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Viewpoint

Enhancing the potential value of environmental services in urban wetlands: An agro-ecosystem approach

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ABSTRACT

This paper proposes a three-tier process for supporting policy planning of urban agroecosystems. It comprises the following steps: (i) definition of the agro-environmental unit; (ii) measurement of the non-market values; (iii) estimation of opportunity cost. An application to an urban wetland agro-ecosystem within Mexico City is used for illustrating our methodology. We estimated that the wetland agro-ecosystem has a lower-bound monetary value between $15.6 million and $31.5 million USD/ha/y. As the land conversion rate is about 3.73 ha/y, the opportunity cost would be between $22,300 and $44,900 USD/ha/y. Such figures are an objective way to appreciate both the potential enhancement value and the opportunity cost of ecosystem services adjacent to urban areas, providing both urban and environmental policy guidance. We argue that this framework allows for multi-scale analysis and may be applied for other urban ecosystems as well.

Introduction

Agro-ecosystems depend on natural environments from which productivity is enhanced in a sustainable way. In contrast to intensive agriculture, they provide not only food but ecosystem services as well (Porter, Costanza, Sandhu, Sigsgaard, & Wratten, 2009; Sandhu, Wratten, Cullen, & Case, 2008; Zhang, Ricketts, Kremen, Carney, & Swinton, 2007). Most agro-ecosystems are either directly or indirectly linked to urban developments. Indeed, urbanization not only refers to increased paved area, but also implies higher demand for natural resources and ecosystem services. Yet, ecosystem services associated with agro-ecosystems or other modified landscapes are poorly understood (Sandhu et al., 2008).

Increasing urban areas not only threaten agro-ecosystems, but other fragile ecosystems such as wetlands (Ehrenfeld, 2000; Lee et al., 2006). For example, Faulkner (2004) describes the main effects on forested wetlands by urbanization, chiefly habitat fragmentation and hydrological and biochemical changes. Such effects might alter agricultural productivity as Hussain and Badola (2008) have demonstrated for mangrove forests in adjacent agricultural land. In fact, wetlands are fragile ecosystems, and their importance is reflected by the fact that they are the only ecosystems protected under an international convention (Turner et al., 2000). Furthermore, according to Costanza et al. (1997), wetlands are the most valued ecosystems in monetary terms, reaching almost 15,000 USD/ha/y. Hence, losing wetlands area implies an opportunity cost to society because wetlands supply a number of ecosystem services in cities, such as: air filtering, micro-climate regulation, noise reduction, rainwater drainage, sewage treatment and recreational and cultural values (Bolund & Hunhammar, 1999; Breaux, Farber, & Day, 1995; Bystrom, 2000; Ehrenfeld, 2000). Additionally, adjacent agricultural land receives benefits as well (Hussain & Badola, 2008). One way to enhance such effects is by means of ecological restoration (Benayas, Newton, Diaz, & Bullock, 2009) and, in the case of agro-ecosystems, by sustainable agricultural practices (Sandhu, Wratten, & Cullen, 2010).

As the rate of land conversion is high, rapid assessment of economic valuation is needed for environmental policy recommendations in urban planning (Faulkner, 2004). This should be a priority due to the increasing importance of agriculture, the increasing loss of ecosystem services, and the potential for agro-ecosystems to enhance global ecosystem services (Porter et al., 2009). In fact, recent work has shown that ecosystem services provided by either wetlands (Tong et al., 2007; Turner et al., 2000) or agro-ecosystems (Porter et al., 2009; Sandhu et al., 2010) are under-valued. Hence, there is still a need of recognizing the value provided by ecosystem services in watersheds where both rural and urban settlements depend on water provision and other services (Postel & Thompson,

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2005). This has important policy implications, as Porter et al. (2009) demonstrated, since non-market ecosystem services contribute between 50% and 70% of agro-ecosystems total economic value in the EU, suggesting that European agricultural systems should move toward enhanced ecosystem services/agro-ecosystems production.

We agree with Ehrenfeld (2000) that urban wetlands have strong differences from those in the wild, and therefore, specific environmental policy planning should be granted to such ecosystems. Therefore, in this paper we present a case study which we consider useful for illustrating a three-tier process for supporting policy planning of such ecosystems, especially when they are directly linked to agro-ecosystems (Hussain & Badola, 2008). Thus, our paper presents three steps for guiding policy planning for urban and agro-ecosystems wetlands. We argue that sustainable agricultural practices and ecological restoration might lead to enhancement of environmental services value. The three steps are:

1. Defining an agro-environmental unit.
2. Estimating ecosystem services values.
3. Estimating the opportunity cost.

We performed our assessment in Xochimilco wetlands, which is an illustrative example of urban wetlands inexorably linked to an important agro-ecosystem, located within one of the major metropolitan areas in the world: Mexico City. Our paper is thus organized as follows: the next section briefly describes Xochimilco wetlands; this is followed by a section focused on methods and another containing our results and discussion. In the latter section, we offer some policy and planning recommendations.

Xochimilco: an urban wetlands agro-ecosystem

Xochimilco is a rural-urban sector in southern Mexico City where traditional agriculture and several ecosystem services are supplied by means of "chinampas". These are plots where traditional agriculture has been carried out for at least six centuries and used to cover a large extension of what is now Mexico City. During the last decades, intensive agriculture (e.g. greenhouse-based) as well as urban development, have shrunk the chinampas area to about 2600 ha. Several efforts have tried to preserve their natural and cultural values. For example, UNESCO designated Xochimilco a World Heritage Site in 1986; moreover, a natural protected area designated as "Ejidos de Xochimilco y San Gregorio Atlapulco" was declared in 1992, and it is listed under the Ramsar Convention on Wetlands since 2004. A more detailed account of both Mexico City's and Xochimilco's context is given in Aguilar (2008) and Wigle (2010), respectively.

The Xochimilco freshwater ecosystem is composed of channels connecting small lakes and a main wetland. This system is a tropical high altitude water body, which produces distinct hydrological and ecological regimes (Zambrano, Contreras, Mazari-Hiriart, & Zarco-Arista, 2009). The hydrological regime at Xochimilco is marked by substantial seasonal change as the rainy season results in substantial ecosystem expansion due to formation of temporary wetlands that attach to permanent water bodies. It is important to biodiversity as it hosts migratory birds and endemic species of amphibians, fish and crustaceans (Valiente, Tovar, Gonzalez, Eslava-Sandoval, & Zambrano, 2010).

Anthropogenic perturbations have been imposed on this dynamic hydrologic regime. Indeed, recent studies have shown that land use is a strong driver of water quality (Zambrano et al., 2009), ecosystem energy paths (Zambrano, Valiente, & van der Zanden, 2010) and biodiversity distribution, such as the Mexican axolotl, an endemic endangered amphibian (Contreras, Martínez-Meyer, Valiente, & Zambrano, 2009). In order to preserve ecosystem services provided by this agro-ecosystem, conservation and restoration policies must be implemented, considering the high heterogeneity in water quality produced by the regional climate, as well as contrasting land uses.

We believe that our three-tier assessment process is worth trying in an agro-ecosystem, which supplies a number of ecosystem services to millions of people. Moreover, our case study can be useful for guiding both urban and environmental policy and planning in urban ecosystems and agro-ecosystems elsewhere.

Methods

Theory

As stated above, Xochimilco wetlands (actually, an agro-ecosystem) provide ecosystem services with both direct (i.e. market values) and indirect use values (i.e. non-market values). On the one hand, direct values refer to assets traded in formal markets, as in the case of agricultural production. On the other hand, non-market values refer to environmental assets without market prices; such as water infiltration and depuration, biodiversity existence, carbon sequestration or cultural and religious importance. According to Sandhu et al. (2008) and Porter et al. (2009), the total economic value of ecosystem services for agro-ecosystems is given by the sum of both market values and non-market values of ecosystem services.

Market values of ecosystem services are estimated simply by the market prices of produce but non-market values of ecosystem services imply indirect estimates of environmental valuation. Contingent valuation methods are an acceptable approach for assessing environmental non-market goods. However, the requirements and assumptions for having robust values imply high costs and, depending on the issue, high sampling effort. Nevertheless, environmental policy decisions frequently need broad estimates that help decision-making in a short span. Hence, alternative valuation methods are warranted. For a review of methods for estimating the value of ecosystem services in wetlands see, for example, Barbier, Acreman, and Knowler (1997) and Brander, Florax, and Vermaat (2006).

We limit our analysis to three main non-market values of ecosystem services in Xochimilco: water quality improvement, carbon sequestration and endemic biodiversity. Water infiltration, although a chief ecosystem service in most wetlands, it is not particularly significant in our area of study due to a highly impervious aquitard in Xochimilco's underground (Serrano, Perevochtichikova, & Carrillo-Rivera, 2008). We are aware that leaving aside important ecosystem services such as microclimate regulation or cultural amenities will result in lower estimates of total economic value. However, we reckon that giving a first baseline (i.e. minimum level) estimate of monetary value is a useful policy instrument. Indeed, as Barbier et al. (1997) points out, valuation should not be considered as an objective but rather as a policy instrument.

Our method comprised three steps: (i) definition of the agro-environmental unit; (ii) measurement of the non-market values; (iii) estimation of opportunity cost. Detailed explanation follows.

Calculation

Please note that details of all calculations are presented in a worksheet file as Supplementary material.

Step 1: defining an agro-environmental unit

As a first step in our analysis, we defined an agro-environmental unit where measures of ecosystem services in both physical and monetary terms could be standardized. Such unit was an experimental restored plot (i.e. chinampa) where sustainable agricultural practices (i.e. assuming cultivation with 100% composting) and ecological conditions would enhance ecosystem services, including biodiversity conservation (Valiente et al., 2010).

The experimental agro-environmental unit is located inside the Natural Protected Area of Xochimilco in a site named "Texhuiloc" (N19°16.502′, W099°05.447′). Restoration began in 2009 after 25 years of abandonment. Transversal trenches were dug out in order to allow a constant water flow between adjacent channels (Fig. 1). Such water flow enhances agricultural production and naturally forms refuges for endemic fish, amphibians and crustaceans (Valiente et al., 2010).

The exact surface area and the volume of water contained in the canals were calculated with the formula of a trapezoid, which is the closest shape of both the actual agro-environmental unit and its associated channels. For the latter, this process was repeated in several segments due to the different depth and width levels. The exact measures of the agro-environmental unit are given in Table 1 and detailed in the Supplementary material.

All calculations made for the experimental agro-environmental unit were standardized to 1 ha and then extrapolated for the whole Xochimilco's chinampas area in order to facilitate policy planning.

Step 2: measuring the non-market values of ecosystem services

Monetary values for three ecosystem services were estimated: water quality improvement, carbon sequestration and endemic biodiversity. First, water quality improvement was estimated by means of the replacement cost method. Such method has been widely used in estimating the ecosystem services of wetlands (Birol, Karousakis, & Koundouri, 2006; Brander et al., 2006). For example, the replacement cost method for assessing water quality improvement has been applied by Bystrom (2000) in Sweden and Dehnhardt and Brauer (2008) in Germany. It is a useful proxy for monetary valuation of water quality improvement, mostly when assessments deal with specific water quality standards (Birol et al., 2006) or when the general public is not familiar with ecological functions, such as nutrient removal (Dehnhardt & Brauer, 2008; Meyerhoff & Dehnhardt, 2007). In fact, unfamiliarity of the general public respondents toward ecosystem functioning is a drawback of "preference stated methods" for monetary valuation of ecosystem services (Barkmann et al., 2008). In contrast, a drawback of the replacement cost method is the fact that no welfare estimates are given (Brander et al., 2006). However, it has been frequently used in environmental policy analyses. For example, Barbier et al. (1997) consider it a useful method when first-best methods are not available. For recent reviews on replacement cost method applications on wetlands and other ecosystems see: Sundberg (2004), Brander et al. (2006) and Hougner, Colding, and Söderqvist (2006).

Table 1
Agro-environmental unit measures[a] for the Texhuiloc plot.

	Surface (m^2)	Volume (m^3)
Farmland	9792	–
Composting facilities	253	–
Water trenches	1655	502
Adjacent channels	2522	616

[a] Detailed computation of estimates is given in a spreadsheet as Supplementary material.

According to Hougner et al. (2006) and Meyerhoff and Dehnhardt (2007), the replacement cost method requires us to choose a technological substitute with similar functions as the natural ecosystem services and with the lowest cost among alternatives. Hence, we used the total cost of the first year of investment and maintenance of a constructed wetland, which ranges between \$0.02 and \$0.12 USD/m^3 (Mazari, Jimenez, & Lopez-Vidal, 2005). Given that such values corresponded to 2001, we adjusted prices with the producer price index (II. Secondary economic sector, 4. Construction) of the Bank of Mexico (www.banxico.org.mx, visited on August 22, 2011). We multiplied the estimates by the total water flow volume of the agro-environmental unit (Table 1).

Carbon sequestration was estimated assuming composting as the main agricultural practice in the agro-environmental unit. In order to know how much carbon was sequestrated from composting production in the agro-environmental unit, we adopted the estimate of Fronning, Thelen, and Doo-Hong (2008) for compost global warming potential ($CGWP = -1{,}811.00\ gCO^2/m^2/y$). Such estimate is obtained by measuring how organic Carbon increases with different soil experimental treatments, including composting (Fronning et al., 2008). It is worth noting that the CGWP is negative since composting mitigates both CO^2 and GHG. We used the CGWP absolute value in order to multiply it by the international spot price of Carbon under the Clean Development Mechanism (www.bluenext.eu, visited on February 22, 2011) at \$16 USD/tonCO^2. Hence, we obtained a rough estimate of the potential value of carbon sequestration by assuming 100% composting at the agro-environmental unit. For allowing price uncertainty, we assumed a range of price between \$12 and \$20 USD/tonCO^2.

Finally, to estimate the endemic biodiversity value, we chose the Mexican axolotl due to its status as an emblematic endemic species in the region since Aztec times (Valiente et al., 2010). We assumed that one axolot in ideal conditions would inhabit one squared meter of channels (Valiente & Zambrano, Personal observations). Thus, we multiplied the area of adjacent channels

Fig. 1. The Texhuiloc plot at Xochimilco, Mexico City. Left: the plot before restoration after 25 years of abandonment. Right: the same plot after trenches were dug and sustainable practices were set up. Photos by Elsa Valiente.

and water trenches (Table 1) by one and by the local market price ($4–$8 USD a piece). Indeed, the Mexican axolotl is occasionally sold in aquariums and locally consumed. Such estimate is not the existence value of an axolotl. Instead, its market value is assumed as a lower bound of the total economic value of biodiversity.

Step 3: opportunity cost estimation

Once we had the estimates of the three main ecosystem services, we summed them up and standardized the values from the actual agro-environmental unit to 1 ha. We then multiplied the value for the number of total area in Xochimilco, which is 2614 ha (Merlin-Uribe, 2009). This allowed us to estimate the value of ecosystem services lost to green-housing facilities, which represent the first step towards definite urbanization.

Finally, market values (i.e. agricultural production value) were also computed for comparison purposes. These were simply estimated by using the actual value of agricultural production with an overhead price for agricultural produce.

Results and discussion

Ecosystem services values

The resulting estimates of ecosystem services monetary values are presented in Table 2. Market values ranged from $9,821.42 to $12,807.13 USD/ha/y and non-market values from $5,966.90 to $12,036.95 USD/ha/y. Grand total ranged from $15,788.32 to 24,844.08 USD/ha/y. The estimated economic value of Xochimilco's agro-ecosystem services is within the range of previously estimated values elsewhere. For example, monetary use-values for Mexican wetlands in Campeche state mangroves have found to be about $2000 USD/ha/y (Lara-Dominguez, Yanez-Aranciba, & Seijo, 1998) and about $37,500 USD/ha/y for Gulf of California mangroves (Aburto-Oropeza et al., 2008). The monetary value estimated by Costanza et al. (1997) for global wetlands was $14,785.00 USD/ha/y. While no monetary estimates of ecosystem services in agro-ecosystems have been calculated in Mexico, a lower bound of $998 USD/ha/y is estimated by Porter et al. (2009) for cereal cultures in Denmark and a value of $19,420 USD/ha/y was found by Sandhu et al. (2008) for organic crops in New Zealand.

In our case, biodiversity represented the highest ecosystem service in monetary value. It contrasts with the findings of Bräuer & Marggraf (2005), Brander et al. (2006) and Tong et al. (2007), who found that water quality improvement was the highest valued ecosystem services of wetlands. Our estimate for the value of water quality improvement (from $31.92 to $239.43 USD/ha/y) is lower in comparison with other studies. This may be due to the method we used, which reflects the low cost of constructed wetlands in Mexico.

Table 2
Economic values[a] estimated for the agro-environmental unit in Xochimilco wetlands (USD/ha/y).

	Lower bound	Upper bound
Market values		
Potential organic production	9821.42	12,807.13
Non-market values		
Water quality improvement	31.92	239.43
Biodiversity	5717.66	11,435.32
Carbon sequestration	217.32	362.20
Total (non-market values)	5966.90	12,036.95
Grand total	15,788.32	24,844.08

[a] Detailed computation of estimates is given in a spreadsheet as Supplementary material.

Hence, if the chinampas area in Xochimilco comprises 2614 ha, its ecosystem services monetary value would range between $15.6 million and $31.5 million USD/ha/y.

These estimates have to be considered as a lower bound of the total economic value, since other ecosystem services were not included, and no preferences assessment was conducted. Among the ecosystem services we did not include is water infiltration. It is, nevertheless, frequently recorded as a valuable ecosystem service in most wetlands (e.g. Brander et al., 2006). Infiltration in these wetlands is negligible due to a highly impervious aquitard in Xochimilco's underground (Serrano et al., 2008). Furthermore, other species were not included in our analysis. Therefore, our estimate does not provide a full accounting of biodiversity value but just a lower bound. Cultural values were not included either. If these were computed, the total economic value would be even higher.

With respect to use (market) values, organic agriculture would have a higher price than the actual traditional production in Xochimilco. The total value of agricultural production is reported by INEGI (2005) at about $9,821 USD/ha/y. According to Torres and Trapaga (1997), organics in Mexico are priced about 30% above traditional produces. Using this estimate would increase the value to about $12,807 USD/ha/y (Table 2). However, comparisons between organic and traditional production is difficult because total yields depend on agro-chemicals input versus compost fertilization. We argue that organic agriculture is capable of supplying more ecosystem services than conventional agriculture (Sandhu et al., 2010) and also has the potential of supplying more food than previously thought (Badgley et al., 2006).

A drawback of the replacement cost method is the fact that no preferences are assessed and, therefore, no welfare estimates are given (Brander et al., 2006). However, according to Birol et al. (2006), the replacement cost method might provide a lower bound of willingness to pay if certain assumptions are met. Hence, our results might be useful for helping further research on monetary valuation of ecosystem services. For example, estimates might serve as a basis for points of departure in contingent valuation questionnaires. In spite of such shortcoming, Porter et al. (2009) and Tianhong, Wenkai, and Zhenghan (2010) agree that ecosystem services valuation is of major importance to environmental policy. Although predicting policy impacts on wetland functioning is rather complex (Turner et al., 2000), we argue that, in our case, opportunity cost projections might be a useful way to guide environmental policy in urban settings. As ecosystem services valuation is difficult due to ecosystems complexity (Bockstael, Freeman, Kopp, Portney, & Smith, 2000; Chee, 2004; Limburg, O'Neill, Costanza, & Farber, 2002), combining contingent valuation methods in order to estimate demand functions, and replacement cost method could prove useful for environmental policy issues (Bräuer & Marggraf, 2005; Hougner et al., 2006). However, the cost of carrying out a contingent valuation on a frequent basis would be too expensive for urban or environmental policy objectives in megacities, such as Mexico City.

An important aspect of restoration efforts is the investment cost, mostly in order to know whether such costs do generate environmental benefits, and in what magnitude (Holl & Howarth, 2000). In the case of the Texhuiloc plot, restoration costs amounted to about 17,000 USD/ha/y (detailed costs are given in the Supplementary material). Even when this amount seems to be high with respect to environmental benefits in the Texhuiloc plot, we have to stress the fact that our estimates are, as stated above, a lower bound of the total economic value, because existence values were not accounted for. We are aware that it is a limitation in our study, but, as Aronson et al. (2010) point out, there are too few available studies on restoration, environmental services valuation and policy implications, and ours tries to link all three aspects, although in a

Table 3
Monetary values and potential opportunity cost estimations[a] in Xochimilco wetlands, Mexico City.

Total surface of "chinampas" area in Xochimilco (ha)		2614.00
Monetary value of ecosystem services in the "chinampas" area (USD)		
	Lower bound	Upper bound
	15,597,482.70	31,464,575.26
Area devoted to green-housing facilities in Xochimilco (ha)		126.60
Opportunity cost of losing ecosystem services due to green-housing (USD)		
	Lower bound	Upper bound
	755,409.84	1,523,877.29
Annual rate of conversion from "chinampa" to green-housing (ha/y)		3.73
Annual loss in monetary terms of ecosystem services (USD/ha/y)		
	Lower bound	Upper bound
	22,256.55	44,897.81

[a] Detailed computation of estimates is given in a spreadsheet as Supplementary material.

preliminary way. In the next section, we discuss some policy implications of our study.

Environmental and urban policy guidance

In the case of Xochimilco, the ecosystem services provided as both wetlands and potential organic crops, represent an important value for millions of inhabitants in Mexico City. Particularly, sustainable practices, such as composting, keep important ecosystem services such as carbon sequestration and contribute to biodiversity conservation. However, in dealing with various wetlands, such sustainable practices are rarely considered by policymakers in many developing countries. Indeed, agro-ecosystems management is a convenient way for conciliating agricultural production and ecological conservation.

Policy prescriptions using objective indicators are a useful way for guiding decision-making in urban and environmental policy and planning. For example, we argue that monetary values give a common language for urban planners who have to decide between urban settlements or preservation areas; for agricultural producers who have to decide between intensive or sustainable agriculture; or even for urban inhabitants who have to decide between gaining paved areas of losing ecosystem services. Monetary estimations of non-market values are meant to give comparable figures between policy options. For example, decision-making would be facilitated by introducing not only market values, but the monetary value of ecosystem services in methodologies such as Analytic Hierarchy Process (Vaidya & Kumar, 2006), where identification of environmental assets with higher opportunity cost would be given priority in policy design. Table 3 shows how opportunity cost considering non-market values can provide insights for decisions concerning urban planning, including the trade-off between ecological restoration, green-housing or urban settlements. We can illustrate this issue by focusing our case study.

In spite of some efforts in urban planning for stopping urbanization in Xochimilco (Wigle, 2010), land conversion occurs at a high rate. During the 1960s, the urbanization rate in the Xochimilco sector was about 5.9%, reaching a peak of 8.6% in 1980. In 2000, the figure was 3.6% (Merlin-Uribe, 2009). Between 1990 and 2000 greater Mexico City's growth rate was about 2.9% (Tortajada, 2006). For example, as the area already converted to greenhouse facilities is 126.60 ha, the opportunity cost would range between \$0.75 million and \$1.52 million USD/ha/y. Accordingly, as the conversion rate of land, from chinampas to green-housing facilities is about 3.73 ha/y, the opportunity cost would range between \$22,300 and \$44,900 USD/ha/y. Such figures are indeed, an objective way to appreciate both the potential enhancement value and the opportunity cost of ecosystem services adjacent to urban areas. Furthermore, agro-environmental units might prove useful for estimating both economic and ecological values in order to provide such urban and environmental policy recommendations, allowing multi-scale analyses.

We believe that a three-tier process for guiding decision-making for urban ecosystems, as the one developed in this paper, could be useful in assessing the opportunity cost and therefore guiding urban and environmental policy and planning. This study is a first attempt to estimate ecosystem services under an agro-ecosystems approach in this important area to Mexico City. We have analyzed the case of an agro-ecosystem but our approach might be well applied to other urban-environmental units for assessing non-market values in cities, such as parks, natural protected areas, green roofs, waterways, reservoirs, seashore, and other urban ecosystems. Although it is difficult to generalize on ecosystem services due to the diversity of urban ecosystems (Bolund & Hunhammar, 1999), more information is needed on the effects of wetlands urbanization (Faulkner, 2004; Zhang et al., 2007). Thus, further research and discussion is warranted.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.cities.2012.08.002.

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