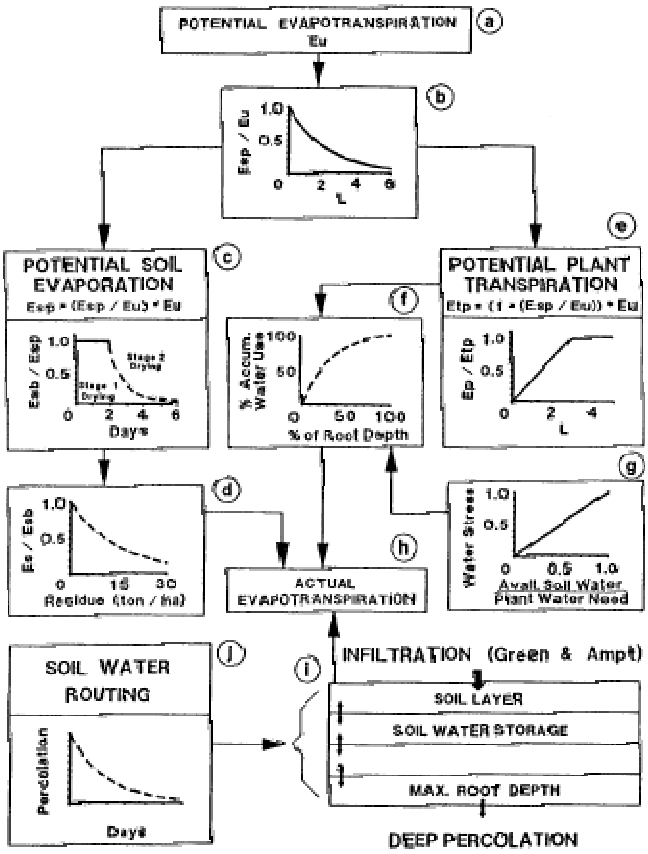

Figure 16 Schematic of the computational sequence of evapotranspiration and soil water redistribution in WEPP. E_u represents daily potential evapotranspiration (Savabi and Williams, 1995)

If the plant leaf area index (LAI) is larger than three, WEPP calculates ET only from plant transpiration. This is done by assuming potential plant transpiration to be a fraction of total potential ET of one third of the plant LAI (Pieri et al., 2007).

Percolation

WEPP calculates percolation using storage routing techniques in order to predict the flow through each soil layer in the root zone. Additionally, it simulates subsurface lateral flow as well as flow to drainage tile and ditches (Savabi and Williams, 1995).

Percolation (Q_p) is represented in WEPP through the exceedance of field capacity. This is calculated with the unsaturated hydraulic conductivity being an estimate of K_s, h_{fc} and h_{wp}, the currently available water for percolation in the soil layer (ith) and the degree of saturation of the layer below (i + 1th) (Dun et al., 2009).

$$Q_p = vv_i\sqrt{1 - S_{i+1}}\left[1 - \exp\left(-\frac{\Delta t}{t_i}\right)\right] \tag{20}$$

$$vv_i = (\theta_i - \theta_{fci})d_i \tag{21}$$

$$S_{i+t} = \frac{\theta_{i+t} - \theta_{wpi+1}}{\phi_{i+1} - \theta_{wpi+1}} \tag{22}$$

$$t_i = \frac{\theta_i - \theta_{fci}}{K_{ui}} \tag{23}$$

$$K_{ui} = K_{si}S_i^{-2.655 \log\left[\frac{\theta_{fci} - \theta_{wpi}}{\phi_i - \theta_{wpi}}\right]} \tag{24}$$

Δt	(s)	time interval
Q_p	($m\ s^{-1}$)	percolation
S_i	[-]	degree of saturation
θ_i	($m^3\ m^{-3}$)	soil water content
θ_{fci}	($m^3\ m^{-3}$)	field capacity
θ_{wpi}	($m^3\ m^{-3)}$	wilting point
ϕ_i	($m^3\ m^{-3}$)	porosity
d_i	(m)	soil thickness
K_{si}	($m\ s^{-1}$)	saturated hydraulic conductivity
K_{ui}	($m\ s^{-1}$)	unsaturated hydraulic conductivity
vv_i	(m)	available water for percolation
t_i	(s)	travel time of percolating water of the *i*th soil layer

The saturated hydraulic conductivity is mainly determined by soil physical properties such as porosity, soil texture and organic matter. In addition, the flow through a layer of the soil can be decreased by coarse fragments, freezing of the layer of a saturated or nearly saturated lower layer. The water which is moving below the root zone is not traced by WEPP and therefore considered as lost (Savabi and Williams, 1995).

Deep Percolation

Deep percolation being estimated following equ. (19) is characterised by percolation through the lowest layer and leaves the model domain except the degree of saturation for the media below the last layer is set to zero (Dun et al., 2009).

In general, the WEPP models links the evapotranspiration and percolation components with the infiltration module in order to maintain a continuous water balance. It adds the infiltrated water to the water content of the upper soil layer and routes it through the lower soil layers. Soil water in each layer is determined by percolation and/or evapotranspiration. The establishment of initial moisture conditions for the infiltration component is dependent on the upper layer soil water content. Additionally, the crop growth component adds daily leaf area index, root depth, total plant biomass and residue cover as input to the evapotranspiration component (Savabi and Williams, 1995).

Subsurface Lateral Flow

Subsurface lateral flow is calculated by the WEPP model when soil water content in a layer is larger than its drainable threshold. This drainable threshold is defined as the field capacity corrected for entrapped air. Lateral flow routines are estimated following Darcy's law using unsaturated hydraulic conductivity of the draining layer and the average surface gradient of the OFE (Dun et al., 2009).

$$R_s = K_l S_p \frac{D_d}{L} \tag{25}$$

$$D_d = \sum d_i \tag{26}$$

$$K_l = \frac{\sum(d_i K_{ui})}{D_d} \tag{27}$$

R_s	(m s^{-1})	subsurface lateral flow
K_l	(m s^{-1})	equivalent lateral hydraulic conductivity
S_p	(m m^{-1})	average slope gradient of the OFE
D_d	(m)	total thickness of the drainable layers
L	(m)	slope length of the OFE

In 2005, the WEPP model subsurface routines were changed in order to account for its problem of overestimation of deep percolation. Therefore, three additional parameters were integrated into the soil input file. Thus, it is now possible to provide information for a restricting layer located at the bottom of the soil profile. An anisotropy ratio of the soil profile and the name or the vertical saturated hydraulic conductivity of the bedrock were integrated into the model as a flag variable. As a result, the user can choose whether or not to use a restricting layer. Additionally, the anisotropy ratio enables the user to consider the relative predominance of lateral versus vertical flow. The WEPP package provides reference values for the anisotropy ratios for general types of bedrock such as sedimentary and crystalline rocks (Wu and Dun, 2005).

WEPP assumes that subsurface runoff is essentially clear and contains no sediment due to its slow rate and undergoing natural filtration processes. If both surface runoff and subsurface runoff are occurring, WEPP assumes surface runoff to dominate water flow and sediment transport processes. Thus, the subsurface runoff is added to the surface runoff. WEPP neglects additional erosion arising due to subsurface runoff, which is assumed to be adequate according to Wu and Dun (2005).

If only subsurface runoff occurs WEPP assumes an event to last 24 hours and records it in the hillslope pass file. Additionally, subsurface runoff occurring when there is no storm is added in the modified WEPP model to the channel flow, increasing the transport capacity of the channel and potential channel erosion (Wu and Dun, 2005).

WEPP includes subsurface lateral flow from the upland OFE as well as soil water input to the current OFE. The daily soil water redistribution is firstly estimated by percolation followed by soil evaporation, subsurface lateral flow, saturation-excess and plant transpiration on a daily basis. Soil water content is then updated after each calculation. Soil water excess is computed by a comparison of soil water content against porosity from the bottom to the top of each layer. An excess of water is then passed to the layer lying above. In case of an exceedance of porosity by soil water content in the upper layer, surface runoff accrues owing to saturation excess (Dun et al., 2009).

Plant Growth Component

The plant growth component of the WEPP model is aimed at the simulation of temporal changes in plant variables influencing runoff and erosion. It is generally based on the EPIC model (Williams et al., 1984). Thus, it predicts biomass accumulation as a function of heat unity and photosynthetically active radiation. Its growth rate is decreased by moisture and temperature stress. The plant growth component of WEPP computes canopy cover and height, growing degree days, root growth, leaf area index, plant basal area, mass of vegetative dry matter (Flanagan et al., 1995).

Soil Parameters

The soil parameters integrated into the WEPP model include:

- *Random roughness*: the decay of random roughness following a tillage activity is calculated from a relationship between a random roughness parameter and the cumulative rainfall since last tillage
- *Oriented roughness*: represents the height of ridges left by tillage implements. The ridge decay is then based upon a relationship including a ridge height parameter and the cumulative rainfall since last tillage
- *Bulk density*: used to determine several variables related to infiltration such as the wetting front suction. It is adjusted according to tillage operations, rainfall consolidation, weathering consolidation and soil water content.
- *Wetting-front suction*
- *Hydraulic conductivity*: represents a key parameter in the model controlling the prediction of runoff and infiltration.
- *Interrill erodibility*: measure of soil resistance to detachment by raindrop impact
- *Rill erodibility:* measure of the soil resistance to detachment by concentrated rill flow. It can also be defined as the increase in soil detachment per unit increase in shear stress of the flow.
- *Critical shear stress*: threshold parameter representing a value above which a rapid increase in soil detachment per unit increase in shear stress occurs

Hillslope Erosion and Deposition

WEPP integrates watershed sediment yield resulting from detachment, transport and deposition of sediment from hillslope and channel areas. Therefore, a steady-state erosion model being based on the movement of suspended sediment on rill, interrill and channel flow areas is applied (Shen et al., 2009). Interrill erosion is treated in the model as soil detachment by raindrop impact and transport by sheet flow on interrill areas (Flanagan et al., 1995). Rill erosion is represented as a function of sediment detachment due to excess flow shear stress from channel bed and bank in combination with the transport capacity of concentrated flow and the load being already present in the flow (Pieri et al, 2007; Dun et al., 2009).

The steady state erosion model solves the sediment continuity equation at peak runoff rate. It is described as follows (Shen et al., 2009):

$$\frac{dG}{dX} = D_f + D_i \tag{28}$$

Where $D_i > 0$ and independent of x
$D_f > 0$ for detachment
$D_f < 0$ for deposition

G	(kg s^{-1}m-1)	sediment load
X	(m)	distance downslope
D_f	(kg $s^{-1}m^{-1}$)	rill erosion rate
D_i	(kg $s^{-1}m^{-1}$)	sediment delivery to the rill

D_f and D_i are calculated on a per rill area basis for the computations. As a result, G is solved on a per unit rill width basis. Sediment yield is shown as sediment yield per unit land area in the model (Shen et al., 2009).

WEPP models the interrill delivery rate as being proportional to the product of interrill runoff rate and rainfall intensity. The equation used to describe the interrill delivery rate includes parameters accounting for soil roughness, slope steepness and adjusted soil erodibility on interrill detachment and transport. Additionally, the detachment arising owing to rainfall during periods when infiltration capacity is greater than rainfall intensity does not contribute to interrill detachment (Flanagan et al., 1995).

The WEPP model calculates net soil detachment in rills for the situation when hydraulic shear stress exceeds the critical shear stress of the soil and sediment load is less than sediment transport capacity as (Foster et al., 1995):

$$D_f = D_c(1 - \frac{G}{T_c}) \tag{29}$$

D_c	kg.s^{-1}.m^{-2}	detachment capacity by rill flow
T_c	kg.s^{-1}.m^{-1}	sediment transport capacity in the rill

The sediment transport capacity is computed as a function of x which is the distance downslope by applying a simplified modification of the equation of Yalin (1963).
In case of exceedance of critical shear stress by hydraulic shear stress of the rill, the detachment capacity, D_c, is expressed as (Foster et al., 1995)

$$D_c = K_r\ (\tau_f - \tau_c) \qquad (30)$$

K_r	$s.m^{-1}$	rill erodibility parameter
τ_f	Pa	flow shear stress acting on the soil particles
τ_c	Pa	rill detachment threshold parameter or critical shear stress of the soil

If flow shear stress is less than the critical shear stress of the soil rill detachment, it is assumed to be 0. Additionally, rill erodibility and critical shear stress are adjusted by WEPP as a function of temporally-varying factors (Foster et al., 1995).
Net deposition in a rill is considered when sediment load*, G*, is higher than the sediment transport capacity, T_c (Foster et al., 1995).

$$D_f = \frac{ß\, V_f}{q}\,(T_c - G) \qquad (31)$$

V_f	$m.s^{-1}$	effective fall velocity for the sediment
q	$m^2.s^{-1}$	flow discharge per unit width
ß		raindrop-induced turbulence coefficient
T_c	$kg.s^{-1}.m^{-1}$	sediment transport capacity
G	$kg.s^{-1}.m^{-1}$	sediment load

If rain drops impact rill flows, WEPP assignes ß a value of 0.5. In cases as for example snow melting or furrow irrigation, ß is assumed to be 1.0 (Foster et al., 1995).
In order to drive the erosion model the following four hydrologic variables, which are calculated by the hydrology component are used (Foster et al., 1995):

- Peak runoff, P_r ($m.s^{-1}$)
- Effective runoff duration, t_r (s)
- Effective rainfall intensity, I_e ($m.s^{-1}$)
- Effective rainfall duration, t_e (s)

In order to normalize the erosion equations, conditions at the end of a uniform slope through the endpoints of the given profile are used. In addition, the erosion component uses four dimensionless parameters including one for interrill sediment delivery to rills, two for rill detachment and one for rill deposition. WEPP solves the normalized sediment continuity equation in an analytical way if net deposition occurs. In case of detachment, it is solved numerically (Flanagan et al., 1995).

Winter Routines

Winter routines include soil frost, thaw, snow melt and snow accumulation (Dun et al., 2009). It calculates hourly temperature, radiation and snow fall values based on average daily

values. The subcomponent calculating soil frost is based on fundamental heat flow theory. Additionally, the frost and thaw subcomponent assumes the heat flow in a frozen or unfrozen soil or soil-snow system to be unidirectional. Adjustments for infiltration and erodibility are calculated using the location of frost or thaw in the soil profile and the soil moisture content.
Moreover, the snow accumulation subcomponent computes the depth of snow on the ground on a daily or hourly basis. Warm temperatures and rainfall consolidate the snow pack wherease snow fall increases it (Flanagan et al, 1995). The snow melt component uses a modification of the generalized snow melt equation developed by the U.S. Army Corps of Engineers (1956, 1960) by Hendrick et al. (1971). It includes the four major energy components air temperature, solar radiant, vapour transfer and precipitation. The snow melt calculations of WEPP are based on the following assumptions (Flanagan et al., 1995):

1) Any precipitation is snowfall if it occurs on a day when the maximum daily temperature is below 0 °C
2) No snowmelt occurs if the maximum daily temperature is below -4 °C (Dun and Wu, 2008)
3) The snowpack is not assumed to melt until the density of snow arrives at a value greater than 0.35 g.cm^3
4) During the melt period the surface soil temperature arrives at 0 °C
5) The albedo of melting snow is approximated to be 0.5.

In 2008, Dun and Wu improved the snowmelt routines of WEPP for the v2008.4 publication. Therefore, they adapted the convection-condensation term for wind velocities to zero as they play an important role in snowmelt patterns in heavily forested areas. As a result, they incorporated their convection-condensation equation for heavily forested areas and a wind velocity adjustment factor equation into the Hendrick et al. model (Dun and Wu, 2008).

Weather Generation

The climate component used in the WEPP model generates mean daily amount and probability of precipitation, time to peak, maximum, minimum and dew point temperature on a daily basis, mean daily solar radiation and mean daily wind direction and speed. The stochastic weather generator uses monthly parameters such as means, standard deviations and skewness derived from historic measurements (Meyer, s.a.). Moreover, the distribution and number of precipitation events are the result of the usage of a two-state Markov chain model. Therefore, the model determines stochastically if precipitation occurs on the current day under the initial condition that the previous day was wet or dry. In order to decide if a precipitation event occurs, a random number lying between 0 and 1, is generated and compared with the appropiated wet-dry-probability. Precipitation occurs if the random number is lower than or equal the wet-dry-probability. As a result, the amount of precipitation of the occuring event is determined using a skewed normal distribution function. Additionally, rainfall duration for single events is calculated applying an exponential distribution using monthly mean durations (Flanagan et al., 1995).

WEPP differentiates between rainfall and snowfall using daily air temperatures. Daily maximum as well as minimum temperatures and solar radiation are derived from normal distribution functions. Furthermore, the climate component uses a disaggregation model to determine time-rainfall intensity (breakpoint) data based on daily rainfall amounts (Flanagan et al., 1995). The random number generator simulates identical results when subsequent runs on the same machine with identical inputs are generated.

One big disadvantage of the generate climate patterns lies in the independence of simulated precipitation from solar radiation and maximum and minimum temperatures being in reality linked to each other (Meyer, s.a.). According to Meyer (s.a.) this is not a big issue in practice, since the monthly trend is preserved certainly.

3.1.3.2.2 Adaptation of WEPP to Forest Applications

Originally, the WEPP model was developed for the application in agriculture, rangelands and subsequently in forests. According to Elliot et al. (1999), Elliot and Tysdal (1999), Elliot (2004), Robichaud et al. (2007) and Dun et al. (2009) WEPP was successfully applied in areas where Hortonian flow is present such as forest applications modelling erosion from forest roads or harvested or burned areas especially by fire. Nevertheless, in most forested areas subsurface lateral flow and channel flow processes are predominant where WEPP underestimates subsurface lateral flow due to its over prediction of deep percolation and the discharge at the outlet of the watershed (Dun et al., 2008).

In order to model agricultural applications, the WEPP model uses a modified Green-Ampt infiltration model to generate runoff. Recently, Elliot et al. (1996) and Covert (2003) recognised that rainfall-excess does not represent the only mechanism generating runoff and therefore affects forest hydrology as well as erosion. Due to the fact, that forest areas are mostly characterised by steep slopes and coarse-grained, young and shallow soils, they need to be distinguished from agricultural soils.

In addition, canopy and residues covers which markedly differ between forests and rangelands or cropping systems leading to big differences in the hydrologic processes as illustrated in figure 17 (Dun et al., 2009).

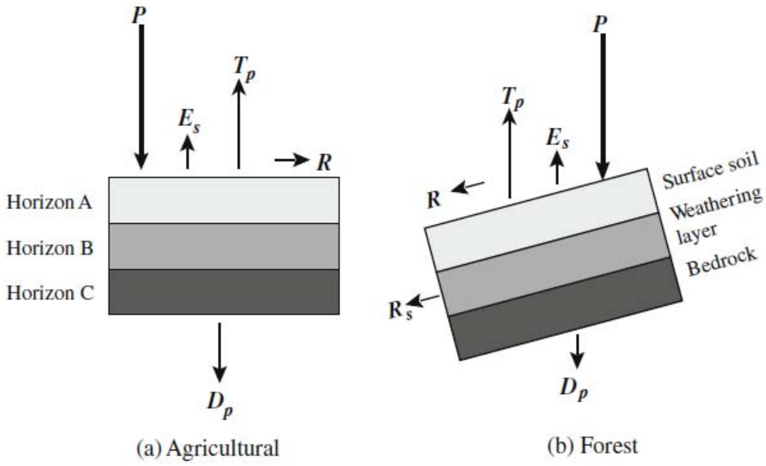

Figure 17 Differences in the hydrologic processes of a) agricultural managed areas and b) forested areas. The relative magnitude or rate of individual processes are represented by the size of the arrows: precipitation (P), plant transpiration (T_p), soil evaporation (E_s), surface runoff (R), subsurface lateral flow (R_s) and percolation through the bottom of the soil profile (D_p) (Dun et al., 2009)

Owing to these differences in the hydrological processes, a modification of the WEPP model was necessary in order to represent forest hydrology properly (Dun et al., 2009). Especially lateral flow processes need to be adequately calculated by the model in order to be able to simulate runoff and erosion on disturbed forest catchments. Therefore Dun et al. (2009) tried to improve the WEPP (v2004.7) subsurface lateral flow routines in such a way that it can be used for forested watersheds to predict runoff and erosion. These adaptations are already included in the description of the subsurface lateral flow in the previous section.

In addition, the input files of the WEPP model do not sufficiently describe the huge variations, which can be found in the hydrologic properties of forested hillslopes. In forested areas saturation overland flow is most common and thus produces overland flow. The WEPP model is not capable of modelling saturation overland flow. In addition, it predicts increased runoff based on an increase in soil water content without reducing infiltration to zero under saturated conditions. As a result, saturated overland flow is not described in an accurate way (Elliot et al., 1995). Due to the fact that forested areas show a large spatial variability in soil hydraulic properties, infiltration capacity and surface erodibility the prediction of runoff generating mechanism is rendered difficult. Despite all these difficulties, the WEPP has been applied to model runoff and erosion with a satisfying accuracy in forested catchments in the USA (Croke 2006).

3.2 Economic Assessment

The economic assessment of ecosystem services offers a high variability of differing approaches of valuation. This chapter aims at the provision of a basic overview concerning frequently applied techniques in the field of valuation of ecosystem services. In addition, it should point out strengths and weaknesses of various methods. Furthermore, it describes the process of model selection in the course of this thesis, provides a more detailed view on the methodologies being applied and discusses the drawbacks of the economic valuation of ecosystem services.

3.2.1 Valuation Methods for Ecosystem Services

Valuing ecosystems for decision-making offers a high diversity of valuation technologies based on different disciplines, philosophical views and schools of thought. So far, lots of methods have been applied, trying to quantify the benefits offered by ecosystem services and using utilitarian approaches. Valuation techniques for ecosystem services are frequently applied for provisioning services. Recently, the improvement of the ability to evaluate regulating, supporting and cultural services has also gained importance. Generally spoken, an appropriately chosen valuation technique needs to be perfectly adapted to the characteristics and availability of data of each case (MA, 2003).
Applied welfare economics assume that individuals know about their preferences over alternative bundles of goods contributing to human well-being. Changes in ecosystem management and their impacts on individual welfare are dependent on the opportunities for, as well as costs of mitigating or defending actions. Thus, estimates of benefits done by economists need to integrate available averting and mitigating activities into the analysis. These welfare measures are based on the amount of money being necessary to compensate the change in question, while maintaining a constant level of utility (Bockstael and Freeman, 2005).

In the last 30 years, economic theory was mainly based on valuation methods observing the behavioural change of individuals in their consumption of goods and services that are complements or substitutes to changes in environmental quality. Thus, it was possible to provide monetary estimates of welfare gains from observed behaviour using *revealed preference methods* (Bockstael and Freeman, 2005). This approach can be further divided into direct and indirect revealed preference methods. In application, this approach is preferable over measures based on hypothetical behaviour (MA, 2003). This second valuation approach is represented by *stated preference methods* such as contingent valuation including direct interviews of people, asking about how they would respond to hypothetical changes in environmental quality (Bockstael and Freeman, 2005).

3.2.1.1 Revealed Preference Methods

Revealed preference methods derive the value of an environmental service in focus by the relationship between a market good and the ecosystem service. This is done using models being mostly based on estimates of substitutability or complementarity between environmental services and the market good. These methods include direct and indirect market valuation in form of household production models, travel cost demand models, hedonic property value and hedonic wage models. Thus, it tries to estimate the values individuals place on ecosystem services and their relationship to their responses to prices and other economic signals (Bockstael and Freeman, 2005).

3.2.1.1.1 Direct Market Valuation

Estimates resulting out of direct market valuation are derived from observed behaviour of producers and consumers. It represents the exchange value of ecosystem services in trade using market prices. They are best applicable for services which are privately owned and traded on the market such as 'commodities' (using production functions), but also information functions, such as recreation, or regulation functions. New York City represents a famous example, which decided to place emphasis on natural water regulation services of largely undeveloped watersheds. The purchase of the water protection areas ensured the delivery of safe water and avoided a $6 billion water filtration plant. (MA, 2003; de Groot et al., 2002)

3.2.1.1.2 Indirect Market Valuation

In case of absence of markets for ecosystem services, more indirect methods of assessing economic values are applied. There is a big variety of valuation methods to assess the willingness to pay or willingness to accept compensation for a change in ecosystem services available such as avoided cost, replacement cost, factor income, travel cost and hedonic pricing (de Groot et al., 2002). If there is no observable actual market behaviour of the particular service in focus, these tools use estimates of actual behaviour on surrogate markets (MA, 2003). In general, these methods are very reliable under the condition of a correct specified relation between the benefit being valued and the surrogate market (International Institute for Environment and Development, 2003).

- Avoided Cost Method

Avoided costs are applied for those services, which allow the society to prevent costs that would have aroused in the absence of these services. This method would be applied in case of flood control avoiding property damages, or waste treatment by wetlands, preventing health costs (Kalof and Satterfield, 2005).

- Replacement Cost Method

The replacement cost method values services at the cost of replacing ecosystem services with human-made systems arriving at the same welfare for both situations (Perman et al., 2003; Sekot and Schwarzbauer, 1995). Thus, it represents a measure of the minimum willingness to pay to ensure the conservation of a particular benefit (Binning et al., 1995). As this measure does not exactly reflect welfare based measures of value, it represents a lower bound of the estimated value of the ecosystem service in focus (Perman et al., 2003). For the application of this method, the costs of the replacement goods need to be good substitutes for the original services provided by nature. In addition, the replacement cost method assumes, that the benefit of the replacement of the ecosystem service is higher than the cost accruing due to the replacement (Binning et al., 1995). It is based on the assumption of rational resource allocation being aimed at high efficiency. The alternative in focus of the analysis needs to be motivated by the private sector having a value which is determined by the market. Thus, the usage of a reference value of an inefficient alternative can be prevented (Sekot and Schwarzbauer, 1995).

A weakness of this method is represented by the lack of consideration of societal preferences for ecosystem services, neither is the behaviour of individuals as a response to the absence of those services included. Additionally, it is limited as it represents only the value of the services replaced, which could only be a small part of the services provided by the ecosystem. As a result, the benefits of the conservation of the properties of the ecosystem would be understated (King and Mazzotta, s.a.).

Frequent examples of this method would be the replacement of a water purification service provided by an ecosystem with a water treatment plant (Perman et al., 2003; King and Mazzotta, s.a.). This thesis applies this method for the ecosystem service of water supply using a rather similar approach.

In this context, it is necessary to distinguish between private and public alternatives. As the Vienna Water works are part of the Viennese city administration, they need to include societal preferences into their decision making. Therefore, this analysis cannot derive the value of the ecosystem service of water supply from the costs of replacement directly, as the public management of the water protection area does not inevitably need to be efficient.

Thus, a comparison of the cost-effectiveness of various land use alternatives can provide more suitable information about the value of the ecosystem services (Grottker, 1999).

- Production Function / Factor Income Method

The factor income method includes the enhancement of incomes by ecosystem services. One example would be the case of natural water quality improvements, resulting in an increase of commercial fisheries catch combined with higher incomes of fishermen (de Groot et al., 2002). The estimate of change in production bases on the physical relation between the level of a non-market benefit and the level of output of a marketed good or service. Therefore, good data on biophysical relationships (dose-response) are required to estimate on- as well as off-site effects of land use change (International Institute for Environment and Development, 2003).

- Travel Cost Method

The travel cost method is based on the travel costs people are willing to pay for visiting environmental amenities. These costs are regarded as a reflection of the implied value of the service. One well-known example is recreation in areas attracting distant visitors. A lower bound of the value of this environmental benign area is represented by the travel costs of visitors (de Groot et al., 2002). In this case, weak complementarity is applied using the assumption that the individual would not visit the park if he or she did not care about that services provided by it (Perman et al., 2003).

- Hedonic Pricing

Hedonic Pricing implies that the demand for a service is reflected in the prices people pay for associated goods. This approach was first used in the context of atmospheric pollution based on the influence of clean air, which is not a traded good, on residential property prices. It suggests that, if all other factors remain constant, a positive relationship can be detected between the willingness to pay for housing and the quality of ambient air. The analysis of property prices can thus render the estimation of the value of clean air possible. This approach also implies the application of weak complementarity (Perman et al., 2003). The application of this methodology requires large data sets in order to enable the researcher to isolate the influence of a non-market benefit relative to other factors (International Institute for Environment and Development, 2003).

3.2.1.2 Contingent Valuation

Contingent valuation represents the most commonly applied direct stated preference method. This technique includes questioning people directly about their willingness to pay for a specified environmental benefit (MA, 2003). The method is called contingent due to the fact that the valuation is contingent on hypothetical scenarios asked to the relevant population. CVM is mainly applied in the analysis of changes in the availability of public goods and environmental commodities being characterised by non-excludability and non-divisibility. The most critical problem in the application of Contingent Valuation is that hypothetical questions are asked to people, whereas indirect methods have the advantage of focussing on observed behaviour. Owing to that fact, the valuation of ecosystems services of more remote ecological systems, of which individuals have little knowledge may yield poor results. On the other hand, CVM has the ability to include use and non-use values into the analysis. Indirect methods just cover use values of ecosystem goods and services. In general, contingent valuation is mostly applied in evaluating non-use, existence or passive-use values (Perman et al., 2003). Scientists show concern about the validity and reliability of the data and additional bias of the results owing to the hypothetical nature of the questions (Bockstael and Freeman, 2005).

An indirect approach of contingent valuation uses contingent ranking as participatory technique. This methodology includes focus groups to elicit preferences for non-market benefits by rankings. As these participatory techniques are not that widespread they are still in an experimental stage concerning reliable estimates of willingness to pay/willingness to accept compensation (International Institute for Environment and Development, 2003).

3.2.1.3 Benefit Transfer

Benefit transfer does not represent a valuation technique for itself. Instead, it implies the use of estimates obtained by any of the methods discussed above. As a result, values derived by a valuation method mentioned above, can be the basis for the estimate of values in a different context (MA, 2003).

3.2.1.4 Application of Valuation Methods

The approaches presented above have been used in a variety of different contexts, including the valuation of cultural benefits (MA, 2003). The following Figure 18 links the total economic value framework with various valuation techniques above. It shows which valuation method suits best for certain types of value.

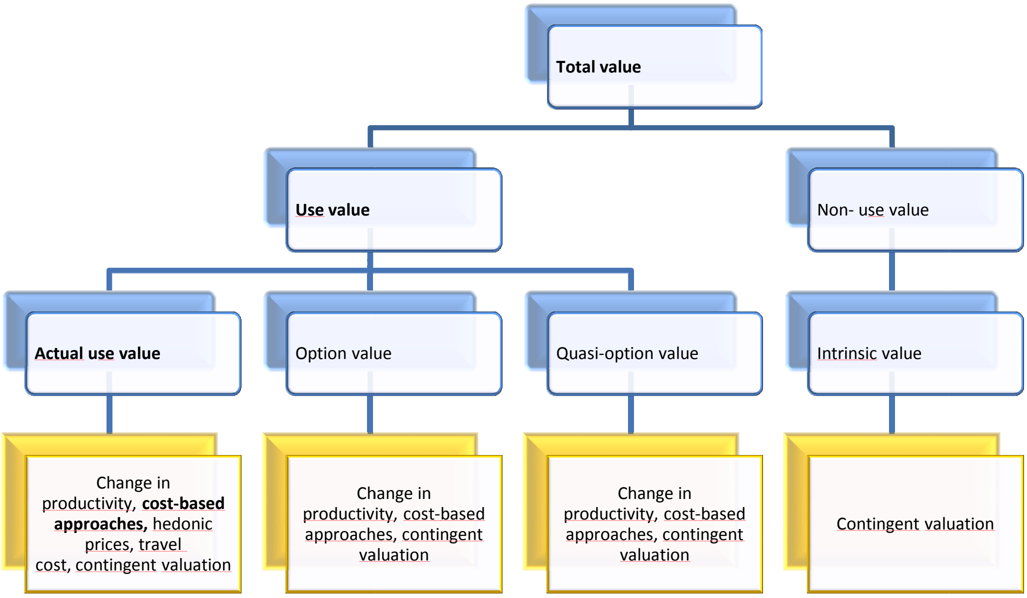

Figure 18 The total economic value framework [blue] including commonly used valuation methods [orange] (adaptation of MEA, 2003)

Each of the methods presented above offers its strengths and weaknesses in differing applications. Table 3 presents an overview published by Careiva et al. (2011) linking ecosystem services with appropriate valuation methods.

Table 3 Ecosystem services and valuation methods (adaptation of Careiva et al., 2011; Bergstrom et al., 2001)

Services	*Types of values offered*	*Valuation method*
Provisioning services		
Subsistence of plant and animals	Direct use values	
	- Consumtive	Direct valuations based on market prices
	- Non-Consumptive	Indirect valuations (revealed expenditure methods, contingent valuation method)
	Indirect use values	(no valuations necessary if plants/animals with direct values are counted)
Regulating Services		
Water filtration, flood control, pest control, pollination of plants, climate stabilisation	Direct and indirect use values	Estimation of service's contribution to profit (holding all else constant), direct valuation based on market prices, indirect measures through travel costs, hedonic pricing
Other services		
Generation of spiritual, aesthetic and cultural satisfaction	Existence value	Indirect valuations (contingent valuation method)
	Non-consumptive direct use value	Indirect valuations (revealed expenditure methods, contingent valuation method)
Recreational services	Non-consumptive direct use value (e.g. bird watching)	Indirect valuations (revealed expenditure methods, contingent valuation method)
Generation of option value	Option value	Empirical assessments of individual risk-aversion

For each ecosystem service, various valuation techniques can be applied (Constanza et al., 1997). As a result, regulating functions have mainly been valued using indirect market valuation techniques, whereas habitat functions have been valued using direct market pricing. Productive ecosystem functions have mainly been assessed through direct market pricing and factor income models. Information functions have been measured using contingent valuation for cultural and spiritual information, hedonic pricing for aesthetic values and market pricing for recreation, tourism or science. These results show that the quality of the valuation of ecosystem services in monetary estimates of human preferences is strongly dependent on the application of the best available valuation technique. In order to prevent double counting of several services an appropriate rank ordering would be necessary in case of the valuation of an entire ecosystem including each of the services provided by it (Bockstael and Freeman, 2005).

3.2.2 Method Selection

The selection of suitable methods for the valuation of ecosystem services needs to represent a consideration of technical, institutional, user-specified and financial aspects. In particular, the context and the further application of the results generated using a specific technique determines its selection. Sinden and Worrell (1979, S.433) as cited in Sekot and Schwarzbauer (1995) published the following apposite statement which still needs to be in mind in applying any method of ecosystem valuation:

'All valuation methods, including market prices, must be used judiciously. They do not eliminate the need for judgement on the part of the planner. They are tools, and it is the way they are used that ultimately determines the usefulness of the results.'

3.2.3 Assessment of the Ecosystem Services of Water Supply provided by Forests

There have been various attempts to evaluate the regulating service of water supply provided by forests since the 1970s (Brauman et al., 2007). These studies tried to quantify the value of hydrological environmental amenities such as water quality. In Hawaii, an estimation of the value of a forested watershed arrived at quantities between $ 1.5 and 2.5 billion, the value of an improvement of dam functionality at the Yangtze river of up to US $ 600,000 and the value of a native forest for water supply in Chile being worth US $ 200 per hectare per year. Even though the value of hydrologic services is apparently recognized in environmental sciences, the value of ecosystems is highly variable. Thus, it is inevitable to conduct a site-specific assessment of these services in order to arrive at realistic lower bounds for the value of hydrologic ecosystem services (Brauman et al., 2007). Table 4 illustrates suitable valuation techniques applied for the estimation of the value of regulating ecosystem services.

Table 4 Relationship between regulating ecosystem services and applied valuation techniques (based on Constanza et al 1987; Bockstael and Freeman, 2005)

Regulating ecosystem functions	Direct market pricing	Indirect market pricing		
		Avoided cost	**Replacement cost**	**Factor income**
Gas regulation		+++	0	0
Climate regulation		+++	0	0
Disturbance regulation		+++	++	0
Water regulation	+	++	0	+++
Water supply	+++	0	++	0
Soil retention		+++	++	0
Soil formation		+++	0	0
Nutrient cycling			+++	0
Waste treatment			+++	0
Pollination	0		+++	++
Biological control	+		+++	++

+++ high suitability, 0 zero suitability

Referring to hydrological services such as flood control, water filtration or flow regulation, Bergstrom et al. (2001) applies the costs which would have come up due to engineering solutions such as the construction of dykes, treatment plants and reservoirs to represent their value. In addition, opportunity cost could be calculated, looking at cost which would arise due to a transportation of water from further upstream or downstream of the source or a transfer to another watershed. As a result, prices of water are not able to properly estimate the value of the hydrologic services of an ecosystem (Bergstrom et al., 2001).

In addition, Sisak (2009) estimated the socio-economic value of hydrological forest services using the costs-of-prevention approach, calculating costs for technical measures substituting the respective hydrological service of the forest. Resulting socio-economic values arrived at 23.1 €/ha per year and a capitalised value of 1,153 €/ha (interest rate: 2 %) for the hydrologic service of water quality provided by forests.

Also Sekot and Schwarzbauer (1995) agree on the suitability of the replacement cost method in the context of an evaluation of water treatment cost in case of different land use alternatives. A differentiated evaluation must be based on an assured causality of forested catchment, water quality and treatment cost. Furthermore, this approach is only applicable in case of total inelasticity of demand (Sekot and Schwarzbauer, 1995).

This thesis is going to use a similar approach applying the replacement cost technique to estimate the value of forests for water supply using treatment cost or higher cost of alternative water sources. The overall value of land management is then generated using the value of water supply based on replacement costs to perform a cost-effectiveness-analysis of the different land use alternatives. This approach of managerial-economics is applied concentrating on the specifically produced hydrologic services derived from silvicultural practices.

3.2.4 Opportunity Cost

Forestry aiming at a maximised protection of water supply implies changes in cash flow for the forest operator as it is usually not in accordance with economically optimal forest management (Rüping, 2009). Thus, it is necessary to integrate this reduction of revenues into an economic analysis of different management strategies.

Opportunity costs represent the benefits which would have arrived if the best alternative option was chosen. Thus, it is an indirect approach of evaluating benefits as the value of a good arrives at the benefit, which could be achieved using the input needed to derive the good for an alternative application. In assuming an efficient resource allocation, the benefit of the realised alternative must be at least as high as the benefit which could be derived by an alternative usage (Sekot and Schwarzbauer, 1995).

One major problem in applying the opportunity cost approach represents the measurement of the lost profit compared to the best alternative or the reference strategy. In most applications it is difficult to identify a relevant alternative and to measure its value (Sekot and Schwarzbauer, 1995). In addition, opportunity costs only represent a lower bound estimate for appropriate compensation payments aiming at compensating the user for the lost revenue (Möhring and Rüpig, 2006).

The base structure of the capitalised value can be divided into three basic steps (Möhring and Rüping, 2006):

- Evaluation of the expected long term effects for a reference management strategy as well as a change in management. Therefore, the quantity structure, input and output in physical units according to the temporal course of production need to be determined.
- Determination of cash flows using the prices of inputs and outputs based on the quantity structure.
- Estimation of the net total revenues of the reference management strategy and the alternative development. Thus, the marginal price being the amount of money which should be at least paid to the owner in order to compensate the losses is derived.

In conformance with Möhring and Rüping (2006), Rüping (2009) and Grottker (1999) this application is using a capitalised value approach using empirical yield tables of spruce and beech assuming that the dynamics of stand development used for forest economical studies can be adequately described by them. The net earnings are represented in € per ha calculating the difference between revenues and costs. It is assumed to have a spatially distribution of all ages covering the same area using a normal operation class (*Normalbetriebsklasse*) (Sekot, 2004).

The forest stand model is dependent on stand age, describing the volume and the diameter at breast height (DBH) for the eliminated and the remaining stand during the temporal course of production. As a result, revenues and costs are determined by DBH, whereat constant rates of cost and revenues are assumed over the whole period. This steady yield model is usually used in classical forest valuations (Rüping, 2009).

Forest revenues are based on mean values of timber prices in the region. Value categories can be derived from stand assortment tables using the portion of trunk wood of the total solid wood in combination with the distribution of quality classes (Rüping, 2009).

For a complete model of production values of mean stand diameters (using calibrated yield tables), solid wood mass respectively stock, growth rates and biomass being regionally adapted, need to be integrated into the analysis. Additionally, site adapted values for the rates of costs and revenues including costs of timber extraction, plantation and stand tending, timber revenues and other costs, arising due to organisation or subsidiary uses need to be investigated.

In case of mixed stands, the calculation of annual timber production values includes different tree species proportionally. As a result, mixed stands are the sum of different idealised pure stands which represents a simplifying assumption (Rüping, 2009). This representation of idealised subareas in mixed stands is also used by Grottker (1999).

In addition, the model assumes that the proportion of represented tree species remain constant over the whole period. Non-linear equations describe the relationship between stand density (stocking level) and stand growth. Thus, increasing stand densities imply a decrease in stand growth (Rüping, 2009).

Moreover, Rüping (2009) includes a discount rate into his analysis, arguing for the application of a marginal interest rate for forest owners in the range of 2 %. On the other hand, Grottker (1999) states that there is a huge range of interest rates, which can actually not be predicted and having a strong impact on marginal income of forests. Thus, he uses a simple approach based on the present conditions of production and revenues. The legitimisation of referring to present values is represented by the compensation of an increase in cost due to feedback mechanisms of the market by an increase in production. Beyond that, Grottker (1999) assumes a homogenous stand age distribution using the Normal Forest Model. Thus, the temporal succession of various production steps is transformed into a spatial coexistence superseding the application of an interest rate (Grottker, 1999).

In conclusion, Grottker (1999) and Rübing (2009) both use empirical yield tables for the estimation of forest revenues. These growth models are based on measurements which were performed on experimental fields. Thus, these values do not always comply with

regionally altering growth conditions. Beyond that, yield tables ignore stand risks as for example windthrow or breakages due to snow (Speidel 1984). A specific consideration of risks regarding probability of stand damages would be principally feasible using a capitalised value but increases the complexity of the analysis (Sekot, 2004).
Additionally, Möhring and Rüping (2004) propose careful usage of yield tables as they seduce to their application without having dealt with the models lying underneath in depth. Therefore, internal records of the forest company, assumptions derived from experts or values derived from different scenarios should be applied to provide a database for the calculation of the capitalised value. Thus, any interpretation of the results of this approach needs to include the quality of the database in order to evaluate the quality of the analysis (Sekot, 2004). As this approach uses mean and temporal invariant conditions, especially concerning revenues and costs, Sekot (2004) recommends the usage of mean values over multiple years instead of actual values.
The biggest advantage provided by the calculation of net earnings lies in the economic analysis of the long-term results provided by different alternatives. Owing to that fact, the comparison of net earnings illustrates which of the reviewed variants of the Normal Forest Model provides higher economical revenues regarding current conditions. Thus it can only be used as a decision support tool using the underlying assumption, that the compared alternatives are already established as an analysis of the conversion of one scenario to another would require a dynamical capital budgeting (Sekot, 2004).

In contrast to the net income approach which was described above, the standard marginal income calculation represents a second approach used for the valuation of timber revenues in forest economics. This approach does not include general expenses (e.g. buildings, administration) and provides tables of guideline values for the valuation of timber revenues. Again, these values need to be critically examined and adapted to regional conditions. The standard marginal income calculation is frequently applied when it comes to the analysis of the contribution of forests to revenues from agriculture and forestry (Sekot, 2004).

3.2.5 Cost-Effectiveness-Analysis

This thesis aggregates timber revenues represented by opportunity cost and the lower bound for the value of the hydrologic services of the forested catchment area indicated by the replacement cost method using a cost-effectiveness analysis (CEA). In contrast to the benefit-cost analysis, the cost-effectiveness analysis does not evaluate all benefits in a monetary way. Thus, the CEA results in a comparison of monetary benefits of a project with the positive, quantitatively/qualitatively expressed outcomes. The monetary described project costs and benefits facilitate the analysis of the most cost-effective alternative to reach the goal of a project. The confrontation of monetary costs with non-monetary accruing impacts is done by a subjective valuation. This includes an implicit valuation of the benefits (Sekot and Schwarzbauer, 1995).

As the CEA approach does not only apply a holistic monetary consideration of all impacts of different land use alternatives, it is easier to apply than a benefit-cost analysis. In addition, the acceptance of CEA in scientific literature is much higher than that of benefit-cost analysis (Sekot and Schwarzbauer, 1995).

On the other hand, changes in net social welfare cannot be analysed with this approach. As cost-benefit analyses are based on a completely monetary approach of valuation, cost-utility analysis represents a multidimensional approach to support decisions. In this context, the cost-effectiveness-analysis takes up an intermediate position, as the estimation of inputs is done using monetary measures, but the attainable goals are estimated in a non-monetary way (Sekot and Schwarzbauer, 1995).

This thesis applies the CEA due to a lack of resources for a full benefit-cost analysis. Additionally, the benefit-cost analysis is highly criticised questioning its approaches of resource allocation and the aggregation of interpersonal benefits. Frequently, an all-embracing benefit-cost analysis is not possible due to a lack of data.

In this context a multi-criteria analysis would be the most appropriate way to capture the hydrologic ecosystem services provided by the forested catchment of the Viennese spring water main. The multi-criteria approach evaluates different alternatives with respect to their contribution to preset goals for various criteria using multiple dimensions (Sekot and Schwarzbauer, 1995). Due to a lack of time, it is not possible to perform an evaluation including multiple criteria in the decision making process.

The cost effectiveness of an alternative in focus can be derived by a division of annually arriving costs of an option by physical benefit measures. These physical benefit measures could be e.g. number of species recovered, tons of emissions of a given pollutant, kilometres of river length restored (Görlach, s.a.).

3.2.6 Criticism towards Cost Based Valuation Methods

In general, the most critical point concerning the application of cost based approaches represents the fact, that costs are not directly linked to the utility function of consumers. This interrelationship being only possible in case of perfect market equilibrium represents a vocal point of the valuation of economic values based on welfare theory. As cost based methods ignore the demand and respectively the valuation of affected people, the benefits for society derived by ecosystem services is just assumed but not derived from the preference of the individual. As a result, the benefit resulting out of an alternative equals the costs. Thus, it is assumed that higher costs correspond to higher benefit automatically and zero or low cost of production leads to a low value of the good or service in focus. In addition, cost based approaches cannot determine an optimal measure of allocation of goods. On the other hand, cost based methods are able to derive information of substantial importance for the decision making process (Sekot and Schwarzbauer, 1995).

4 Application

The water supply of Vienna represents a source of drinking water of best quality providing intangible value. The goal of this case study is to identify the processes enabling the production of the ecosystem services of water supply in this region and to convert them into monetary units providing a lower bound estimate of the value the ecosystem services. Based on the three steps of assessment published by Daily et al. (2009), the methodology which was described in the previous chapters is applied.
Therefore the first step of valuation of ecosystem services includes the description of suitable management alternatives in the catchment area. In the headwaters of Vienna it is most important to preserve the protective functions which are provided by the forests of the region. Due to this fact, this analysis evaluates different approaches concerning forest management. The second step of evaluation represents the modelling phase, in which the Water Erosion Prediction Project is applied and which serves as foundation for the assessment phase. As a last step the translation of the consequences of the modeled alternatives into monetary units takes place which is done using the replacement cost technique as well as a cost-effectiveness analysis.

4.1 Description of the Catchment Area

In Austria, more than 90 % of the drinking water does not require any treatment. It consists to 49 % of spring water and 51 % of groundwater. In general, 87 % of the Austrian population is supplied with drinking water by central water works. This is also the case in the water supply of the city of Vienna (Sailer, 2011; Neunteufel, 2010).

In Vienna, 97.5 % of the drinking water for the population of 1.760.000 inhabitants is provided by spring water of high quality. The drinking water is transported by two large pipe systems, which are called the first and the second spring water main. At the moment, 44.8 % of the drinking water which equals 63.7 mio. m^3, are provided from the Rax/Schneeberg region by the first spring water main having an average flow time of 16 hours. This free level canal has a length of 120 km length and was opened in 1873.
In addition, 50.9 % (72.4 mio. m^3) of drinking water are transported by the second pipe from the Hochschwab region. Its travel time takes 36 hours in average. This pipe having a length of 170 km was constructed from 1890 to 1910 (Sailer, 2011; Vienna Water works, 2011, Neunteufel, 2010). Figure 19 shows the track of the Viennese mains. In addition, the spatial distribution of the springs in the catchment is illustrated.

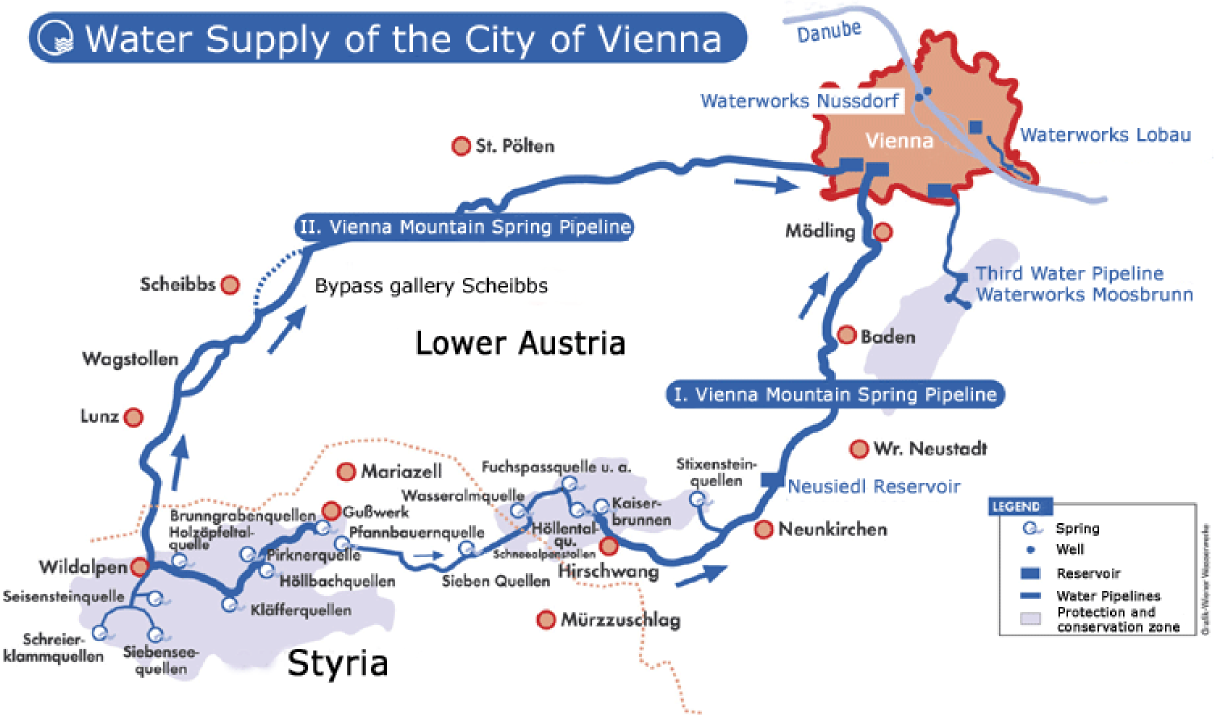

Figure 19 The system of water supply for the city of Vienna (Vienna Water works, 2011)

In normal operation 100 % of the drinking water of Vienna stems from the karstic Alps. Only in case of high demand (max. daily demand of Vienna: 530.000 m³) or during maintenance works, water is extracted from groundwater forming in total 5 % of the water supply on a yearly basis. The average daily consumption arrives at 380.000 m³/d which equals 240 l/inh.,d, and is strongly seasonally altering as the actual demand mostly depends on tourism. The drinking water of Vienna is managed by the Vienna water works which is a municipal authority (MA 31) and thus part of the city administration (Sailer, 2011; Vienna Water works, 2011; Neunteufel, 2010).

Viennese drinking water shows high quality, due to the large scale spring protection areas in the catchment. In 1965 the entire region of Rax-Schneeberg-Schneealpe was declared a water protection zone. This protection zone extends to an area of approximately 900 km² and is owned by the city of Vienna. Ducts and galleries collecting water from several well-springs transport the water to the city of Vienna by flow by gravity. Using this gradient, the Vienna water works produce electricity of 65 mio. kWh per year in their drinking water plants. At Vienna, storage reservoirs form the end point of the spring water mains balancing the water yields with the demand of the city. They provide a maximum storage capacity of 1.5 mio. m³ which equals the average consumption of Vienna of four days. From these tanks water is distributed to the city using a pipe network of more than 3000 km.

Owing to the high quality of the drinking water, Vienna does not need any treatment, except disinfection using chlorine (ClO_2). In case of contaminations, an online monitoring system

controls the quality parameters such as pH, temperature, SAC[3], conductivity, turbidity, oxygen, TOC/DOC[4] and radioactivity of the water. In case of exceedance of these values, there is a pre-warning time of 20 hours to take measures. In general, the Vienna Water works strictly divert contaminated out of the supply network using the stored amount of drinking water compensatory (Sailer, 2011; Vienna Water works, 2011; Neunteufel, 2010).
The water fee in Vienna is uniform for all customers, private as well as industrial purchasers, arriving at 1.73 € (including 10 % VAT) per cubic meter of drinking water. There is no base fee or quantity rate implied (Sailer, 2011; Vienna Water works, 2012a).

The water protection zone covers an area of 900 km² ranging in altitude from 470 m ASL (in the Eastern valley) up to 2277 m ASL (Mount Hochschwab). It is situated in the North-Eastern Calcareous Alps at about 47 °N and 15 °E (Köck et al., 2007). Figure 20 shows a representation of the altitudinal zones represented in the catchment area. In addition, it shows the location of the catchment areas of the 1. spring water main (Rax, Schneeberg and Schneealpe in the East) and the catchment of the 2. spring water main in the area of Hochschwab in the West (IWHW, 2012).

[3]spectral absorption coefficient

[4] Total organic carbon / dissolved organic carbon

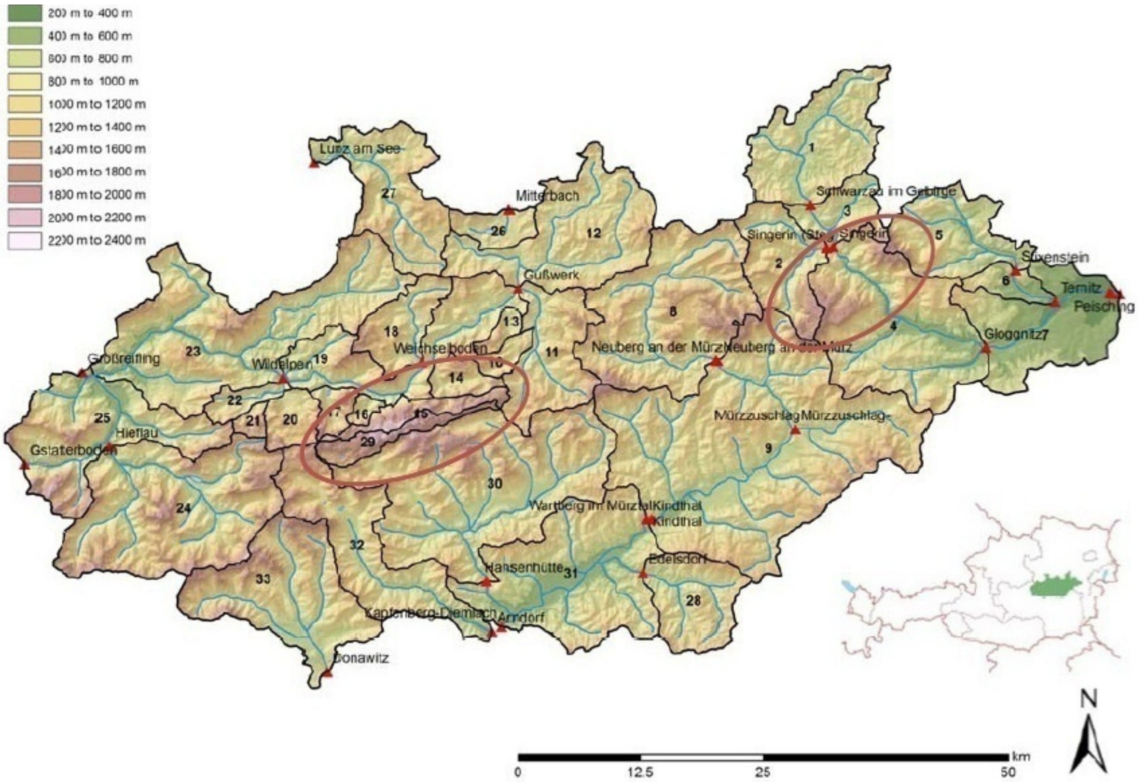

Figure 20 Distribution of altitudinal zones of the catchment areas of the 1. and 2. spring water main of Vienna including the location of the catchment zone in Austria (IWHW, 2012)

The humid climate of the catchment area is influenced by the Atlantic Ocean providing a comparatively mild climate for Europe. Annual mean temperatures were lying in the range of 6.7 °C in Mariazell (875 m ASL) and 9.4°C in Reichenau (485 m ASL) in the time period from 1961 – 1990. Mean annual precipitation was measured arriving at values about 1071 mm to 614 mm (Köck et al., 2007).

4.1.1 Actual Land Use

The following figure 21 illustrates the present land use categories in the catchment area based on the hydrological atlas of Austria (digHAO). It differentiates six land use categories including deciduous and mixed forest, coniferous forest, grassland, cultivated area, areas of settlement, rock and water bodies (IWHW, 2012).

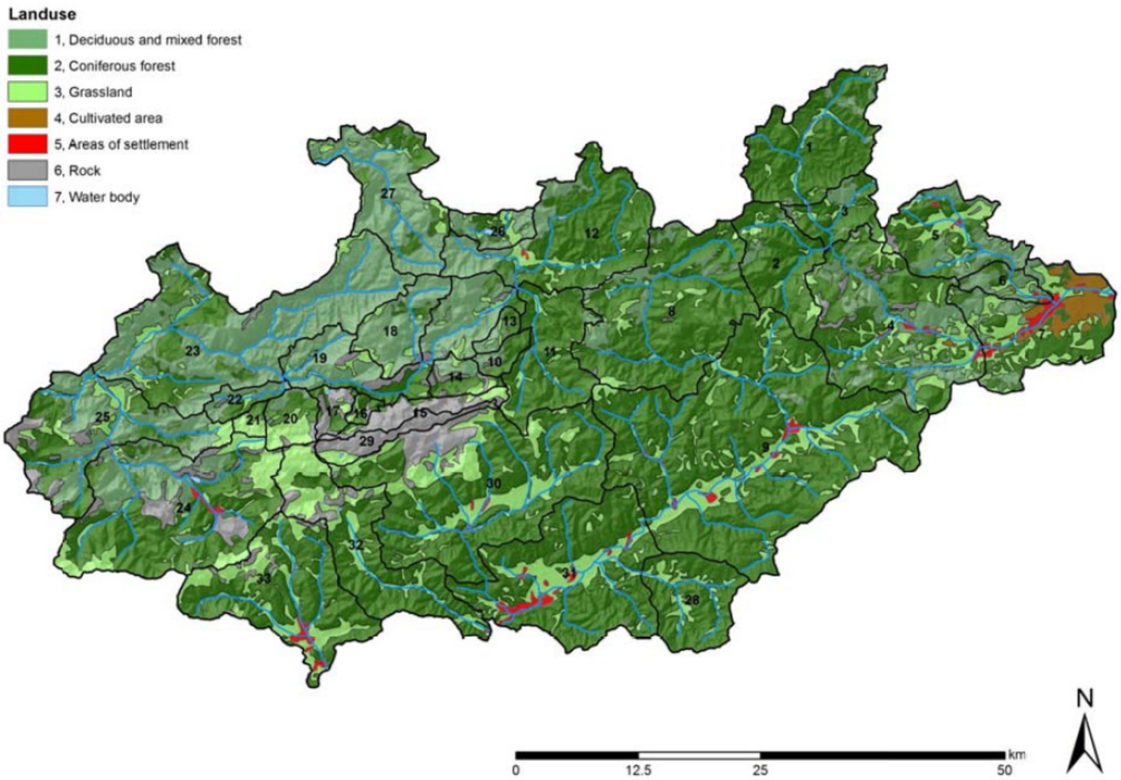

Figure 21 Distribution of land use categories in the catchment (IWHW, 2012)

Two thirds of this region are covered by forests, providing drinking water for the population of Vienna (Köck et al., 2007). 32,900 ha of the area of the water protection zone are currently cultivated and are owned by the city of Vienna. 13,700 ha of this area represent non-forested areas such as alpine pastures and rock formations (MA 49, 2012; Köck et al., 2007).

In the catchment area of Schwarza and Mürz coniferous forest represents the biggest fraction. In contrast, mixed forests are dominating in the North-Western region. At mount Hochschwab, mainly areas without vegetation, rocks and grasslands are present (IWHW, 2012). Figure 22 shows the areal representation of different kinds of land use in the catchment area, with coniferous forest forming the predominant vegetation type.

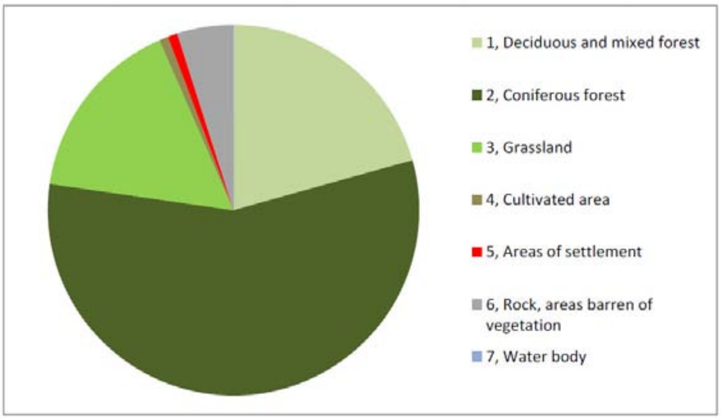

Figure 22 Areal distribution of different kinds of land management in the catchment area (IWHW, 2011)

Owing to the wide range of altitudinal zones being represented in the catchment area, there are many different kinds of forest communities/hydrotopes present in the region. Each of these communities shows a different tree species composition potentially occurring on the site in focus. As a result, site specific factors including naturally occurring forest communities, soil types, altitudinal ranges and exposition determine the hydrologic behaviour of the region. The spruce-fir-beech forest hydrotope represents the most common forest community in the headwaters (Köck et al., 2007).

In former times, historical land use including clear-cuts as well as the production of fuel wood and charcoal displaced at many sites the naturally occurring forests and created homogenous conifer plantations. Additionally, beech or other broadleaved species were systematically eliminated aiming at the creation of single layered spruce stands (*Picea abies* L. Karst.). Clear-cutting as the common technique to extract timber neglected forest-hydrological aspects (Köck et al., 2007). These areas are now at high risk in case of strong winds, large amounts of snow and pest infestations. Therefore, the municipal forest authority seeks to convert spruce dominated stands into naturally occurring, site adapted forest stands. Also natural forest regeneration represents a basic goal in the forest management of the area (MA 49, 2012).

Additionally, dwarf pine belts are wide spread in the water protection zone. Due to the conversion of these dwarf pine belts into alpine pastures for grazing of cattle in the past, this vegetation zone was displaced in many regions. During the last decades, the area of alpine pastures decreased, resulting in a mosaic of dwarf pine zones and alpine pastures (Plan, 2005).

In the cultivated forested regions, the forest department of Vienna (MA 49) aims at the conservation of good soil conditions and an site adapted site structure in order to secure the protection of high water quality (MA 49, 2012). Therefore, the MA 49 seeks to apply single tree extraction or small scale interference instead of clear-cuts, enhance natural stand regeneration, foster ecologically adapted tree species and niches such as wetlands, oligotrophic and xeric grasslands and prevent old-established trees from extraction. The management strategy focuses on high stand stability and a minimisation of the stand's vulnerability using unevenly age-distributed trees. A high diversity of autochthon tree species should be fostered in order to prevent risk of windthrow and to build up humus layers offering high filter and storage capacities. The application of chemical agents including insecticides, fungicides and the extraction of mineral fertilizers is banned in the whole water protection area (MA 49, 2012).

Timber extraction is mostly carried out using cable yarders, as this technique is suitable for relatively steep terrain and prevents high levels of soil disturbance (MA 49, 2012). In former times, timber extraction in the region of forested plateaus such as Kuhschneeberg, Scheibwald and Unterer Kesselboden after big events of windthrows was performed using heavy machinery leaving behind approximately 30 km of striking marks in the landscape. Especially in steeper terrain, these extraction techniques led to soil compaction and erosion (Plan, 2005).

Forest roads are built based on geologic expertise in order to prevent negative impacts on water quality (MA 49, 2012). There are 45 km of fixed forest roads present in the catchment area (Plan, 2005). The MA 49 therefore seeks to buy forest roads only at vital locations. Approximately 35,000 m³ of timber is logged in the water protection forests each year (MA 49, 2012).

Eleven operated cabins, 28 unmanaged cabins, a monument and two chapels represent the buildings being present in the catchment area. Additionally, approximately 145 km of hiking trails can be found in the region. Pasture management takes place in three different regions of the area (Ochsenboden on Schneeberg and Kuhschneeberg, Taupentalalm) of which no clear boundaries can be drawn (Plan, 2005).

4.1.2 Geology and Soils

The dominating geological formations in the headwaters of the water protection area are limestone and dolomite being partly under lied by silicatic sandstone (Werfener Schichten) (Köck et al., 2007 cited Cornelius, 1952; Mandl et al., 1994). Figure 23 presents an overview of the catchment area of these karstic regions.

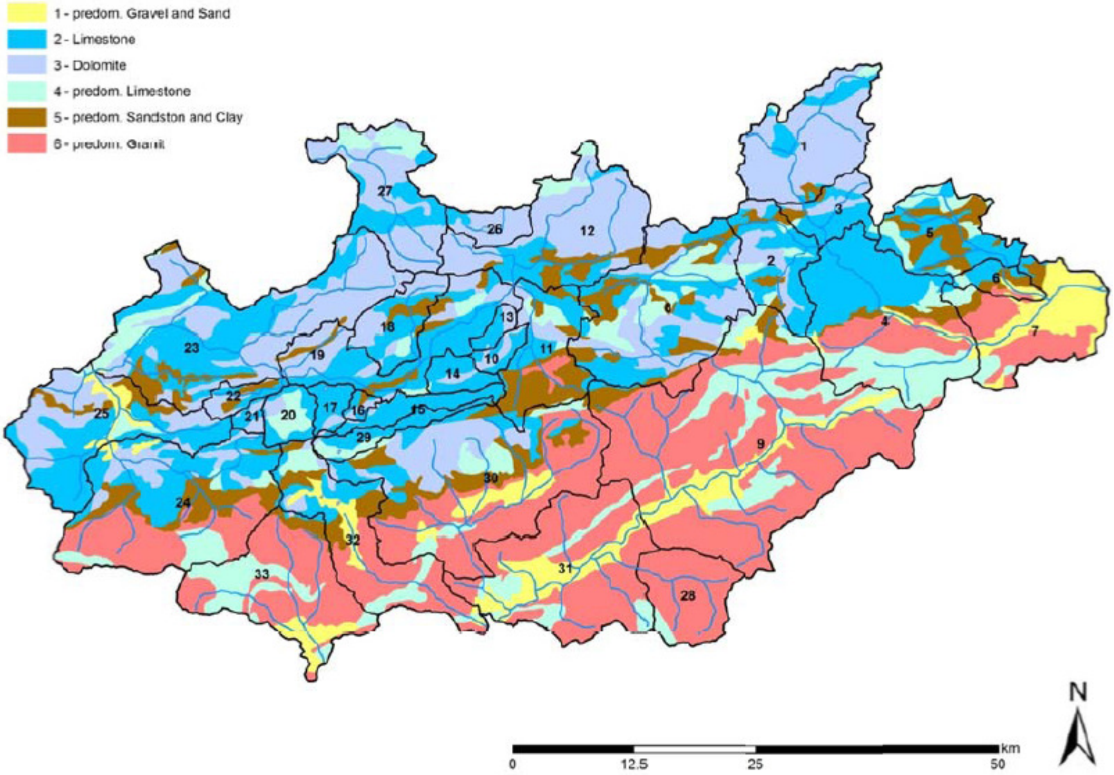

Figure 23 Geological overview of the catchment area. No. 15/29 Hochschwab, Nr. 2/3 Schneeberg & Rax, Nr. 8/9 catchment area of Mürz, Nr. 27 Ois/Ybbs (IWHW, 2012)

The high plateaus of Hochschwab, Schneeberg and Rax including steep mountainsides represent characteristic features of the area. The Northern part of the headwaters including the whole catchment area of Salza as well as the upper reaches of Mürz and Schwarza is situated in the Northern calcareous Alps (IWHW, 2012).

Within the headwaters region, Wetterstein-limestone represents the most important karst-aquifer. It builds up several of the mountain massifs (Rax, Schneeberg, large parts of Hochschwab) and reaches a thickness of up to 1500 m at Hochschwab.

Wetterstein-dolomite forms the most prevalent form of dolomite in the region. It is very wide spread in the headwaters and is weathered easy due to tight fissures. In case of in-situ weathering of the dolomite-formations, rocks are loosened and thus easily eroded on steep mountainsides. As a result, deep erosion gutters can be developed and soils are mostly shallow or not-existent. Additionally, it can act as an aquifer offering a high potential of water retention (Köck et al., 2007).

Furthermore, rendzic leptosol, chromic cambisols (Köck et al., 2007 cited FAO, ISRIC and ISSS, 1998) and transitions of them represent the predominant soil types on the widespread carbonatic bedrock of the catchment area. Additionally, silicatic bedrock (sandstone) is sparsely distributed forming leptic cambisol, dystric planosols and gleysols (Köck et al, 2007 cited FAO, ISRIC and ISSS, 1998).

Rendzic leptosols are widespread in the headwaters of the water protection zone and represent typical soils of limestone and dolomite with a low impurity. This soil type is rich in humus and soil skeleton but offers a low soil water storage capacity (Köck et al., 2007).

Additionally, mosaic forms of rendzic leptosol and chromic cambisol/rendzic leptosol compounds (Insellehm-Komplexe) represents inhomogeneous soil complexes frequently occurring in the North-Eastern calcareous Alps. Its soil matrix does not manifest a horizon with loamy soil substances and its water storage capacity is low. These complexes can be found on small scales about 10m² as well as large scales of more than 10 m² (Köck et al., 2007).

The karstic headwaters are characterised by heterogeneous soil formations due to boundary conditions of geology. Soil depths in this region show a strong variation resulting in highly variable water storage capacities in the headwaters of Vienna. In addition, moder humus mainly dominates the area. Also a high proportion of thick humus layers being of high importance for water protection and highly vulnerable occurs in this region (Köck et al., 2007).

According to Köck et al (2007), about 80 % of the investigated sites of the water protection area show stable humus dynamics. This rather high value indicates a low amount of erosion processes being of significant importance for the protection of water quality. Only the area of Hirschwang shows a high percentage of decomposed humus layers (6 %). On these sites, the water protection functionality of the forest is decreased by former clear-cuts or windthrows resulting in erosion processes of the humus layers (Köck et al., 2007).

4.1.3 Vulnerability of the Aquifer

The karst aquifers of the drainage basin are represented by typical morphology, special landforms and subsurface drainage to the major springs. As a result, karst morphology, geology and soil coverage represent the main factors influencing the vulnerability of the aquifer. Especially areas of rapid infiltration such as depressions like dolines and streams sinking into ponors represent places asserting high risk of contamination. They show extremely high infiltration rates being in contrast to the normal pattern of diffuse infiltration in karstic areas which is illustrated in figure 24.

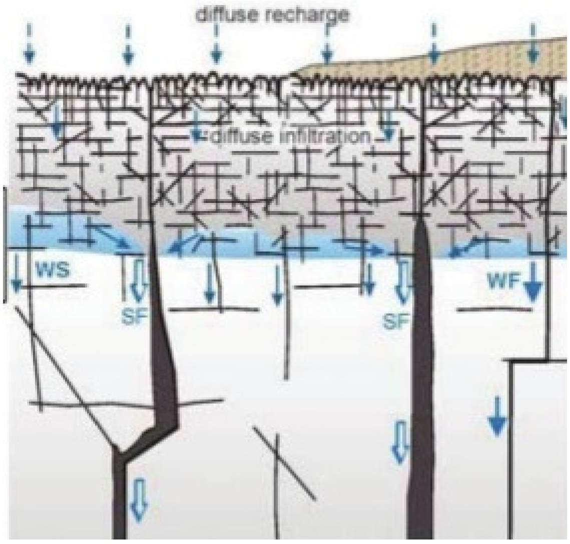

Figure 24 Hydrologic features of the epikarstic zone (Plan et al., 2007 after Klimchouk, 2004)

Their vulnerability shows a positive correlation with the size of these features and the size of the catchment area (Plan et al., 2007). Plan et al. (2007) detected potential hazards in the catchment area of the Viennese water supply asserted by pasture, tourism, forestry and hunting.

The IWHW (2012) investigated the sensitivity of the springs to turbidity in the catchment area of the 2. spring water main of Vienna. According to the modelling approach the different runoff components (surface flow, interflow and baseflow) were simulated for each catchment. It turned out that those springs with a discharge less or equal to the base flow component were not affected by turbidity. In contrast, those wells where the spring discharge was composed of several runoff components were quite sensitive to turbidity originating from extreme rainfall events. As a result, the Kläffer and Schreierklamm spring were identified as the two springs being most sensitive to high values of turbidity. High values of turbidity could mainly be reported seasonally after snow melt and partly due to instantaneous precipitation events. In case of high initial intensities of precipitation events, the attenuation of an increase in turbidity decreases more slowly with an increase of event duration. Increased values of turbidity are mainly correlated to erosion and surface runoff resulting out of heavy precipitation events. In addition, a future increase in turbidity due to climate change is not expected to occur (IWHW, 2012).

4.2 Land Use Alternatives in the Water Protection Zone

The evaluation of the hydrological services of forests in the catchment area of the Viennese spring water mains is based on the analysis of two different land use alternatives. In this mountainous region, the forest provides a range of ecosystem services. Therefore, not only its positive impacts on water supply but also soil conservation is in the viewpoint of forest management. As the forests of this area are protected by the Austrian forest law from 1975, § 21 section 2 the following analysis is restricted to a comparison of different forest management practices in the region, being the only suitable form of land use.

Forests strategies emphasizing the water protective value of forested areas in comparison to forest management aiming at a maximum economic profitability of them are characterised by different measures of forest management. Table 5 shows important forest measures seeking to maximise the water protective impacts of forests and their impacts on the intensity of timber production.

Table 5 Water protective forest measures with regard to their implications on forest productivity (adaptation of Rüping, 2009)

Forest practices to protect water supply	Changes in biological production					Changes in technical production
	After regular usage of existing stands	Change in stand management				
	Change in tree species composition	Premature extraction	Change in strategy of treatment	Prolongation of the production period	Permanent waiving of usage	
Encouragement of broad-leaved trees	X	X				
Alteration of non-site-specific stands	X	X				
Premature logging		X				
Intensity of logging			X			
Abandonement of clear-cuts						X
Encouragement of large-scale forested areas						X
Encouragement of horizontal stands						X
Abandonement of tillage operations						X
Encouragement of natural regeneration			X			X
Whole tree cutting						X
Soil protective tree extraction						X
Avoidance of timber storage in the stand						X
Usage of environmental friendly materials						X
Construction of forest roads with regard to water conservation						X
No usage of plant protecting agents						X
Deadwood				X	X	

The impacts of generalised water protective measures on water supply can differ significantly being determined by site-specific properties such as soils, climate and stand history (Rüping, 2009).

Evaluating the impacts of forest management on water supply, this thesis differentiates two strategies of forest management seeking to maximise different goals. Therefore, continuous

cover forestry seeking to provide a maximum protection of the water bodies is compared to intensive forest management aiming at maximum forest revenues.
In the last centuries, forestry practice such as clear-cutting in combination with afforestation using Norway spruce (*Picea abies* Karst.) was frequently applied in Austria. Thus, the comparison of a homogenous spruce dominated stand with a natural mixed spruce-beech stand represents a realistic confrontation (Köck et al., 2007). These alternatives are presented in the following section.

4.2.1 Permanent Forest Coverage

In this context, permanent forest coverage refers to a forest strategy aiming at the protection of water supply through the application of best management practice. It represents a low to intermediate disturbance regime having a high forest density (Novak, 2011). Thus, the sustainability and multifunctionality of the ecosystem should be preserved. The support of natural structures and natural regeneration pattern represents a basic principle of close-to-nature forest management (Pilas et al., 2011).

In order to prevent erosion, best management practice includes the vegetative coverage of the surface of the whole catchment area and continuous natural regeneration under adult trees. In addition, planting of trees with deep rooting systems to stabilise soils as well as careful road construction and maintenance can prevent sedimentation (Pilas et al., 2011). Therefore, the idealised concept of continuous cover strategy presented in this thesis aims at a mixed stand including a tree species distribution of 50 % beech and 50 % spruce. In terms of high flow prevention, uneven-aged stands with dense canopy covers, diverse vertical structure and an even distribution of growth-phases represent mitigating properties (Frehner et al., 2005). According to Twery and Hornbeck (2001) the total share of non-forested areas and that of regenerating of younger trees than 10 years should represent less than 25 % of the drainage basin area in order to decrease high flows. Additionally, the canopy cover of all forest stands in the basin is meant to be more than 70 % (Twery and Hornbeck, 2001). As a result, the stand considered in this thesis is characterised by trees in different growth stages having a dense canopy cover with small areas without coverage. Timber extraction takes place using cable yards and thus minimising soil compaction.
The intensity of any intervention in the forest should be held as low as possible as each harvested tree leads to higher concentration of solutes in the soil, root material dies off and formerly bound substances such as nitrate is released and higher decomposition of litter takes place. Therefore, tree extraction should be minimised to 10-15 % of the total stand volume combined with a 10-year interval after each interference (Köck, 2008). Additionally,

continuous cover forestry guarantees a high stability in the humus dynamics of the stand (Köck et al., 2007).
In general it can be stated, that soil vegetation is not able to replace the absence of trees on any site. Only an optimal forest cover, as it is given in continuously covered stands, in combination with the presence of soil vegetation cover represents the best prevention when it comes to processes such as erosion of soil or humus (Köck et al., 2007).

4.2.2 Intensive Forestry

Intensive forestry is the economically oriented alternative to continuous cover forestry presented in this analysis. This management strategy is characterised by Norway spruce being the most dominant species (80-100 %) due to its economic attractiveness derived by rapid growth, high yield and good workability (Diaci, 2002). Spruce stands grow on many sites across Central Europe, regardless of regional suitability. Thus, they exhibit higher drought risk, risk of bark beetle infestation and pest infections as well as degraded soil conditions due to monoculture and air pollution (Pichler et al., 2011). Single-layered, homogenous conifer stands being situated on sites were mixed forests would represent the original tree species composition are significantly at risk of being affected by strong winds or snow damage (Köck et al, 2007). This increase of stand vulnerability due to external factors cannot be integrated in any modelling. As a result, the risk of high rates of erosion accompanied with high sediment yields increases with the intensity of forest management.
In this forest management strategy, timber extraction is conducted by clear-cuts using heavy machinery at lower areas with adequate slopes. In higher, steep terrain, cable yarders are used. The Austrian forest law prohibits clear-cuts being bigger than 0.2 ha limiting the amount of interference due to timber extraction to a certain amount.
Nevertheless, soil compaction resulting out of skid trails, forest roads and the usage of heavy machinery in the stand have severe effects on redistribution and infiltration rates of soils. As a result of clear-cutting, the input of litter drops sharply causing a decrease of the organic layer, preventing drops impact. Also matrix flow is promoted by the high mineralisation rates of the forest floor. Thus, water repellency of the soil surface is increased (Novak, 2011). As a result, there are ubiquitous impacts on the forest hydrological regime due to intensive forest management.

4.2.3 Impacts of Intensive Forest Management on Stand Vulnerability

The conversion of continuous cover forests to forests being managed in an intensive way implies an increase in stand vulnerability. Unstable, homogenous forest stands being dominated by conifers have been established on clear-cut plots during the last centuries. As a result, these stands have been damaged by storm events on wide spread areas. The establishment of homogeneous coniferous stands on area where beech-forests would represent the natural vegetation leads to a decreased stability of forest stands in storm events, snow damages and insect calamities. Köck et al. (2007) clearly outlined the danger of a reduction of forest cover owing to clear-cutting or windthrow. Additionally, changes of crown cover percentage influences the risk of windthrow significantly. As a result, openings of the forest stand leads to high risks of storm damages and subsequent erosion processes. In case of windthrow of forest stands, the elevated risk of root-plate erosion may lead to introduction of soil material into the karst aquifers (Köck et al., 2007).

4.3 Modelling of the Hydrological Processes

This thesis applies the Water Erosion Prediction Project representing a process-based approach to model the forest hydrology of the catchment area of the Viennese spring water mains. Therefore it mainly focuses on the generation and alteration of runoff and sediment yield arising out of different management options for forests.

4.3.1 Model Input

The input for the WEPP model includes a climate, slope, soil and vegetation input files. The setup of these files for the case study area is presented in the following section.

4.3.1.1 Climate Input

The climate input file was produced by CLINGEN the climate generator of WEPP. It was based on climate data from Mariazell covering the period of 1st of October 1971 to 31st of December 1991 using statistical methods to produce the climatic input for a 100 year period of simulation. In order to get the suitable format of data, the CLIGEN station file was derived from online GDS data provided by a database of international daily temperature and precipitation data. This database was published by the Agricultural Research Service of the United States.

The program *GenStPar* was used in order build up the top section of the station file (.par) and to convert this file into a (.top) file. In addition, the program *FindMatch* compared the USFS station files being available with the climate file of Mariazell. As a result missing values in the file of Mariazell were exchanged with those from a U.S. station fitting best to the local climate. This was done using an algorithm applying least squares statistics (Meyer, s.a.).

The resulting daily precipitation amounts simulated over a period of 100 years are illustrated in figure 25. The same pattern of precipitation is used for each simulation of different management alternatives on the slope.

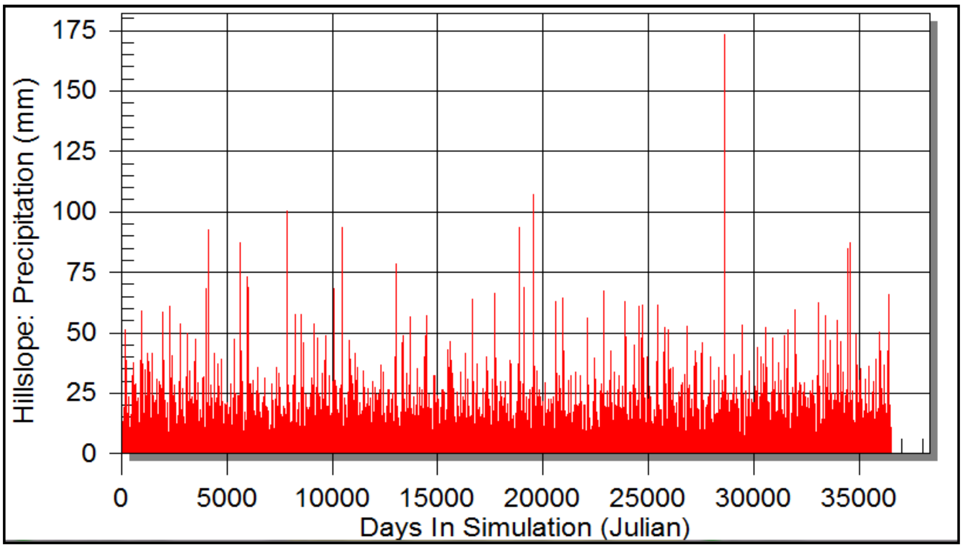

Figure 25 Simulated daily precipitation amounts over a 100 year period using CLIGEN

A comparison of the generated precipitation input values stemming from CLIGEN and average annual precipitation values observed by the ZAMG[5] of Vienna and the Landesstatistik Steiermark is presented in Table 6.

Table 6 Listing of available sources for values of average annual precipitation for the climate station of Mariazell including means, standard deviations and skewness for different observation intervals

	Observance interval	Mean [mm]	Standard deviation	Skew-ness	Source
CLIGEN	100 y	1015	128	0.26	Simulation
Statistics Styria	2003 - 2011	1184	254	-0.57	Landesstatistik Steiermark (2012)
	1971 - 2000	1081	No data	No data	Landesstatistik Steiermark (2012)
ZAMG Vienna	2001 - 2006	1157	246	-1.12	data material of IWHW - station 11171 (ZAMG, 2011)
	1971 - 1987	981	130	-0.25	data material of IWHW - station 11172 (ZAMG, 2011)

In general, there is a big variation in mean values and standard deviations of annual precipitation which is strongly dependent on the observance interval. It can be stated, that CLINGEN slightly underestimates precipitation amounts as well as its variation. This statement is also underlined by Zhang and Garbrecht (2003). According to their analysis of various CLINGEN-simulations for climate stations located in the United States, means and standard deviations of daily precipitation amounts could reasonably well be modeled. Based

[5] Zentralanstalt für Meteorologie und Geodynamik

on the Wilcoxon and Kolmogorov-Smirnov test it was also possible to reproduce probability distributions of monthly and yearly precipitation adequately. Nevertheless, the standard deviation of annual precipitation was mostly under predicted by 15 % on average (Zhang and Garbrecht, 2003).

Beyond that, the comparison of monthly average maximum temperatures, average minimum temperatures as well as average monthly precipitation revealed quite reasonable simulated values. The values which are presented in Table 7 show the comparison of the simulated climate parameters of CLIGEN and the observed climatic data from 1971 to 2000 published by the ZAMG (2012).

Table 7 Root Mean Squared Error of average monthly maximum temperature, minimum temperature and precipitation sums comparing simulated values and observed values of ZAMG Vienna (1971-2000)

	Root mean squared error	
Av. monthly maximum temperature	0.78	°C
Av. monthly minimum temperature	0.43	°C
Av. monthly precipitation sums	9.74	mm

In general the monthly pattern of precipitation which was generated by CLIGEN is slightly smoother than the observed values. Thus, the weather pattern provided by CLIGEN delivered fewer extremes than those which were observed by the ZAMG.

Considering the number of precipitation events >1 mm the simulations also show rather satisfying results. ZAMG (2012) observed on average 140 events > 1 mm per day over one year. CLIGEN generated on average 180 events per year from which approximately 30-40 events are < 1 mm per day.

4.3.1.2 Slope Input

The slope input file is aimed at modelling commonly present representative slopes in the catchment area. Therefore, ArcGIS was used to study inclinations and variations in height of various slopes in the region of interest. Figure 26 shows the slope, resulted out of this analysis representing the basic building unit of the modelling of WEPP.

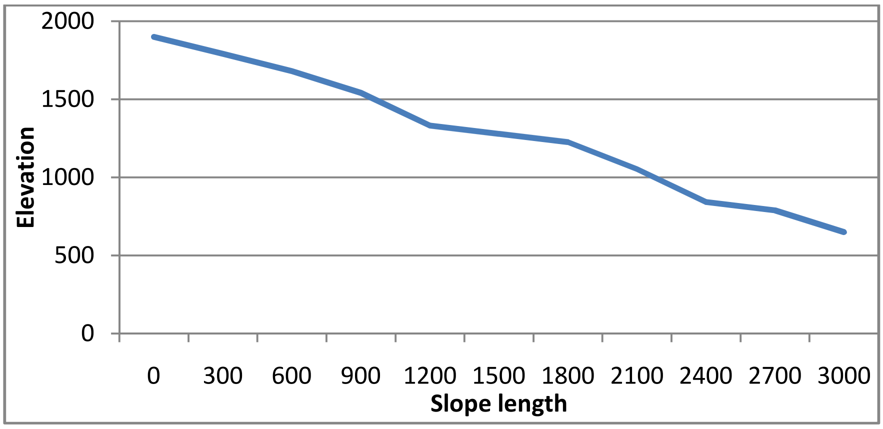

Figure 26 Representative slope which is used as slope input for the WEPP

During the course of this analysis the length of the slope was gradually changed having constant inclinations.

4.3.1.3 Soil Input

The distribution of soil types in the headwaters of the city of Vienna is mainly dominated by rendzic leptosols. Beyond that, the area is characterised by mosaics of different soil types where compounds of rendzic leptosols and chromic cambisols represent widespread soil formations. Especially when it comes to different forest management practices, this soil type offers an intermediate soil type on which intensive forestry would still be possible (Köck, 2011). As a result, the soil input for WEPP is oriented on the soil compound of rendzic leptosol and chromic cambisol (Kalklehm-Rendzina).

This soil type shows loamy or clayey components within the soil matrix occurring between fissures of rocks or between the compounds of carbonatic bedrock. The soil water storage capacity of this soil type can be classified as low to medium. Its rock content varies between 1-19.9 % (Köck et al, 2007).

In the WEPP model, the soil profile is separated into three layers representing the A, B and C horizon. As illustrated in Table 8 each layer is characterised by a specific depth, its content of sand, clay, organic material and rock as well as its CEC[6].

[6] Cation Exchange Capacity

Table 8 Parameter description of the 3 soil layers in the WEPP model

Layer	Depth [mm]	Sand [%]	Clay [%]	Organic [%]	CEC [meq/100g]	Rock [%]
1 = A	250	3	40	20	32	10
2 = B	350	3	40	20	32	10
3 = C	200	3	40	20	32	10

These values were derived from Delaney and Katzensteiner (2012) and Köck (2008). The CEC for forest soils is calculated as the content of clay multiplied with 0.8 as proposed by Elliot and Hall (1997). The resulting values also match with those proposed by the WEPP application help for clay loams/ clays.

Furthermore, the albedo input parameter was set to 20 % and the initial saturation level of the soil is estimated as 75 % in concordance with the values proposed for the COSERO model by the IWHW. Interrill erodibility, rill erodibility, critical shear as well as effective hydraulic conductivity were calculated by the WEPP model using the given fractions of sand, clay and organic matter due to absence of any data material concerning these parameters. The soil input file for WEPP includes the use of a restricting layer of limestone and dolomite. As a result the saturated hydraulic conductivity of the restricting layer is assumed as 0.36 $mm.h^{-1}$ (Domenico and Schwartz,1999). In addition, the WEPP application help suggests the user to set the anisotropy ratio to a default value of 25 if there is no data available. It describes the relative predominance of lateral in contrast to vertical flow (Flanagan and Livingston, 1995). Domenico and Schwartz (1997) advise a range of anisotropy ratio of 1-1000 for most geologic materials.

4.3.1.4 Parameters for Initial Management & Vegetation Input

In the Water Erosion Prediction Project the input file for a specific management of vegetation is based on the parameters for initial management in order to set the initial conditions for the growth period. The values inserted for the following parameter sets were mainly derived in concordance with the expertise of Hochbichler and Köck (2012). The most important parameter for the initial management represents bulk density after last tillage being set to 0.9 $g.cm^{-3}$ based on the findings of Delaney and Katzensteiner (2012). Additional parameters include initial canopy cover, days since last tillage, initial interrill cover etc. which are mainly set to zero as the model starts at the beginning of the vegetation period. Detailed information about that can be looked up in the Appendix 7.1.1.

In order to model the impacts of different strategies of forest management different vegetation files were used. Actually, it was planned to differentiate between altitudinal zones and tree species composition. Due to restrictions of the WEPP model to integrate different forest properties including the dormant season of broadleaved trees, the modelling was restricted to changes in runoff and sediment yield arising out of clear-cuts of perennial spruce forests. Table 9 shows the plant specific parameters which were used to represent a spruce stand in WEPP.

Table 9 Vegetation Input File for WEPP representing a homogenous spruce stand

Parameter description	***unit***	***Spruce***	***Source***
Plant growth and harvest parameters			
Biomass energy ratio	kg/MJ	0.265	Kantor et al. (2009); Korhonen et al. (2007)
Growing degree days to emergence	degrees C.days	5	WEPP
Growing degree days for growing season	degrees C.days	0	WEPP
In-row plant spacing	cm	3500	counsel by Hochbichler and Köck (2012)
Plant stem diameter at maturity	cm	500	counsel by Hochbichler and Köck (2012)
Height of post-harvest standing residue; cutting height	cm	40	counsel by Hochbichler and Köck (2012)
Harvest index (dry crop yield/total above ground dry biomass)	%	48	WEPP
Temperature and radiation parameters			
Base daily air temperatur	degrees C	2	WEPP
Optimal temperature for plant growth	degrees C	20	WEPP
Max temp that stops the growth of a prennial plant	degrees C	40	WEPP
Critical freezing temperature for perennial crop	degrees C	-40	WEPP
Radiation extinction coefficient		0.65	WEPP
Canopy, LAI and root parameters			
Canopy cover coefficient		2	Kantor et al. (2009); Korhonen et al. (2007)
Parameter value for canopy height equation		3	Kantor et al. (2009); Korhonen et al. (2007)
Maximum canopy height	cm	2500	counsel by Hochbichler and Köck (2012)
Maximum leaf area index		8.50	counsel by Hochbichler and Köck (2012)
Maximum root depth	cm	50	counsel by Hochbichler and Köck (2012)
Root to shoot ratio (%root growth/%above ground growth)	%	33	counsel by Hochbichler and Köck (2012)
Maximum root mass for a perennial crop	kg/sq.m	4	counsel by Hochbichler and Köck (2012)
Senescence parameters			
Percent of growing season when leaf area index starts to decline (0-100%)	%	0	no senescence
Period over which senescence occurs	days	0	no senescence
Percent canopy remaining after senescence (0-100%)	%	0	no senescence
Percent of biomass remaining after senescence (0-100%)	%	0	no senescence

The determination of the biomass energy ratio, the canopy cover coefficient and the parameter value for the canopy height equation and leaf area index was based on publications by Kantor et al. (2009) and Korhonen et al. (2007). Based on these publications it was possible to derive average values of tree height, biomass growth and the development of canopy cover of spruce stands at different ages. As a result, an iterative approach applying different parameter sets in exemplary runs enabled the derivation of target growth rates. For example, the canopy cover coefficient could be improved by performing multiple runs and comparing their goodness of fit at different stand ages with measured values of canopy cover in a specific state of the development period. As the development of the canopy cover over the lifetime of a spruce stand is known, it is possible to derive the canopy cover coefficient by various runs of the model.

Other parameters were either set according to the counsel of Hochbichler and Köck (2012) or based on the proposals of WEPP for forest application being presented in the user help database. The senescence option of the model was not used in this application, due to a lack of reliable input parameters.

Initially, it was also planned to include the simulation of mixed stands using vegetation sets being individually prepared for beech into the modelling. Therefore, the responses of areas covered by mixed stands would have been derived from the combination of the results obtained of pure beech and pure spruce stands. Nevertheless, the representation of beech in the WEPP model represents a serious problem as it is not possible to properly implement the dormant season in the model. The WEPP user guide accounts for deciduous trees being dormant every year assuming a minor resistance towards freezing. Therefore, this value is set to 0°C instead of -40°C which is used for spruce. As a result, the whole beech population dies at critical temperatures and regrowth starts with the beginning of the vegetation period from zero again. As a result, this approach does not represent the behavior of beech stands appropriately as not the whole tree should die at critical temperatures, but only the leaves should be lost instead. As a result, any output concerning beech stands cannot be considered as being scientifically valid in this context due to the fact, that the differences in evapotranspiration of deciduous and evergreen forests could cause significant changes in runoff. Thus, these potential differences in runoff occurring due to different tree species distribution of stands could not be integrated in this analysis.

4.3.2 Results of the Modelling

In order to simulate the forest hydrological processes in the headwaters of Vienna WEPP was run using a 100-year simulation period, as it is not capable of performing longer runs. The aim of this analysis is the comparison of the development of a pure spruce stand which is used continuously with a spruce stand being harvested after 90 years. As a result, it should be possible to clearly attribute a change in average annual runoff, average annual soil loss and average annual sediment yield to the alteration of forest management. Figure 27 shows the representative slope of WEPP including one management layer, one slope layer and one soil layer as it is displayed in the user interface.

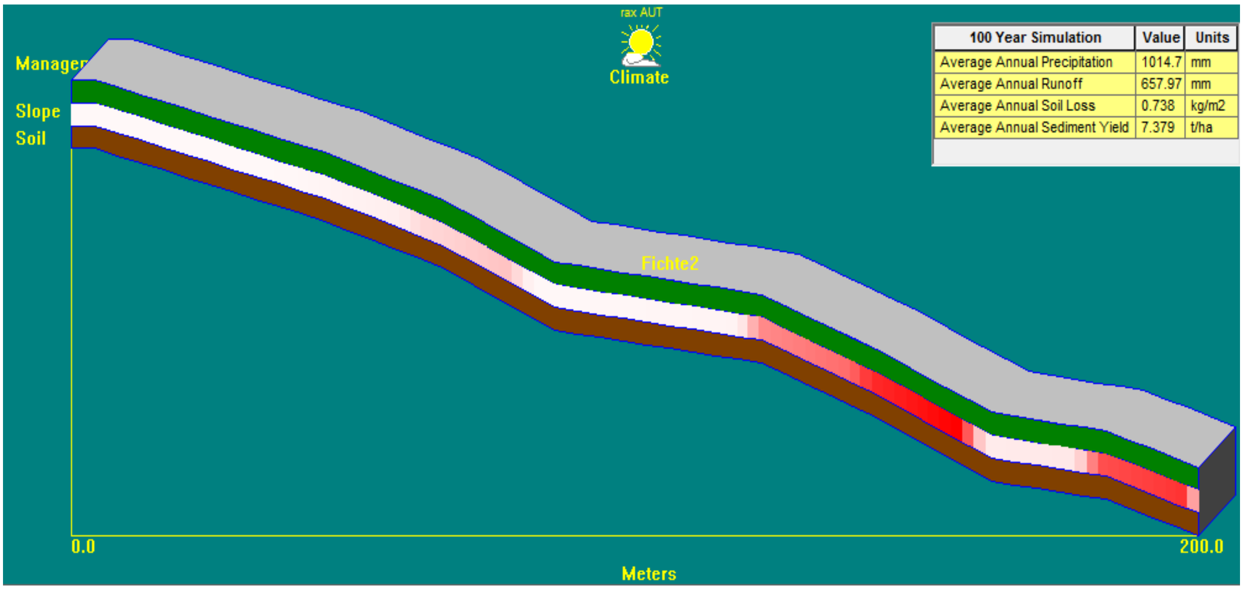

Figure 27 Slope representation in the WEPP user interface

After an intensive period of initialization in order to generate suitable input parameters for the model, it was possible to obtain reasonable results modelling the impacts of a clear-cut in pure spruce stands.

4.3.2.1 Description of Output Variables

The following figures present the behavior of various factors during the course of the simulation period comparing the modelling of continuous cover forestry with the simulation of a clear-cut in year 90 of the vegetation period.

Figure 28 illustrates the development of the canopy height of a spruce stand over 100 years compared to a spruce stand which was harvested in year 90. The instant of time when the clear-cut was performed is clearly visible.

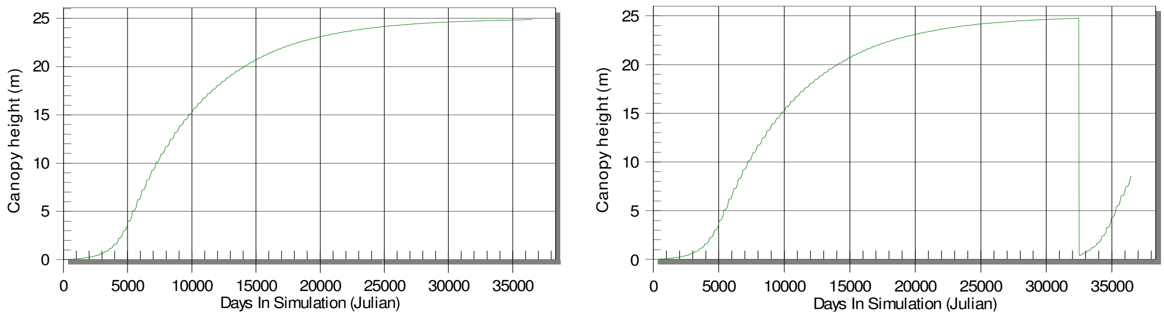

Figure 28 Canopy height of the continuous spruce stand (left) in comparison to the simulation of clear-cut(right) over a 100-year simulation period of WEPP

The development of canopy height (0-25 m) and canopy cover (0-1) of the forest stand represents a similar growth pattern over the years. In addition, leaf area index represents another important factor of stand development. In figure 29 the increase of leaf area index with the beginning of the growth period can be clearly observed. With the clear-cut in year 90 there appears a sudden cut in the right part of the figure.

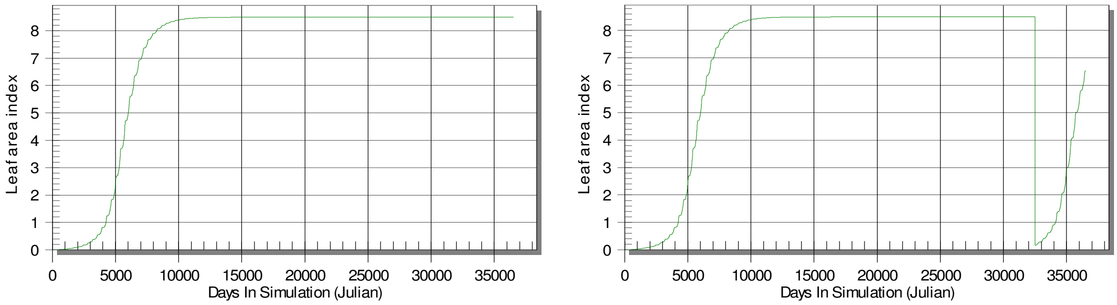

Figure 29 Development of leaf area index in a spruce stand with constant growth (left) compared to a stand with a clear-cut after 90 years (right)

Comparing stand evapotranspiration of the deforested stand and the continuously covered stand as presented in figure 30 the harvest interference in year 90 is also clearly visible.

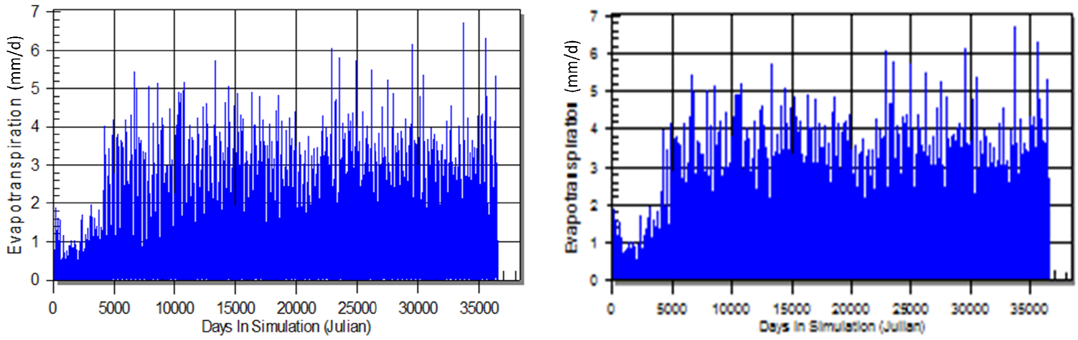

Figure 30 Evapotranspiration rates comparing continuous cover forestry (left) with a stand which was deforested (right) in mm per day

Moreover, the evaluation of changes in runoff according to different forest management practice shows only marginal differences as presented in figure 31.

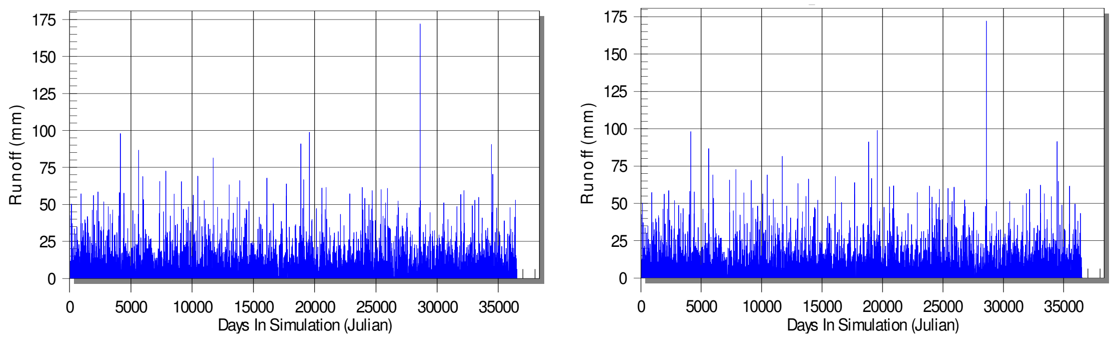

Figure 31 Changes in runoff on a spruce stand without interference (left) compared to that at a stand with clear-cut (right)

Owing to the absence of a clear change of runoff as a consequence of the clear-cut of the whole area, the assumption that runoff is mainly determined by precipitation stands to reason.

Figure 32 illustrates the evolution of sediment yield in the course of plant development simulating one slope being completely harvested in year 90 and another slope without interference.

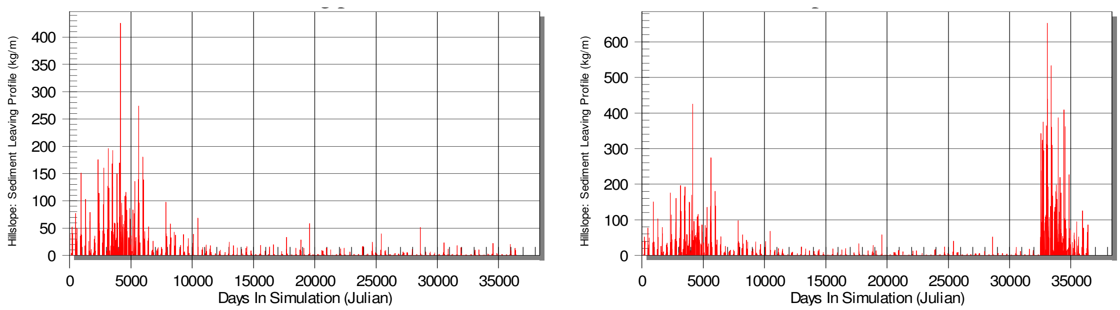

Figure 32 Comparison of sediment leaving the profile in the continuous stand (left) compared to that with a clear-cut in the 90th year of development (right)

There is a clear sign of sediment yield going up immediately after the clear-cut as displayed in the right picture. One problem which still remains concerning the modelling of the spruce stand is clearly visible in these figures. As the initial canopy height is automatically set to zero, I cannot assume that the forest stand is already existent in case of continuous cover forestry at the beginning of the simulation period. Instead, WEPP assumes for both cases that the forest stand starts growing with the start of the simulation. If it was possible to differentiate between the continuous cover forest management including already existing stand structures and the intensive forest management starting the growth period from zero, the difference in sediment yield between these two alternatives would be much bigger. In this case, the peak in the left side of the graphs would not be existent as the continuous cover actually does not imply a starting phase without vegetation cover.

A more detailed evaluation of runoff and sediment yield is going to be presented in the following sections.

4.3.2.2 Impacts of Clear-cutting on Runoff and Sediment Yield

In this analysis different slopes of variable length were studied including slope lengths ranging from 100 to 3000 m. This range is taken due to the tendency of WEPP, which was originally developed for shorter slopes, to overestimate its results on longer slopes. As this study was actually meant to analyse the impact of clear-cuts on larger slopes including the whole range of elevation zones present in the catchment area, an interval until 3000 m of slope length was considered. Additionally, it is possible to simulate changes in runoff and sediment yield resulting out of clear-cuts on a smaller scale with slope lengths from 100 m up to 3000 m.

Table 10 illustrates the absolute as well as procentual increases of runoff, soil loss and sediment yield due to a clear-cut of a 90-year old spruce stand over the whole area simulated on different slope lengths using a simulation interval of 100 years.

Table 10 Synopsis of increase in average annual runoff, average annual soil loss and average annual sediment yield resulting out of a clear-cut of the spruce stand over the whole area, depending on the simulation of different slope lengths in WEPP

Length of slope	Increase in av ann runoff		Increase in av ann soil loss		Increase in av ann sediment yield	
	mm	%	kg/m²	%	t/ha	%
100x20	5.53	0.83	0.40	151.71	3.99	151.69
200x20	8.60	1.31	1.20	162.60	12.00	162.64
300x20	5.06	0.76	1.00	136.53	9.98	136.52
500x20	2.98	0.44	3.10	118.31	30.96	118.36
700x20	4.30	0.65	4.09	96.78	40.86	96.76
900x20	6.70	1.02	5.72	109.79	57.20	109.78
1000x20	11.84	1.78	6.87	109.75	68.65	109.77
1500x20	8.80	1.33	9.32	98.34	93.17	98.34
2000x20	6.07	0.92	10.27	81.08	101.29	81.19
2500x20	6.98	1.12	12.33	84.81	110.91	85.61
3000x20	7.16	1.12	32.72	94.72	130.92	94.71

The increase in average annual runoff does not reveal a reasonable relationship as it is strongly varying. Beyond that also the increases in average annual soil loss and average annual sediment yield do not provide satisfactory results due to a high variability of outcomes. Due to the fact, that these values are generated by averaging over the whole observation interval, a more detailed investigation of the last 10 years of simulation representing those years showing a difference in vegetation, should provide clearer results.
Figure 33 presents the change in average annual runoff generated by WEPP after a clear-cut of the whole spruce stand in year 90 of the simulation period.

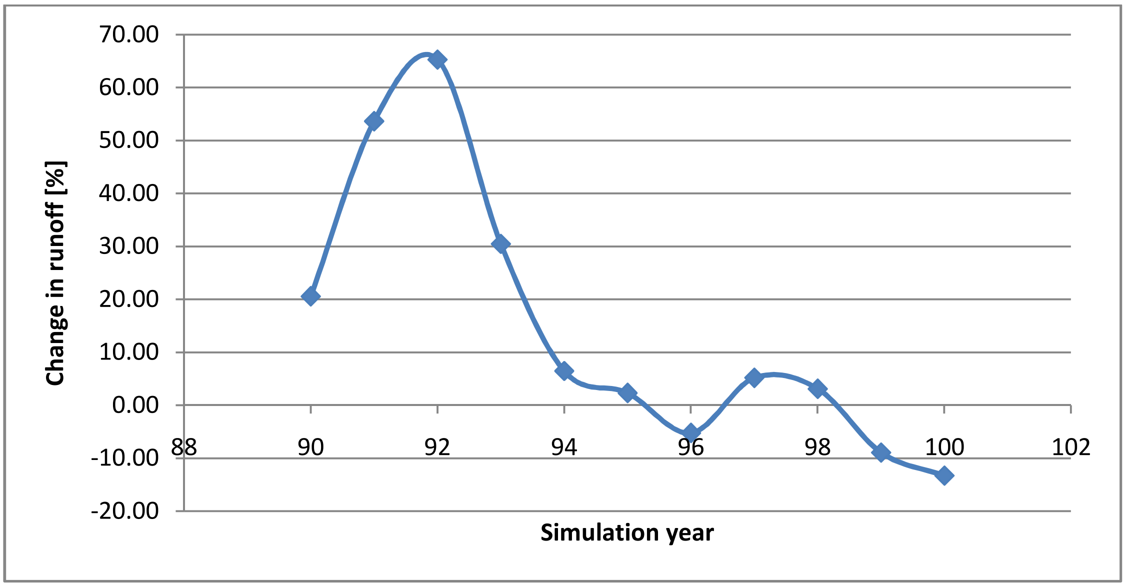

Figure 33 Procentual change in average annual runoff due to different forest management strategies in the years 90 to 100 of the simulation period on a slope of 200 m length

It shows a clear increase of runoff immediately after the harvest interference, which is decreasing steadily after a peak in year 92. In year 96, the runoff decreases which can be traced back to the random effect of temperature and precipitation. After the second increase of runoff in the following years, it again decreases getting negative in year 99 and 100.

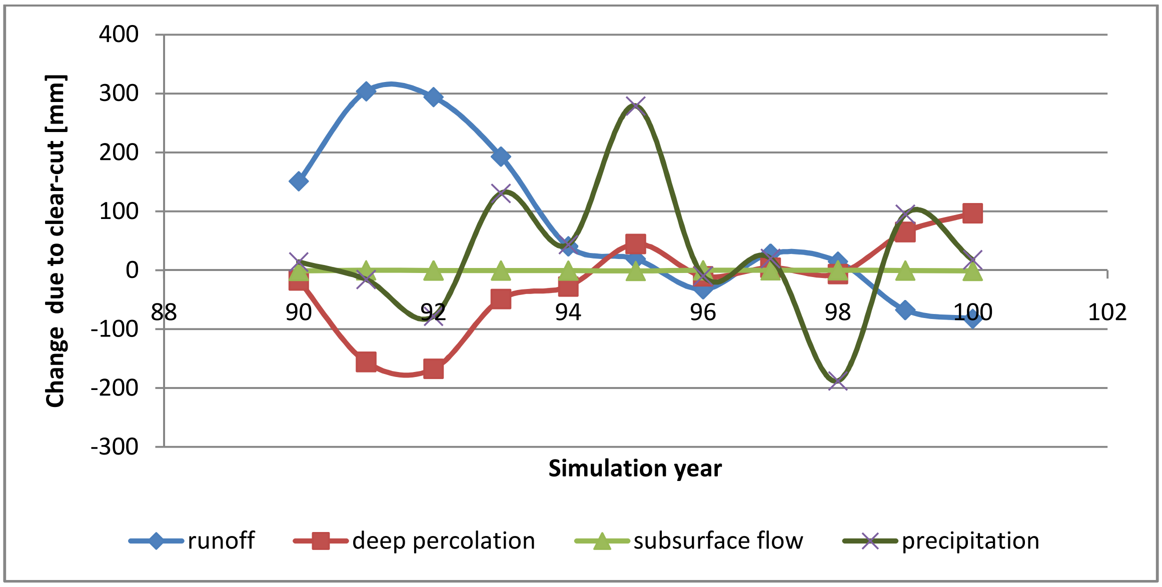

Figure 34 Change of average annual runoff, deep percolation and subsurface flow due to clear-cut of a 200x20m slope in the simulation years 90-100. The precipitation pattern is represented by its annual variation from the average

Figure 34 depicts the two overlapping processes which influence runoff and deep percolation. Firstly, there is the impact of clear-cutting, leading to a comparable high increase of runoff in the first year after the interference which decreases with time.

Secondly, precipitation and temperature affects the change in runoff as shown in this figure. In general, the decrease of deep percolation and increase in runoff agrees with other studies dealing with the impacts of clear-cuts on forest hydrology previously published.

In addition, figure 35 presents the change in sediment yield due to the clear-cut of the forested area.

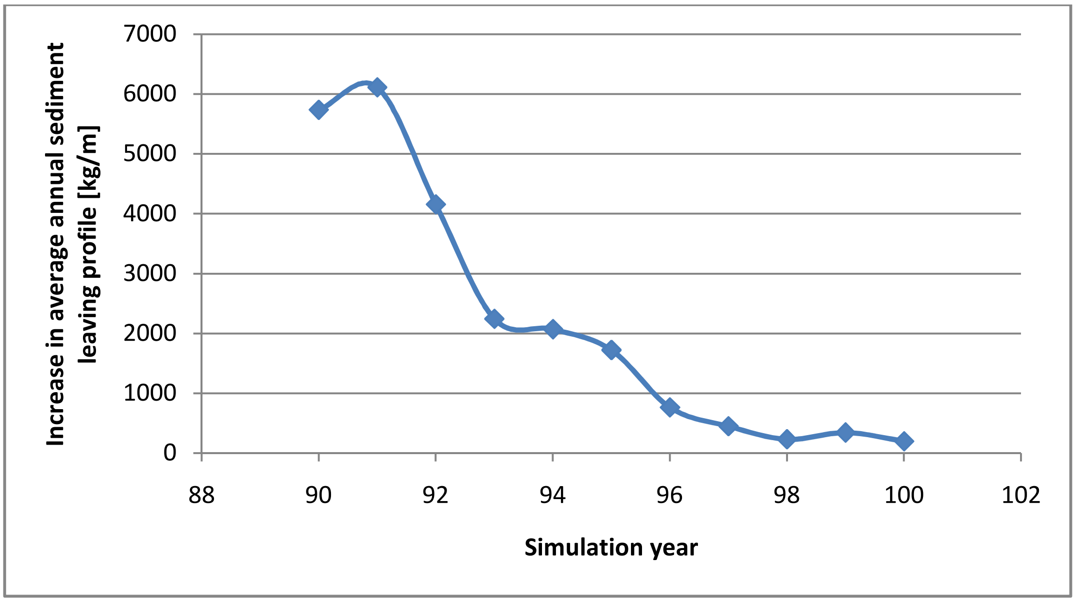

Figure 35 Increase of average annual sediment yield due to clear-cut of 100% of the spruce stand on a slope of 200m length from year 90 to 100 of the simulation interval

It again shows a peak in sediment yield shortly after the deforestation of the slope. In contrast to runoff it still shows augmented levels in the last years of the simulation.

In general, the increase of sediment yield after a clear-cut of a slope of 200x20 m area could be indicated satisfactory. The model also clearly shows an increase in runoff in the first years after the harvest interference. Thus, the results provided by WEPP for the forested catchment area of Vienna coincide with the findings of various authors previously cited.

4.4 Economic Assessment of the Ecosystem Services

The following chapter presents the economic assessment of the ecosystem services of clean water provision in the catchment area. Firstly, the economic assessment of forest revenues, forming the foundation for the estimation of the opportunity costs of various strategies of forest management, is described. To follow up, the application of the replacement cost technique as well as the cost-effectiveness analysis in the catchment area are illustrated.

4.4.1 Economic Assessment of Forest Revenues

This thesis applies two different approaches in order to quantify the opportunity cost of various management alternatives. As described in chapter 3.2.4 the capitalised value approach as well as the marginal income calculation represents different ways to get monetary estimations for intensive forest management in comparison to continuous cover forestry in the catchment area.

In general, the municipal forest department of the city of Vienna would represent the best source for monetary values building the foundation for the estimation of opportunity cost of different alternatives of forest management in this region. Unfortunately it was not possible to get any information concerning yields, costs of harvesting, stand maintenance, etc. and revenues being typical for this area by the responsible authorities. Due to this lack of data material, this assessment is based on the rather general approaches which are described in the following sections.

4.4.1.1 Results of the Capitalised Value Approach

Based on the expertise of Sekot (2012) this case study applies a simplified approach of the capitalised value concept representing an approximate estimation of the opportunity cost accruing due to forest management being solely aimed on water protection. The assumptions which were needed in order to set up the simplified model are based on the counsel of Sekot (2012) and the data collection for the Austrian forest documentation (Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft, 2004) respectively the actual values provided by Sekot (2012).

In order to derive estimates of potential yields in the region of the headwaters of Vienna, the classification of Mayer (1974) was used. Growth region 5.2 can be categorized as Northern edging alpine spruce-fir-beech forest belonging to the Eastern growth district. As a result, the yield table of Spruce-Bruck, using the yield class 8.0, can be applied. The yield table Beech-Braunschweig estimating a yield class of 5.0 was used for beech.

For the reference model of forest management which is optimising revenues a pure spruce stand with a rotation time of 90 years is assumed.

In contrast, the reference model of continuous cover forestry aiming at maximum water protection implies a mixed stand of 50 % spruce and 50 % beech with a rotation period of 120 years for spruce and 140 years for beech.

Additionally, an amount of 20 % of harvest losses is estimated for the conversion of growing stock volumes into sales volumes. The distribution of quality classes is divided into saw timber and pulpwood, where firewood is considered as pulpwood. Stand assortment tables are used for the whole stand where spruce includes saw timber from strength class 1b. Strength classes for beech also include a part of 2b. The prices used for the analysis represent actual average assortment prices according to the price statistics for agriculture 2001-2010 based on the reference year 2010/2011 which were published by the Statistik Austria (2012).

In general, the model does not provide any differentiation between timber harvest cost, management cost or other costs and revenues connected with different tree species or management strategies. This generalisation results out of the fact, that it is not possible to draw conclusions concerning different comparisons of growth in mass in relation to the dimension of timber of various species or costs arising out of more expenses due to artificial regeneration of beech and various implications of such factors.

In this analysis, the indicators of performance of various management alternatives are derived from actual mean values of companies over 500 ha of the test enterprise network of the production area 5 (Calcareous Alps). These mean values include unit cost of timber harvest as well as net costs of forest roads, silviculture, equipment and administration. They form an average of the years 2001-2010. The model used in this analysis refers to idealised, sustainably structured companies using a normal operation class.

Table 11 illustrates revenues which can be obtained due to different forest management strategies.

Table 11 Revenues resulting out of different forest management strategies and varying tree species composition concerning the reference strategy of water protection (database provided by Sekot, 2012)

Reference management: Income [€/ha]	Fraction Spruce/Beech [%]	Reference management: Water protection [€/ha.y]
111.41	100/0	111.35
	70/30	71.32
	50/50	44.64
	30/70	17.96
	0/100	-22.06

It also shows a differentiation between different tree species compositions of spruce and beech. It clearly compares the reference management income always assuming a tree species composition being 100 % dominated by spruce with different amounts of revenues connected with the water protection strategy and the fraction of beech in the forest stands. Revenues arising out of a tree species distribution of 50 % beech and 50 % spruce are used a basis for the economic valuation in this analysis in order to use an idealised strategy of continuous cover forestry.

In addition, figure 36 displays the percentage of revenues obtained by water protection forestry compared to reference strategy to maximise income.

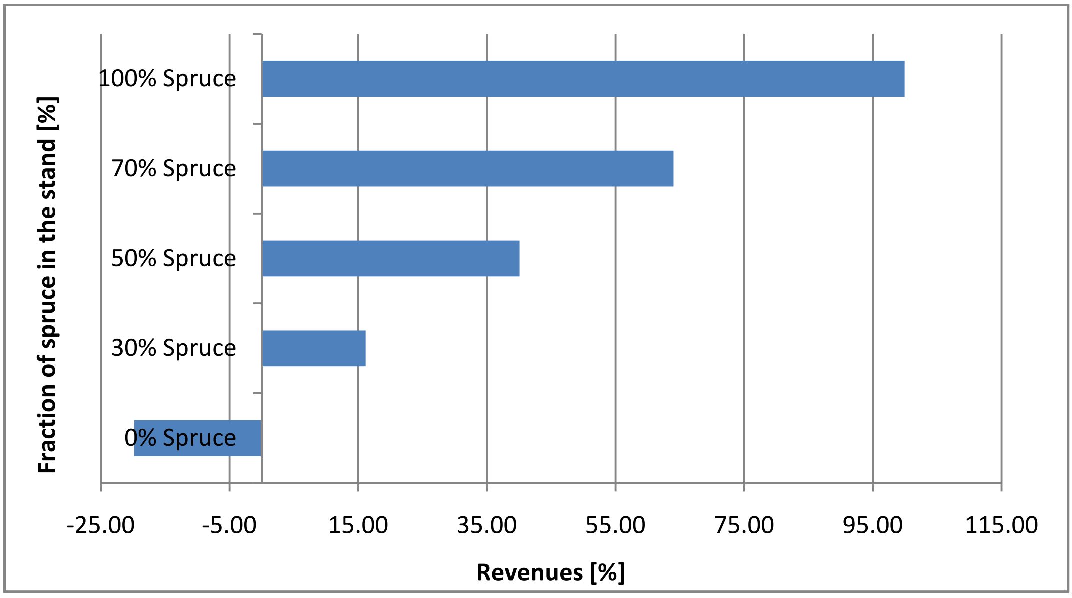

Figure 36 Revenues (in percent) of continuous cover forestry compared to intensive forestry assuming different tree species compositions in the water protection forest (database provided by Sekot, 2012)

To sum up, forest management which is aimed at water protection can achieve considerable revenues to those of an intensive strategy if it is based on spruce dominated stands. The more beech is represented in a stand, the higher is the decrease in profitability of the management strategy. These results are confirmed by previous results of Sekot (2004).

4.4.1.2 Results of the Standard Marginal Income Calculation

Next to the capitalised value approach, the derivation of revenues connected to different forest management strategies is possible using a standard marginal income calculation based on “Deckungsbeiträge und Daten für die Betriebsplanung 2008” published by the Bundesministerium für Forst- und Landwirtschaft, Umwelt und Wasserwirtschaft. This approach provides the selection of a suitable yield class for different tree species and would provide additional results comparable to the results of the capitalised value approach.

However, it is not possible to decide about individual rotation periods or growth rates using the standard marginal income calculation. Additionally, it uses a differentiation between assortment classes being not directly comparable to the one which was used for the capitalised value approach. Beyond that, it was not possible to consistently adapt the prices used in the calculations to the reference year 2011 as there is a differentiation in initial use and final use included. Due to this fact a comparison of resulting marginal incomes of different tree species is rendered difficult and the values generated by the simplified capitalised value approach represent the basis for further analysis.

4.4.2 Results of the Overall Economic Assessment

The aim of the assessment of the ecosystem services provided by the forested catchment of the case study area is the provision of an estimate of the replacement cost on the one hand and of the cost-effectiveness of various alternatives on the other hand. Based on the findings provided by the modelling of forest hydrology and the estimation of the opportunity cost accruing due to different forest management strategies, an approximate estimation of the monetary value of water provided by the headwaters of Vienna should be obtained. Therefore, an evaluation of qualitative as well as quantitative aspects of the water provided by the forested catchment would be necessary. The following analysis concentrates solely on the water quality which can be obtained from the springs in the case study area due to a lack of reasonable data for a monetary estimation of the quantitative services provided by the forest.

As previously discussed, this analysis evaluates the impacts of two different strategies of forest management on the water quality provided by the catchment area of Vienna. In case of a contamination of drinking water, Vienna is able to supply itself self-reliantly for up to four days thanks to storage reservoirs having a maximum storage capacity of 1.5 mio. m³ available. Considering the actual forest management being clearly aimed at water protection, there is no risk of a deficit of clean drinking water exceeding the amount of the storage reservoir. Only if there is a long-enduring contamination which might could occur due to a forest management strategy not being aligned to the goals of water protection, this possibility would have to be considered.
Assuming an increase in erosion linked to intensive forestry three different alternatives can be considered:

- Filtration of turbid water
- Filtration of turbid water in combination with the delivery of water provided by the water work Moosbrunn
- Diversion of turbid water

For the possibility of a filtration of turbid water full costs of 0.3 €.m^{-3} were assumed based on the estimation of the water works of Vienna (Werderitsch, 2011). The same value was used for the second alternative of a combined approach using filtration of turbid water in conjunction with the water work Moosbrunn. This water work is intended to provide additional drinking water in case of high demand and during maintenance works of the spring water mains. It includes two horizontal filter wells providing a delivery rate of 64000 $m^3.d^{-1}$ (Vienna Water Works, 2012b). The mean operating costs for the water work Moosbrunn are set to 0.01 €.m^{-3} (Werderitsch, 2011).

In contrast to the first two alternatives, the diversion of water high in turbidity does not provide a long term solution for the problem. Thus, it represents a rough estimate which would have to be combined with the cost accruing due to other sources for drinking water. For the diversion of turbid water the actual water price of the city of Vienna which equals 1.73 €.m^{-3} is used as it represents the cost arising due to the loss of revenues from drinking water.

The increase of storage reservoirs in order to provide higher storage volumes for the case of water of high turbidity would represent another alternative. It is not included into this analysis due the difficulty in the estimation of any costs associated with this alternative.

Considering all three alternatives, it is assumed that cost for monitoring, infrastructure, etc. remain the same for the Vienna Water Works. In addition, there are no additional costs for turbidity coming up under continuous cover forestry included into the analysis, due to the fact that the turbidity occurs only on a short temporal scale under specific conditions in water protection forests.

The analysis of the replacement cost as well as the cost-effectiveness of different alternatives included two steps. In a first step, relevant values including the full amount of water provided by the forested catchment area of the spring water mains were taken into consideration. The second step included an evaluation only focused on the Kläffer and Schreierklamm springs owing to the fact that they are the two most sensitive towards increased turbidity. Increased sensitivity of these springs can be traced back to the comparably high amount of surface runoff influencing the spring discharge. Table 12 displays an overview of the values of catchment area and spring discharge of Kläffer and Schreierklamm spring which were used as basis for the replacement cost and cost effectiveness analysis. According to the IWHW (2012) it is difficult to provide exact numbers of the catchment area and spring discharge of the Kläffer spring due to the high complexity of this spring.

Table 12 Overview of catchment area and spring discharge of the two springs most sensitive to turbidity(data from the Vienna Water Works, 2009)

	Catchment area [ha]	Spring discharge [m^3/s]
Headwaters of Vienna, forested	9000	
Kläffer spring	5740	3.45
Schreierklamm spring	890	0.42

The forests of the headwaters of Vienna cover an area of 9000 ha. This analysis assumes the whole area to contribute in a similar way to water purification. In fact, an all embracing analysis would have to account for differences between various forms of habitats which can be found in the catchment area stemming from different types of soil, microclimate, slopes, etc. Additionally, this heterogeneity of stand conditions influences revenues which can be obtained from forest management.

In general, this whole analysis is based on the assumption of an increase in runoff and sediment yield as a result of a change from continuous cover forestry to intensive forestry. The foundation for this assumption is provided in chapter 3.1 by the theoretical examination of publications dealing with forest hydrological responses to management impacts as well as the modelling of the hydrologic processes in the catchment area. In this context, the WEPP model could only provide the evidence of an increase in sedimentation due to clear-cuts without being able to reproduce information concerning the amount of increase which can be expected. In addition, the conversion of a spruce monoculture into a mixed stand is not considered in the modelling. Due to these facts, the investigation of the results generated by the replacement cost method as well as the cost-effectiveness analysis has to account for their strong dependency on the magnitude of turbidity resulting from different management alternatives. Beyond that, the change in stand vulnerability concerning windthrow or bark beetle infestations is not included in any modelling approach at all. Thus, a deforestation of large areas owing to the increase of stand susceptibility to external factors may also cause erosion and high values of turbidity.

Figure 37 is based on the spring discharge provided by Kläffer and Schreierklamm spring being the ones most sensitive to turbidity. It shows the cost arising out of continuous cover forestry representing the opportunity cost of not being able to carry out forest management maximizing profits and the three alternatives being linked with intensive forestry which are described above.

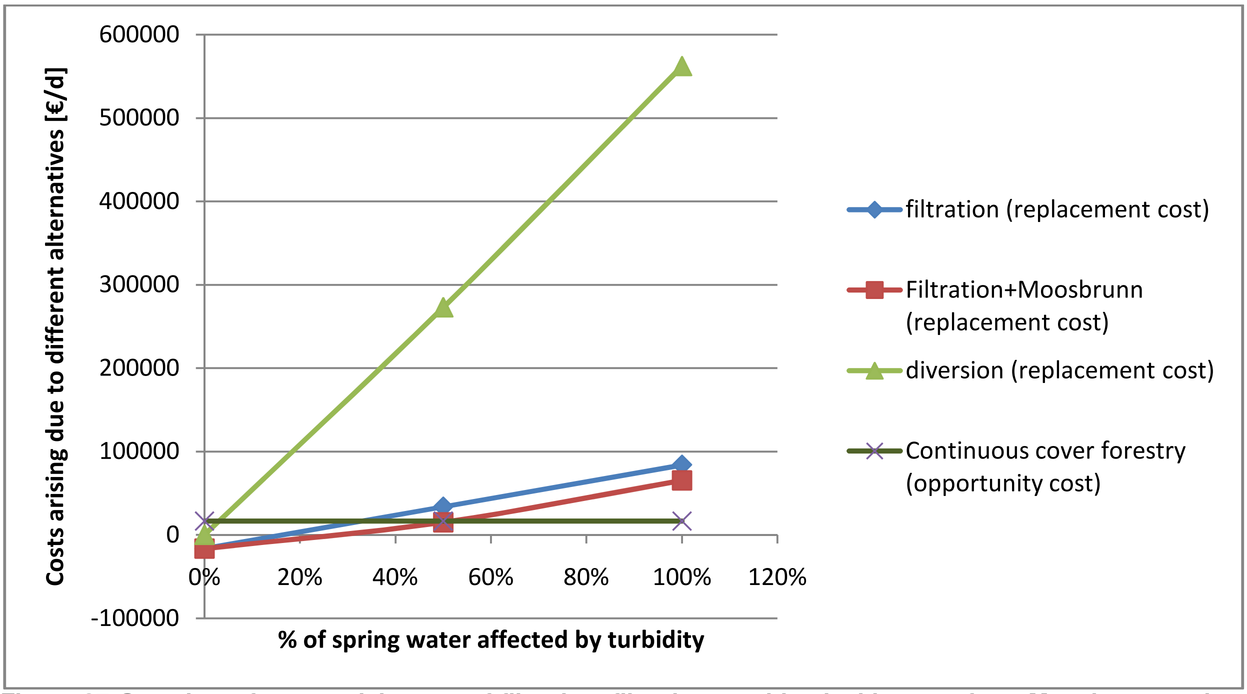

Figure 37 Overview of costs arising out of filtration, filtration combined with water from Moosbrunn and diversion and the opportunity cost of continuous cover forestry dependant on the degree of turbidity of water

Figure 37 reflects the costs arising out of different alternatives assuming a discharge of 3.87 m³/s for the Kläffer and Schreierklamm springs. The forest revenues which are included in the calculation are produced on an area of 90000 ha, which represents the forest area of the whole water protection zone. This figure clearly shows a strong dependency of replacement costs of filtration, diversion or the combined alternative on the degree of turbidity of drinking water. Diversion of turbid water clearly seems to be the most expensive alternative, followed by filtration. The combined approach of filtration and water pumped from the water work Moosbrunn appears to be the most preferable of those three alternatives. Opportunity costs of the water protection forest are solely dependent on forest economics. According to this graphs the combined strategy is more profitable until a degree of turbidity arriving at approximately 50 % of the spring discharge of the Kläffer and Schreierklamm springs. If this threshold is exceeded, replacement costs of any alternative are higher than the opportunity costs of continuous cover forestry.

4.4.2.1 Results of the Replacement Cost Technique

The replacement cost analysis is based on the three alternatives resulting of a deterioration of water quality due to intensive forestry. .

Table 13 presents the values being used in order to calculate monetary estimates of the replacement cost of pure filtration, filtration in combination with the water work Moosbrunn and the diversion per hectare and year.

Table 13 Calculation of replacement cost of pure filtration, filtration combined with water work Moosbrunn and diversion

		Pure filtration	Filtr+Moosbrunn	Diversion
Treated Water	[m³/d]	334368	270368	0
Pumped Water	[m³/d]	0	64000	0
Diverted Water		0	0	334368
Costs				
Filtration	[€/m³]	0.3	0.3	0
	[€/d]	100310	81110	0
Pumping	[€/m³]	0	0.01	0
	[€/d]	0	640	0
Water price of Vienna	[€/m³]			1.73
				578457
Opportunity cost CCF	[€/ha.y]	45	45	45
	[€/d]	11007	11007	11007
Revenues				
Timber	[€/ha.y]	111	111	111
	[€/d]	27471	27471	27471
Total replacement cost	[€/d]	83847	65287	561993
	[€/ha.y]	**340**	**265**	**2279**

Figure 38 shows an overview over the replacement cost of the three alternatives in focus. It assumes that 100 % of the discharge of the Kläffer and Schreierklamm spring is high in turbidity.

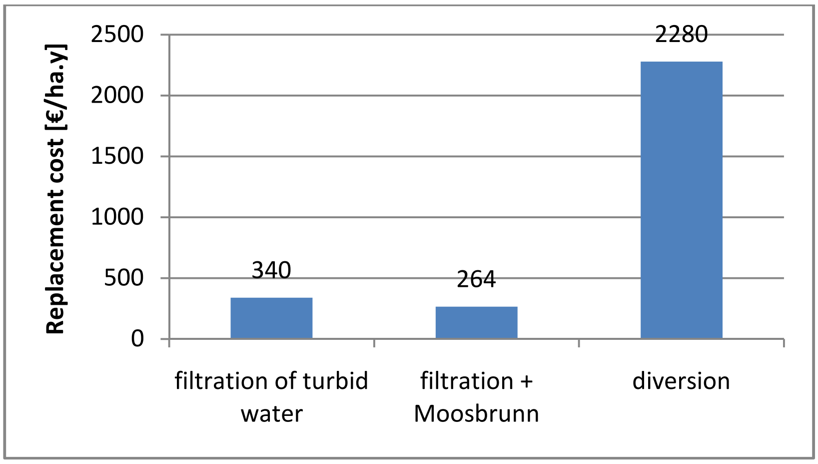

Figure 38 Replacement cost of various alternatives in case of 100% turbid water provided by Kläffer spring and Schreierklamm spring

Finally, the combined approach using a filtration plant and the maximum amount of drinking water which can be provided by the water work in Moosbrunn represents the alternative having the lowest replacement cost of 265 €.ha^{-1}.y^{-1} in case of 100 % of turbid water.

4.4.2.2 Results of the Cost-Effectiveness-Analysis

Based on the replacement cost of the three alternatives the cost-effectiveness of each of them can be derived by dividing annual costs by the physical benefit measure of cubic meters of drinking water obtained. Due to the fact, that the possibility of diversion does not produce drinking water at all and needs to be combined with any other source instead, the cost-effectiveness of this alternative cannot be evaluated. Figure 39 shows a comparison of the cost effectiveness of a full filtration of turbid water, the combination of filtration and pumping from Moosbrunn and the continuous cover forestry assuming a high deterioration of water quality.

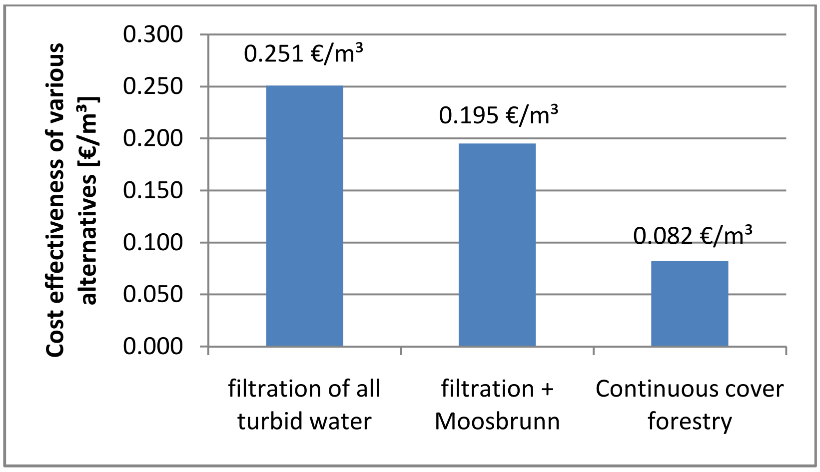

Figure 39 Cost effectiveness analysis of the alternative of filtration, filtration in combination with the water work Moosbrunn and continuous cover forestry

As a result, forest management aiming at maximized water protection represents the most cost-effective alternative. Out of the alternatives arising in case of intensive forestry leading to a severe increase in turbidity in the water provided by the forested catchment area the combined approach of filtration and the water work Moosbrunn represents the most profitable alternative.

5 Discussion & Conclusion

5.1 Discussion

This thesis applied an interdisciplinary approach to the evaluation of ecosystem services of forests in the intake area of the Viennese water works. Therefore, the applied methodologies were best possible adapted to this study in order to provide a high degree of validity and reliability. Nevertheless, the values which were generated in this evaluation need to be used with caution. The results generated during this analysis are based on a fundamental set of simplifying assumptions being dependent on a high number of different factors.
In addition, the selection of a model aiming at the integration of forest hydrology as accurate as possible represents a complex issue as most hydrological models do not offer the possibility to include silvicultural treatments in a highly differentiated way.
In this context, the Water Erosion Prediction Project represents a tool offering a high degree of user friendliness representing its biggest advantage. On the other hand, it was developed for agricultural applications. Due to this fact, the representation of vegetation input parameters is oriented on crop specific values which are often not known or published concerning different tree species. Also the implementation of the dormant season of tree species such as beech was not possible rendering the analysis of mixed forest stands in contrast to spruce dominated stands impossible.
In addition, this evaluation was meant to model a slope including different altitudinal and vegetation zones. Therefore, it was planned to have a slope of 3000 m length being extended over an altitudinal elevation of approximately 1800 m height. As the WEPP model overestimates runoff and sediment yield on longer slopes, a modelling of such a long slope would not have represented the processes in the ecosystem accurately. A highly differentiated approach would have also implied the inclusion of temperature lapse rates as well as precipitation lapse rates. To sum up, the application of WEPP includes a lot of disadvantages. A sensitivity analysis of WEPP using different overland flow elements revealed a difference in the routing in runoff and sediment yield compared to the same land use on an entire slope. The reactions of runoff and sediment yield to an alteration of land use on different overland flow elements could not provide satisfactory results. Thus, the integration of overland flow elements only allows the identification of the importance of buffer strips of vegetation at the end of a slope protecting water quality of subadjent streams, as clear-cuts in the lowest overland flow element lead to biggest soil loss.

In contrast, the Distributed Hydrology Soil Vegetation Model published by the University of Washington would have provided all the modelling possibilities which were not possible in

WEPP. Unfortunately, it was not possible to run this model in the course of this analysis. Nevertheless, DHSVM represents the most suitable model for the representation of the impacts of intensive forestry versus continuous cover forestry on forest hydrological processes including a broad set of forest specific input parameters. In addition, the stand growth routines can be specifically adapted to the conditions which are needed by the user with DHSVM.
Independently of the specific model which is applied, the next step of modelling would represent the integration of forest roads and skid trails in this application. Also an evaluation of the forest roads would have to be based on basic assumptions estimating the density of the road network dependent on different slopes and elevations due to a lack of reliable data. Nevertheless, forest roads can significantly contribute to increases of sediment yield in intensively managed forests, asserting even bigger impacts than harvested areas in most of the cases (Croke and Nethery, 2006).
In order to increase the quality of the understanding of the hydrological processes in the catchment area the application of a combined approach is of major importance. Therefore the modelling of the ecosystem needs to be combined with field experiments in order to provide data for model calibration and validation. This thesis mainly refers to previous findings of studies dealing with the impacts of forest management on below-ground hydrological processes representing an endorsement of the modelling. These impacts cannot be generalized due to the high spatial and temporal variability of hydrological processes and different parameters. Thus, they must be used with care.

What's more, the validity of the economic assessment is strongly restricted by the weak availability of data. Especially the quality of the forest economic assessment is strongly dependent on the availability of realistic assumptions concerning revenues and costs arising due to different forest management strategies in the headwaters of the city of Vienna. Despite to several attempts, it was not possible to gain any estimation dealing with these values from the forest department of Vienna. Further research could also emphasize the alterations of costs-efficiency arising out of different tree species contributions in the continuous cover strategy as this analysis only included the 50 % spruce, 50 % beech alternative.
Moreover, the application of Bayesian belief networks would provide another approach to complement the results generated by the modelling. As it represents a graphical tool to build decision support systems, it offers a simple tool to collect and structure complex knowledge facilitating a holistic system understanding.

In order to include all relevant aspects of ecosystem valuation into the analysis, an extension of the economic assessment to a multiple criteria analysis also appears to provide promising results. The integration of non-economic factors could provide the possibility to e.g. integrate stand vulnerability into the evaluation which could increase the quality of management decisions significantly.

Due to various assumptions integrated in the whole analysis, the results generated in this evaluation should be regarded as lower bound estimates of the specific ecosystem services in focus. It can only provide a frame of value providing a fundamental estimation but not absolute numbers.

In conclusion, the superior aim of any analysis dealing with the services which are provided by an ecosystem should represent the inclusion of all ecosystem services being present there. A holistic analysis would therefore cover the services provided by the catchment area including not only hydrologic services. In the context of the Vienna water works, the first step to enhance this evaluation represents an economic analysis including the values generated owing to the decrease of runoff and the balancing of peak flows in forested catchments.

5.2 Conclusion

In this thesis the ecosystem services of forests were investigated. Its emphasis was placed on the ecosystem services generated by forested catchments for drinking water provision. The main benefits of forests are represented by

- Water storage (buffering of heavy rainfall events)
- Erosion reduction
- Water purification

In order to investigate different land use options, two different forestry management practices were composed to assess these benefits:

- Continuous cover forestry
- Intensive forestry

As a result, continuous cover forestry turned out to be more cost-effective than intensive forestry as the latter one would induce an increase of turbidity. Therefore, the replacement cost of a change of continuous cover forestry to an intensive way of forestry aiming at profit maximization could be assessed using three different alternatives of mitigating a deterioration of water quality:

- Filtration of turbid water
- Combined approach of water filtration and pumped water from the water work of Moosbrunn
- Diversion of turbid water

Finally, the value for the ecosystem service of clean water provision provided by continuous cover forestry arrived at a value of 265 €.$ha^{-1}.y^{-1}$. This value represents the cost which would accrue due to the combination of filtration and Moosbrunn as this alternative would cause least replacement cost.

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7 Appendix

7.1 Modelling

7.1.1 Parameters for Initial Management

Parameters for Initial Management	units	spruce
Initial plant		
Bulk density after last tillage	(g/cub.cm)	0.90
initial canopy cover (0-100%)	%	0
days since last tillage	days	0
days since last harvest	days	0
intial frost depth	cm	0
initial interrill cover (0-100%)	%	100
Initial residue cropping system		Perennial
cumulative rainfall since last tillage	mm	0
initial ridge height after last tillage	cm	0.1
initial rill cover (0-100%)	%	100
initial roughness after last tillage	cm	0.1
rill spacing	cm	0
rill width type		Temporary
initial snow depth	cm	0
initial depth of thaw	cm	0
depth of secondary tillage layer	cm	10
depth of primary tillage layer	cm	20
intial rill width	cm	0
initial total dead root mass	kg/sq.m	0.5
initial total submerged residue mass	kg/sq.m	0.5

7.1.2 Runoff - analysis of the last 10 years of simulation (90-100) forming the basis for figure 36/37.

Runoff [mm]

Year	Spruce	Spruce CUT	differences	change in %
90	733.76	884.57	150.81	20.55
91	565.96	869.49	303.53	53.63
92	450.06	743.81	293.75	65.27
93	632.63	825.33	192.70	30.46
94	625.42	665.71	40.29	6.44
95	827.10	846.04	18.94	2.29

96	620.30	587.63	-32.67	-5.27
97	544.89	572.97	28.08	5.15
98	472.24	486.78	14.54	3.08
99	754.73	687.04	-67.69	-8.97
100	620.76	538.07	-82.69	-13.32
Mean	622.53	700.68	78.14	12.55

7.1.3 Interflow - analysis of the last 10 years of simulation (90-100) forming the basis for figure 37.

Interflow [mm]

Year	Spruce	Spruce CUT	differences	change in %
90	1.07	0.00	-1.07	-100.00
91	0.13	0.00	-0.13	-100.00
92	0.34	0.02	-0.32	-94.12
93	0.37	0.01	-0.36	-97.30
94	0.54	0.00	-0.54	-100.00
95	1.08	0.04	-1.04	-96.30
96	0.18	0.01	-0.17	-94.44
97	0.23	0.03	-0.20	-86.96
98	0.05	0.03	-0.02	-40.00
99	0.48	0.00	-0.48	-100.00
100	1.15	0.14	-1.01	-87.83
Mean	0.51	0.03	-0.49	-95.02

7.1.4 Deep percolation - analysis of the last 10 years of simulation (90-100) forming the basis for figure 37

Deep percolation [mm]

Year	Spruce	Spruce CUT	differences	change in %
90	19.67	2.08	-17.59	-89.43
91	157.69	1.90	-155.79	-98.80
92	185.24	17.82	-167.42	-90.38
93	113.21	64.22	-48.99	-43.27
94	213.22	185.29	-27.93	-13.10
95	151.63	195.95	44.32	29.23
96	160.25	149.90	-10.35	-6.46
97	172.95	177.08	4.13	2.39
98	149.22	142.60	-6.62	-4.44
99	84.35	149.08	64.73	76.74
100	103.85	200.32	96.47	92.89

Mean	137.39	116.93	-20.46	-14.89

7.1.5 Sediment leaving profile – analysis of the last 10 years of simulation (90-100) forming the basis for figure 38

	sediment leaving profile [kg/m]			
Year	spruce	spruceCUT	increase	incr in %
90	11	5745	5734	52736
91	28	6136	6108	21678
92	20	4175	4155	20682
93	7	2248	2242	34298
94	12	2080	2068	17596
95	57	1778	1721	3016
96	15	777	762	5151
97	13	459	447	3553
98	5	234	229	4226
99	41	382	341	838
100	33	230	196	586

7.2 Economic Assessment

7.2.1 Basis for Assessment of costs of different alternatives (figure 41)

Costs of various alternatives	100%	50%	0%	
filtration (replacement cost)	83847	33691	-16464	€/d
Filtration+Moosbrunn (replacement cost)	65287	15131	-16463.8356	€/d
diversion (replacement cost)	561993	272764.484	0	€/d
Continuous cover forestry (opportunity cost)	16464	16464	16464	€/d

7.2.2 Basis for Cost-Effectiveness-Analysis (figure 42)

cost effectiveness of various alernatives		
diversion		
filtration of all turbid water	0.2508	€/m³
filtration + Moosbrunn	0.195253626	€/m³
Continuous cover forestry	0.082157859	€/m³