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Water flow regulation for mitigating river and coastal flooding

Authors: Neville D. Crossman (University of Adelaide, Australia and Murray-Darling Basin Authority, Australia), Stoyan Nedkov (Bulgarian Academy of Sciences, Bulgaria), Luke Brander (VU University Amsterdam, The Netherlands)

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Key issues and challenges

- The water flow regulation ecosystem service can be subdivided into river flood regulation and coastal flood regulation. They are quite different and there are major differences in biophysical processes, scientific disciplines, data, models and methods.
- The measurement of river flood regulation relatively is very well studied, whereas coastal flood regulation much less so.
- Water flow regulation in coastal and inland ecosystems is functionally related to the provision of multiple other services so care needs to be taken with defining ecosystem service boundaries.
- Beneficiaries can be spatially disjointed especially for river flood risk reduction where upstream vegetation mitigates damage downstream – this is a challenge for scale and selection of appropriate spatial units.
- The demand for water flow regulation by ecosystems is determined by the magnitude of the costs of flood risk (the minimised sum of incurring and/or mitigating the damage) which is highly context specific.
- It is not possible to generalise the value of the service using a fixed unit value (e.g. US$/ha/year) because both the demand for and supply of water flow regulation service are highly spatially variable.

1. Description of the ecosystem service

The ecosystem service of water flow regulation to mitigate extreme events is the process of vegetation or other ecosystem structures acting as a barrier or buffer to water flow and thereby reducing the frequency and severity of flood events (citation?). The TEEB (2010) classification defines this service as "Moderation of extreme events". Extreme weather events or natural hazards include floods, storms, tsunamis, avalanches and landslides. Ecosystems and living organisms create buffers against natural disasters, thereby preventing possible damage". This definition is somewhat broader than the service we address in this paper since we examine the regulation of extreme water flows (or floods) only. The Common International Classification of Ecosystem Services (CICES) version 5.1 defines this service as "Hydrological cycle and water flow regulation (including flood control and coastal protection)[1]".

[1] CICES code 2.2.1.3.

This ecosystem service is provided by a wide range of ecosystems. Regarding the regulation of river flooding, the most relevant ecosystems are wetlands and forests in watersheds; regarding the regulation of coastal flooding, the most relevant ecosystems are mangroves, coral reefs and dunes; but also kelp forests, oyster beds, seagrass, and unvegetated sediment. Although type of water flows and the bio-physical processes underlying this service differ between river and coastal flooding, the logic chain linking regulation of water flows to reduced flooding and avoided damage costs is broadly the same in both cases. For this reason, the paper jointly addresses the regulation of water flow for mitigating both river and coastal flooding. Distinctions between the two cases are made where relevant.

In terrestrial ecosystems the presence of vegetation in floodplains and watersheds can reduce the occurrence and severity of flooding by slowing water flows, enhancing percolation and storage, and allowing gradual release of water, thereby maintaining base flows and reducing peak flows. In coastal ecosystems the physical barrier formed by vegetation and other ecosystem structures (e.g. coral reefs and dunes) reduces wave and storm surge impacts. The role of ecosystems in flood control is sometimes overlapping with, or complementary to, artificial structures such as dikes and breakwaters. In such cases it is important to assess the added value of the ecosystem in delivering the service as distinct from the overall service provision.

This ecosystem service is functionally related to the provision of multiple other services so care needs to be taken with defining ES boundaries. For example, a riverine wetland that regulates water flow and flood risk may also deliver more reliable water supply – these are distinct but highly related services. An example for coastal flood regulation is provided by a coral reef that acts as a physical barrier to storm surges and also provides a cultural service in the form of biodiversity that can be viewed while scuba diving – both these services might contribute to the tourism sector but are distinct benefits provided by the reef.

The beneficiaries of water flow regulation for mitigating extreme events are the people that face lower flood risks due to the presence of ecosystems, e.g. households and firms located in exposed coastal areas and floodplains. In the case of coastal flood mitigation, beneficiaries are likely to be in close proximity to the ecosystems providing the service; whereas for river flood mitigation, beneficiaries and ecosystem units may be spatially distant.

In this paper we deal with the ecosystem service of water flow regulation for mitigating both river and coastal flood mitigation. From the perspective of quantifying the economic value of the service, the methods for valuing changes in flood risk are broadly the same. From the perspective of quantifying the biophysical nature of the service, however, there are major differences in terms of biophysical processes, scientific disciplines, data, models and methods.

For clarity regarding the scope of this paper, we do not address the ecosystem service of water flow regulation for the provision of water as an input into consumption or production; nor do we address the ecosystem service of water flow regulation to mitigate soil or coastal erosion.

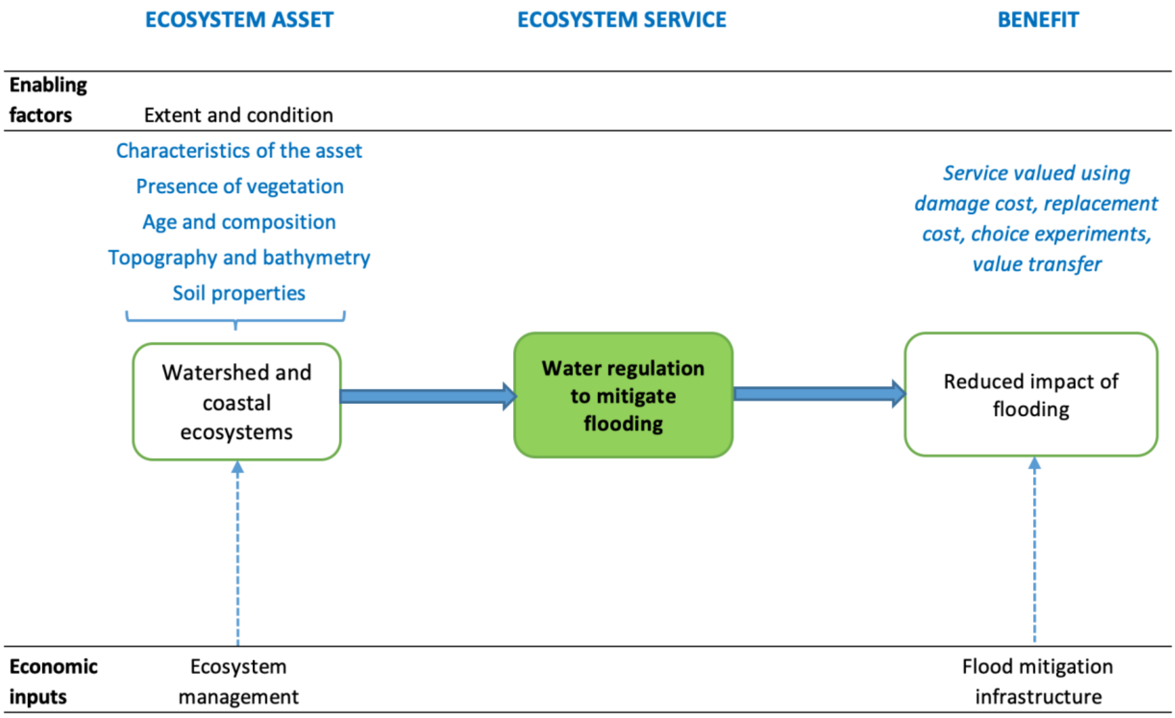

Figure 1. Regulation of water flow to mitigate river and coastal flooding

2. Measuring the ecosystem service

As mentioned in the introduction, river and coastal flood regulation are different in biophysical processes and in applicable methods for their modeling and assessment. Therefore, in this section of the paper they are presented separately.

The regulation of water flows reduces the negative effects of water-related disasters and is one of the most important regulating ecosystem services. A clear distinction between ecological functions, ecosystem services and benefits is difficult to make for mitigation of extreme events through water flow regulation. Therefore, the contribution of ecosystems to flow regulation has to be clearly defined in biophysical terms. De Groot et al. (2002) consider that the ecosystem services derived from water flow regulation are "flood prevention" and "drainage and natural irrigation" as a part of the group of ecological functions "disturbance prevention" and "water regulation". They also emphasize the influence of ecosystem structures on dampening environmental disturbances and the role of land cover in regulating runoff and river discharge. The role of wetlands, near shore marine habitats, floodplains and forests in dampening extreme flood events as well as their role in water infiltration and gradual release of water is linked to the related ecological processes and central components of water cycling (de Groot et al. 2010). They provide natural hazard mitigation by intercepting water flows and influencing water retention capacities (de Groot et al., 2010). Interception depends on the above ground structure of the ecosystem (land cover) while infiltration is strongly determined by the soil properties. Therefore, it is necessary to identify indicators that can represent the regulation function of these elements. The function of the ecosystems to regulate water flow in cases of flood event can be divided into preventing and mitigating. In the first case, the ecosystems (i.e. forests) redirect or absorb parts of the incoming water (from rainfall), reducing in such a way the surface runoff and consequently the amount of rivers discharge. This ecosystem service plays its role before flood occurrence and in some cases it can even prevent it. The role of the vegetation in this case is by collecting water through the process of interception. However, the flood mitigation function comes into effect when the flood is already formed. The ecosystems (i.e. flood plains and wetlands) provide retention space for the water surplus to spill, thus reducing the flood's destructive power (Nedkov and

Burkhard 2012). The vegetation in this case acts as a barrier which redirect and decrease the river flow. These functions are result of different processes and the methods as well as the indicators for their assessment should be different.

In coastal and marine ecosystems, the presence of mangroves and dunes, and near shore structures such as coral reefs, kelp forests, oyster beds, seagrass, and unvegetated sediment provide physical barriers to dissipate wave energy during periods of storm and tidal surges and provide some protection against tsunamis. The properties of these ecosystems that provide the service and benefit are predominantly structural. Mangroves can act as a barrier that reduces the negative affect of flood causing phenomena such as waves, storm surges and tsunamis. They reduce the height and energy of wind and swell waves passing through them, by decreasing their energy as they pass through the tangled above-ground roots and branches. The wave height can be reduced by between 13 and 66% over 100 m of (Guannel et al. 2014) thus the potential flooded area can also be reduced. During storm surges mangroves can reduce the peak water level by 5 to 50 cm, while in case of tsunamis water depth can be reduced up to 30% over 500 m of mangroves (World Bank, 2016). Coral reefs act as natural breakwaters by absorbing wave energy and reducing the flooding effect on the coast. They can reduce the wave energy up to 97% which is mainly due to the dissipation function of the reef crests (Ferrario et al. 2014). Both mangroves and coral reefs have also erosion regulation function which is part of the coastal protection they provide. The methods for measurement of these two services (coastal protection and flood control) could be similar but in this paper we focus only on flood control function.

2.1 Literature review/state of the art

Water flow (or flood) regulation is mentioned in many studies but only few of them focus specifically on that service. Ming et al. (2007) estimate the flood mitigation role of wetland soils in a case study in North-Eastern China by measuring the water content parameters of the soils. Nedkov and Burkhard (2012) utilize the GIS based Automated Geospatial Watershed Assessment (AGWA) modelling tool to assess the flood regulation capacity of ecosystems at watershed scale in Bulgaria. Ryffel et al. (2014) study the land use trade-offs for flood protection using visualizations in stated preference studies. Sturck et al (2014) map the flood regulation supply and demand at European scale utilizing the hydrological model STREAM. Flood regulation has been included in several complex ecosystem service studies that refer to multi-layered relationships between biodiversity and ecosystem services (Mace et al., 2012), the role of the lowland floodplains (Postumus et al., 2010), optimising the outcomes of river rehabilitation (Gilvear et al., 2013), and service providing and benefiting areas (Syrbe and Walz, 2012). Vigerstol and Aukema (2012) provide a review of four tools/models that provide appropriate means for water flow regulation. They compare two traditional hydrological models (SWAT and VIC) and the water related modules of two ecosystem services tools (InVEST and ARIES). The main conclusion from their work is that traditional hydrological models provide more detailed results whereas ecosystem services tools tend to be more accessible to non-experts and can provide a good general picture of the water-related ecosystem services.

The measurement of extreme event mitigation ecosystem service by mangroves and near shore marine ecosystems has been less studied, although there are plenty of studies on the function itself. Tsunami waves, typically 3-18m in height above mean sea level, can reach up to 48m, as was found with the 2004 Indian Ocean tsunami (Choi et al. 2006). Storm surges from cyclones can see waves reach up to 9m above mean sea level, with 3-5m the norm (Marois and Mitsch 2015). Mangrove forests of at least 100m in width can significantly reduce the wave flow pressures (Alongi 2008), and the presence of mangroves between the sea and human settlements can reduce casualties by up to 8% (Das and Vincent 2009, Laso Bayas et al. 2011). Numerical models have shown that mangroves are better at reducing surge heights during faster moving storms (~40 km/hr), but the reduction varies non-linearly with wetland size. Most storm surge or wave height reduction is achieved in the first few hundred metres, with the extent of reduction decreasing exponentially after that (Zhang et al., 2012). During Hurricanes Katrina (2005) and Wilma (2005) it was observed that intact mangrove wetlands reduced storm surge heights by up to 9.4 cm/km inland (Krauss et al., 2009). Isolating the exact role mangroves play in protecting against storms and tsunamis is complicated by other marine and coastal characteristics that also contribute to storm protection. The presence of communities and built infrastructure in areas at risk of storm and tsunami damage also determine the level of benefit provided by mangroves and near shore marine ecosystems (Costanza et al., 2008). Flood protection is an important part of the ecosystems provided by complex coastal regions and its role depends on both the connectivity of natural systems and the complexity of the governance framework (Sousa et al. 2016).

2.2 Methods for biophysical assessment of water flow regulation to mitigate extreme events

Biophysical methods for the assessment of water flow regulation are based on quantification of different parameters of biotic and abiotic structures and can be divided into three main categories according to the character of the measurement and how the necessary information is extracted (Vihervaara et al., 2018). These are direct measurements, indirect measurements and modelling methods.

The direct measurement category includes methods such as field observations and remote sensing, which deliver biophysical values in physical units of the indicator representing the service. In the case of water flow regulation they are applicable for measuring floodplain topography in order to define its capacity to collect water during flood events, measuring the flooded areas using RM data and field sampling of soil water properties to define the soil capacity to hold rainfall water. For example, Ming et al. (2007) collect soil samples to measure saturation water content, residual water content and bulk weight which are used as indicators to define the flood mitigation benefits of wetland soils. In coastal and marine ecosystems, the shape of near-shore bathymetry, the presence of coral reefs offshore, distance inland, elevation above sea level of potentially impacted areas, differences in root and trunk structure and the composition of mangrove ecosystems all influence the level of wave attenuation and mitigation benefits of extreme events (Marois and Mitsch 2015). Measurement of wave height at points with different mangroves in different cases of wave surges enables the calculation of mean wave height reduction (Vo-Luong and Massel, 2006; Bao, 2011). A field study by Stark et al. (2015) measured surge attenuation rates of 5 cm/km to 70 cm/km from the presence of salt marshes in an estuary in The Netherlands. The main advantage of the direct measurements is that they ensure precise quantitative data derived directly from the studied area. However, they are often impractical and expensive beyond the site level, and therefore are usually used as an input for biophysical models or to validate certain assessment elements (Vihervaara et al., 2018).

Indirect measurement methods are also based on biophysical values in physical units but involve further interpretation or data processing. Remote sensing and earth observation derivatives such as standardize indexes derived from satellite images (NDVI, Soil-Adjusted Vegetation Index, Enhanced Vegetation Index) can be used to identify the vegetation cover and its condition as indicators for the capacity of ecosystems to regulate water flow. Spatial proxy methods are a subgroup that includes various methods that rely on certain assumptions, or need to be combined in a model with other sources of environmental information before they can be used to measure ecosystem services. The spreadsheet method (also known as matrix method) is a quick and simple way to get an overall spatially-explicit assessment based on linking tabular and spatial data together. The application typically involves land use or land cover (LULC) datasets, although other datasets can be used. The services can be assessed as expert evaluations or constructed from indicators or statistics. In this case, water flow regulation is usually assessed together with a group of other services. Vihervaara et al. (2010) assess the flood prevention service along with another five regulating services in Lapland; while Schneiders et al (2012) assess flood protection in a spreadsheet of 24 services the in the region of Flanders.

Modelling methods include several groups of modelling approaches (e.g. phenomenological models, macroecological models, process-based models, statistical models) from ecology, mathematics and statistics or other earth sciences fields such as hydrology, climatology, soil science etc. (Vihervaara et al., 2018). Integrated modelling frameworks are also considered here. They include tools designed specifically for ecosystem services modelling that integrate various biophysical, but also social and economic methods. They are usually organized in modules, where each of them is designed for assessment of particular service. The most widely known are InVEST and ARIES. The most relevant modules for river-related water flow regulation are the hydrological models (from process-based models group), which can be used to derive different parameters of the water cycle. These parameters are used as indicators to represent the prevention or mitigation function of the ecosystems in cases of flood events. Nedkov and Burkhard (2012) use surface runoff, peak flow and soil infiltration derived from KINEROS (Kinematic Runoff and Erosion model) as indicators to define water flow regulation supply in mountain watersheds in Bulgaria. Sturk et al. (2014) utilize STREAM (Spatial Tools for River basin Environmental Analysis and Management) model to calculate the river discharge as an indicator for water flow regulation supply in Europe. Each hydrological model has specific requirements to the input data, time step of the modelled parameters, spatial scale and output values. Table 1 gives an overview of the most popular models and tools that are available for quantification of river-based water flow regulation.

Table 1. Models and tools for quantification of river-based water flow regulation (adapted from Vigerstol and Aukema, 2012; Bagstad et al. 2013; Sturk et al. 2014; Palomo et al. 2017)

Model/tool	**Service/function**	**Time step**	**Resolution**	**Scale**	**Platform**
KINEROS (AGWA)	Water yield	Hourly	<50m grid cell	Small watershed	ArcGIS
ArcSWAT	Water yield	Daily	<50m grid cell	Medium to large watersheds	ArcGIS
VIC	Water yield	Hourly to daily	1-50 km grid cell	Large watersheds	Linux
STREAM	Flood regulation	Daily	1km grid cell	Large watershed	Stand alone
ARIES	Flood control	Monthly to annual	30m-10km grid cell	Not specified	Web-based
InVEST	Storm peak mitigation	Annual	30m-10km grid cell	Not specified	ArcGIS/stand alone
LUCI	Flood regulation	Annual	Not specified	Medium to large watershed	Open source GIS toolbox

The biophysical modelling methods for the mitigation of damage from mangrove and near shore marine ecosystems are far less common. A numerical wave and erosion model developed by Guannel et al. (2014) appears as a tool in the InVEST toolbox. Simulation modelling of idealised marshes was used by Loder et al. (2009) to illustrate the effects of wetland continuity and bottom friction on reduction in flood heights. A third-generation wave model SWAN (Simulating WAves Nearshore) was used by Narayan et al. (2010) to simulate waves passing through a mangroves island by calculation of energy dissipation of a wave propagating. The WAPROMAN (WAve PROpagation in MANgrove Forest) model was developed specifically for waves in mangroves to predict typical levels of wave attenuation basen on data on the wave parameters, local bathymetry and topography (Vo-Luong and Massel, 2008).

The application of some modeling methods is limited to particular areas as they require detailed data and extended resources which are not appropriate at national level. Especially with large scale watershed models (such as KINEROS) it is not appropriate to model each watershed individually. For the accounting needs it is necessary to make a typology of the territory and define representative watersheds which can be modeled and the results can be used for value transfer. There are also upscaling techniques which can be used for transferring information from large to medium or small scale. Wigmosta and Prasad (2005) propose four general methods that are typically employed in upscaling including averaging of input data, effective parameters, average model equations, and fully distributed numerical modeling. The modeling methods are also useful for scenario simulations. Most of them have land use/land cover (LULC) as main input and it can be transformed according to particular scenario for development of the area. The models can be run with each different LULC transformation to assess the effect of each scenario on the flood regulation service.

3. Valuation of the ecosystem service

Valuing water flow regulation by ecosystems that mitigates flooding involves measurement of the demand for and supply of reduced flood risk.[2] The focus here is on valuation of the final ecosystem service (water flow regulation) as an input into the production of goods and services (products). The products that utilise flood mitigation as an input in production include both SNA benefits (e.g. housing, transportation, agriculture, etc.) and non-SNA benefits (e.g. non-market recreational use of nature, personal safety).

The *demand* for water flow regulation by ecosystems is defined by the benefits of reduced flood risk. The benefits of reduced flood risk are in turn largely determined by the costs of flooding that are avoided, which comprise of two distinct components: 1. Damage costs to assets and people in the event of a flood; and 2. Mitigation costs including flood protective infrastructure/measures (e.g. levies, dikes, seawalls, beach nourishment), relocation, and avertive/defensive behaviour (e.g. growing flood resistant crops). It is important to recognise that economic units (households and firms) faced with flood risk will attempt to minimise the sum of these two cost categories and that the mix of cost-minimising responses to flood risk is highly context specific. In brief, demand for flood mitigation by ecosystems is determined by the magnitude of the costs of flood risk (the minimised sum of incurring and/or mitigating the damage).

The *supply* of water flow regulation by ecosystems is defined by the capacity to provide the service and the cost of doing so. In some cases, land may be managed for the purpose of delivering water flow regulation (e.g. maintained dunes, designated areas for flood water retention) and the costs are relatively well understood. In many cases, however, the provision of water flow regulation by ecosystems is an uncompensated public good, in which case the level of supply is determined by other considerations (i.e. private land use decisions, government regulation, protected area designation etc.) and the costs of delivering the service are largely unknown. Nevertheless, for the purposes of estimating the value of the service it is necessary to quantify the cost of delivery, which is generally measured as the opportunity cost of the land on which ecosystems are present (i.e. the next highest value alternative use of the land).

With information on the demand for and supply of water flow regulation for an ecosystem unit, it is possible to compute the welfare value of the service as the difference between the benefits and costs (for a given change in flood risk resulting from a change in the ecosystem extent and/or condition). To estimate the exchange value involves multiplying the quantity of water flow regulation (e.g. change in probability of a flood event that is attributable to the ecosystem) by the price of the service, which

[2]Flood risk is defined as a function of flood hazard (the frequency, extent, depth and duration of inundation), exposure (the assets and people located in the flooded area) and vulnerability (the extent of damage in the event of a flood).

is challenging since this requires an assumption on the institutional nature of a market for the service and derivation of a simulated price (Caparrós et al., 2015). Note that the estimated demand (avoided damage and/or mitigation costs) gives an upper bound to the price since it would be irrational for an economic unit to pay a price higher than the expected benefit of water flow regulation; and the estimated supply (opportunity cost of land on which the ecosystem unit is located) gives a lower bound to the price since the price should at least cover the cost of providing the service.

A substantial challenge in estimating welfare and exchange values for water flow regulation provided by ecosystems is that both the demand for and supply of this service are highly spatially variable, meaning that it is not possible to generalise the value of the service using a fixed unit value (e.g. US$/ha/year). Spatial variation on the demand side is driven by the location of people and assets exposed to flood risk and the costs of mitigation; and on the supply side by a wide range of bio-physical factors determining the potential role of ecosystems in regulating water flows (slope, soil and rock type, coastal bathymetry etc.) and the opportunity costs of land. The spatial dimension of this service is further complicated by the geographic separation of ecosystem unit producing the service and the beneficiaries of the service (i.e. located downstream or inland from the ecosystem unit) (Stürk et al., 2014).

Figure 2 represents the spatially variable determinants of supply of and demand for water flow regulation by ecosystems and the consequentially high spatial variation in demand, supply and value of this service.

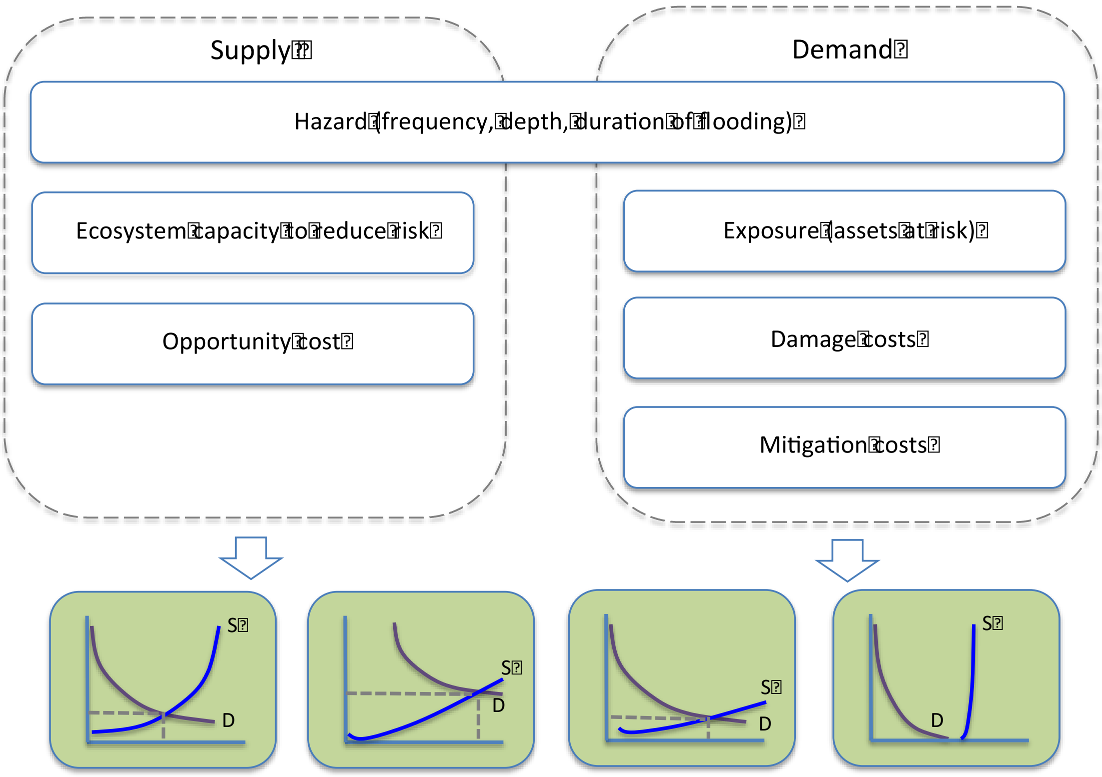

Figure 2. Spatially variable determinants of demand for and supply of water flow regulation by ecosystems

3.1 Review of valuation literature

Many studies have estimated values for water flow (or flood) regulation by ecosystems. We use data from four global meta-analyses for wetlands (Brander et al., 2013), forests (Hussain et al., 2011), mangroves (Brander et al., 2012a) and coral reefs (Brander et al., 2012b) to provide a brief synthesis of the literature. We note, however, that there are a large number of studies published in recent years that are not included in these databases. Table 2 provides a summary of the number and magnitude of value estimates for water flow regulation by inland wetlands, forests, mangroves and coral reefs. Value estimates have been standardised to a common set of units and year of value, namely US$/hectare/year in 2007 prices. A key observation from this summary of estimates is the very large variation in values, both across and within ecosystem types. This is indicative of the spatial variation in demand for and supply of this service and substantiates the point that it is not advisable to use fixed unit values for this service in accounting applications.

Table 2. Estimates of the value of water flow regulation ecosystem service (US$/hectare/year; 2007 price level)

	Inland Wetlands	Forests	Mangroves	Coral Reefs
N	41	6	17	16
Mean	79,396	172	24,953	8,631
SE of Mean	70,610	115	22,651	2,955
Median	744	6	1,953	1,298
Minimum	1	1	1	15
Maximum	2,901,786	682	387,266	34,292

Stürk et al. (2014) develop an approach to measuring spatial variation in demand and supply for water flow regulation and apply it at the European scale. The measurement of demand and supply, however, is in the form of biophysical indicators rather than monetary units of benefits and costs. The analysis therefore does not deliver estimates of economic value but does take a step towards quantifying spatial variation in demand and supply, which is necessary to value this service.

In the context of coastal water flow regulation by coral reefs, Van Zanten et al. (2014) develop and apply an analytical framework for spatial assessment and valuation of coastal protection services by coral reefs. This framework links a series of models to spatially assess hazard, exposure, vulnerability and the value of damage. The approach is not explicitly defined in terms of demand and supply for the flood protection service but contains many of the elements needed to estimate exchange values.

3.2 Methods for valuing water flow regulation

Commonly applied methods for valuing water flow regulation that mitigates flooding are the damage cost approach, replacement cost method, hedonic pricing, choice experiments and value (benefit) transfer.

The damage cost method (also termed “damage cost avoided” or “expected damage function”) estimates the value of damages that are avoided by the presence of ecosystems that reduce flood risk. This method is an adaptation of the production function methodology of valuing the environment as an input into a final benefit (Barbier, 2016). In this case, the ecosystem service of water flow (or flood) regulation is an input into the production of infrastructure, housing, crops and public health. The damage cost method requires data on (i) the population, property, and human infrastructure at risk from flood damage, and (ii) the probability of damages given the estimated frequency of flood events, and (iii) determination of the extent of protection provided by natural ecosystems (Salcone et al., 2016). If there is a reduction in ecosystem extent or condition, the resulting welfare loss is measured as the total increase in expected damage from flood events. A limitation in the general application of this approach is that no account is taken for risk aversion or stress related to flood exposure, i.e. the material cost of flood damage may not fully reflect the welfare cost (Freeman, 2003). A potential extension of this approach, motivated above, is to estimate avoided costs more generally (damage and/or mitigation costs) since it is unreasonable to assume that an economic agent would suffer flood damage if there are lower cost mitigation options available.

The replacement cost method estimates the value of flood protection provided by ecosystems as the cost of replacing the service with human-built infrastructure (e.g. dams, dikes, seawalls etc.). An extension of this approach is to estimate the cost of restoring a degraded ecosystem to an extent and condition that it provides the original level of water flow regulation (i.e. flood protection). The replacement cost method can provide lower bound estimates of the value of an ecosystem service, but only if the following conditions are met: (1) the human-built infrastructure provides the same level of service as the ecosystem being replaced; (2) the human-built infrastructure should be the least-cost alternative; and (3) there should be substantial evidence that the service delivered by the infrastructure would be demanded by society if it were provided at cost (Shabman and Batie, 1978). In practice, most applications of the replacement cost method do not meet these conditions and tend to greatly over-estimate the value of ecosystem services (Barbier, 2016). This is because the cost of infrastructure is not a good proxy of the benefits that it delivers (benefits can be lower than costs if the infrastructure is redundant); and the selected replacement infrastructures used in many studies are not the least-cost alternative. The replacement cost method is widely used due to its relative convenience (costs of human-built infrastructure are widely available) (World Bank, 2016) but when used inappropriately, delivers misinformation on the value of ecosystem services.

The hedonic pricing method is a revealed preference method that estimates the value of component characteristics of a marketed good (Freeman, 2003). It is often applied using house price data to estimate the value of individual characteristics of a property including location dependent environmental characteristics such a noise, air pollution and proximity to green open space (Brander and Koetse, 2011). It has been used to a limited extent to estimate the value of exposure to flood risk (Landry et al., 2003). A challenge in applying the hedonic pricing method to value environmental characteristics such as flood risk exposure is that house buyers need to be fully aware of variation in the characteristic across properties, which not necessarily the case. A potential further source of information on the value of flood risk exposure to property is variation in insurance premia.

The choice experiment method (also termed “discrete choice experiment” or “choice modelling”) is a stated preference method that involves asking respondents in a public survey to make repeated choices between alternative options that are described in terms of a set of characteristics including some form of price. By observing the choices that respondents make, it is possible to estimate the relative importance of each characteristic and compute trade-offs between them. The trade-off with price gives an estimate of willingness to pay for a specified change in the quantity of each characteristic. There is a growing use of this approach to estimate willingness to pay for changes in flood risk (e.g. Botzen and van den Bergh, 2012; Brouwer et al., 2009; 2013; Hagedoorn et al., 2019). This information can be used to estimate demand for water flow regulation and to simulate market prices.

Value transfer (also termed “benefits transfer”) is the use of research results from existing primary valuation studies at one or more “study sites” to predict welfare estimates or related information for other locations or “policy sites” (Brander, 2013a). This approach is widely used for estimating the value of water flow regulation services but can result in inaccuracies due to differences in the bio-physical and socio-economic contexts of study and policy sites. In cases where study and policy sites are highly similar, simple unit value transfer may be sufficiently reliable. In cases where study sites and policy sites are different, or the application is at a landscape scale for multiple ecosystem units, value function or meta-analytic function transfer offers a means to systematically adjust transferred values to reflect variation in factors determining demand for and supply of water flow regulation (see Brander et al., 2012; 2013b for applications).

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