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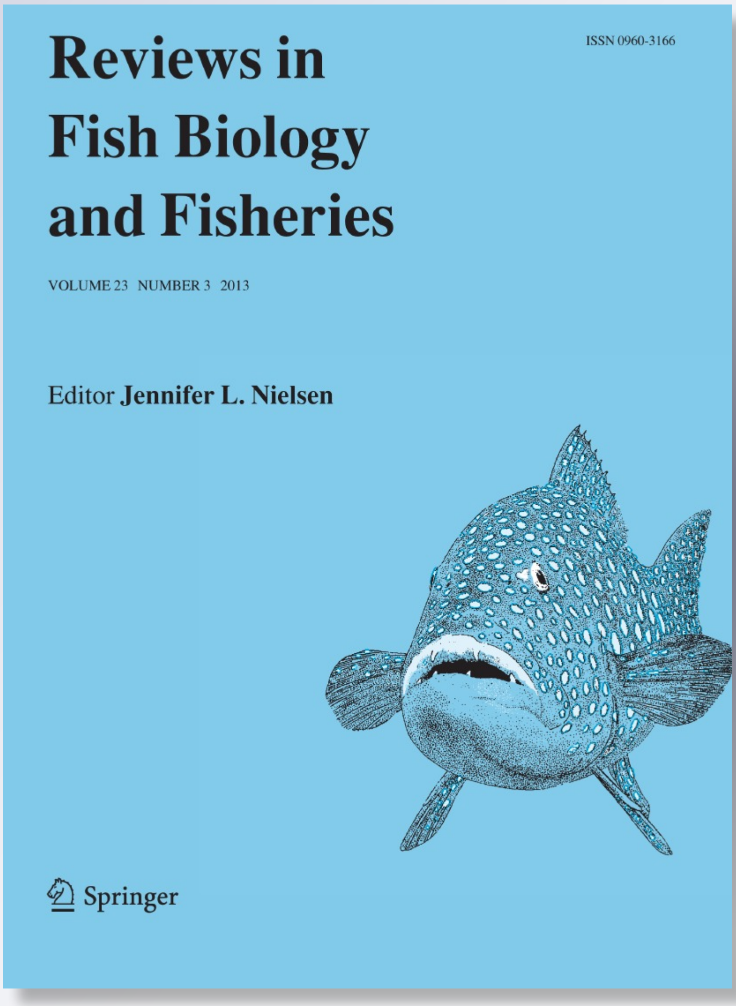

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RESEARCH PAPER

Hydrological attributes and rheophilic freshwater fish: stock assessment

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Abstract The mathematical and statistical advances in fitting stock assessment models enabled the emergence of the paradigm of "integrated analysis", which fits all available data to a single model of population dynamics that traditionally has total catch as the only forcing function of the system. This approach, however, allowed us to include, in a flexible way, the effect of hydrological regime as an additional forcing function. We tried to achieve this flexibility by making the annual recruitment rates and spawning biomass adjustable to the attributes of the hydrological cycle data. Our models showed that these attributes are influential in the population dynamics of *Brycon hilarii* of the Northern Pantanal, and their inclusion in the models allowed best partial fits (which considered fits only to the data components length- and age-compositions, CPUE of juveniles and adults) than the Base-case (without hydrological attributes). The best partial fits where obtained when the attributes "delay of floods" and "intensity of floods" were forcing the spawning biomass and the annual recruitment respectively, indicating that these characteristics of the population may be influenced by specific attributes of the water level. The use of integrated modeling contributed with the advancement of population ecology knowledge of rheophilic fish. It is recommended that freshwater fisheries management be integrated into the hydrology management.

Keywords Fisheries · Cuiabá River · Floodplain · Inland fisheries · Migratory fish · South America

Introduction

The two last decades of the 20th century were entitled by Quinn (2003) as "The Golden Age" for the assessment of fish stocks due to the substantial advances in computational, mathematical and statistical tools for building and fitting population dynamics models. These advances enabled the emergence of the paradigm of "integrated analysis" (Fournier and Archibald 1982) that comprises fitting all available data set to a single model that formalizes the hypotheses about the population dynamics and has the catches as forcing function. The development of this paradigm has enabled for the emergence of flexible stock assessment models (Ralston and Ianelli 1998) such

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as the Stock Synthesis Model (SS, Methot 2005). Integrated analysis allows for fitting population dynamics models to data as diverse as mark and recaptures data, age- and size-compositions and abundance indices while also including external information such as parameters with fixed values, penalties and prior probability density functions for some parameters. However, those advances were mainly done for marine fisheries, and more often, for industrial fisheries (Quinn 2003).

Although inland fisheries play an important role in food security and poverty alleviation, particularly the artisanal fisheries in developing countries (World Bank et al. 2010; FAO 2010), the datasets available for assessment of exploited continental stocks are unsatisfactory both in quantity and quality (Welcomme 1990). Poor and scattered data, or lack of data altogether, impairs the use of sophisticated analysis such as integrated analyses to assess the status of the stocks and to explore hypothesis about the dynamics of the populations. Seeking to change the current scenario, the Brazilian freshwater ecosystems have been the subject of some specific initiatives of fisheries monitoring and data collection, mainly related to requirements for the environmental licensing for hydropower dam constructions, but also as a result of LTER—Long Term Ecological Research projects, which leveraged the beginning of a new phase for assessment studies (Alves and Minte-Vera 2012).

The feature that makes most large- and medium-size rivers unique systems for fisheries is that they have floodplains associated with them that are essential habitats for juvenile fish. These ecosystems are periodically inundated by flood pulses, the main driving force influencing the biotic and abiotic factors of these environments (Welcomme 1979; Junk et al. 1989; Junk 1997). In these systems, the floods provides greater availability of resources, triggers reproduction events of many aquatic species, influences positively the survival and growth of early stages of fish and impacts the population regulation and recruitment (Agostinho et al. 2001, 2004).

The aim of this study was to fit integrated stock assessment models for a migratory fish species of a large tropical river which is also targeted by fisheries, assuming both catches and the flood pulses as the system forcing functions. The models were implemented in software Stock Synthesis (SS-V3.10, Methot 2005). The stock of *piraputanga* (*Brycon hilarii*, Characidae) of the Northern Pantanal was used as a case study. Pantanal is the largest continuous floodplain of the world comprising an area of about 140.000 km^2 flooded by the Paraguay river, in the Midwest region of Brazil (Souza et al. 2006; Britski et al. 2007). Pantanal is a highly important area for biodiversity conservation (Harris et al. 2005), recognized as a Wetland of International Importance by the Ramsar Convention of Wetlands Areas in 1993 and as a Biosphere Reserve and Natural Heritage by UNESCO in 2000. Like many large river ecosystems (Nilsson et al. 2005), this region is experiencing a growing number of dam constructions. For example, Northern Pantanal, our study area, had its hydrological dynamics changed since 1999, when the Manso dam was implemented.

Methodology

Study area

This study was conducted in the Cuiabá river basin, Mato Grosso State, between the cities of Barão de Melgaço and Nobres (coordinates 57°00′–54°40′W and 14°18′–17°00′S). The Cuiabá and the Paraguay Rivers are the main tributaries of the Pantanal of Mato Grosso. The Cuiabá river is formed by the confluence between the Manso and Cuiabazinho Rivers and covers an area of about 850 km, which drains a watershed of about 100,000 km^2 (Severi 1999). The climate in the region has a hot and rainy summer (average 27.4 °C in December) and dry winter with relatively mild temperatures (average 21.4 °C in July) (Severi 1999; Girard et al. 2003). The geological formation of the Cuiabá River Basin and its characteristic climate determine a seasonal pattern with periods of high rainfall, when floods hit the upper Pantanal of Mato Grosso (Severi 1999).

The Manso River is the main tributary of the Cuiabá River and has about 200 km of total extension to the confluence with the Cuiabá River (Netto et al. 1993). The Manso River had its natural flow interrupted by the impoundment of the Area of Multiple Use of Manso dam (*Área de Aproveitamento Múltiplo de Manso*—APM Manso) in November of 1999 which flooded upstream a stretch of the Manso River and the low portions of the Casca, Palmeiras and Quilombo Rivers.

History of fishing

Currently three types of fishing take place in Pantanal: subsistence, commercial-artisanal and recreational (sport) (Catella et al. 2008). Recreational fishing in the Pantanal attracts large groups of fishermen from across the country. Sport fisheries would typically impose larger fishing mortality than other fisheries upon the migratory predators such as *pintado* (*Pseudoplatystoma corruscans*, Pimelodidae), *barbado* (*Pinirampus pirinampu*, Pimelodidae) and *dourado* (*Salminus brasiliensis*, Characidae). In contrast, the omnivorous *piraputanga Brycon hilarii* it targeted almost exclusively by the commercial-artisanal fisheries (Netto and Mateus 2009).

In 2008 total production of inland fisheries in the state of Mato Grosso was 6,566 t (Brazil 2010), corresponding to about 24 % of the inland fish production of Brazil, which is the tenth inland fisheries production worldwide (FAO 2010). In 1997 the “Conservation Plan of the Upper Paraguay River Basin” stated that in 1995 commercial-artisanal fisheries catches of *piraputanga* reached 28.3 t in the State of Mato Grosso. This amount may be underestimated due to the lack of systematic monitoring of landings in the region (Mateus and Estupiñán 2002). The first program of systematic monitoring of landings in the State of Mato Grosso was the “System Monitoring and Control of Fishing in Mato Grosso—SISCOMP/MT”, which was implemented in 2006 (Catella et al. 2008).

The “Fishing Policy of the State of Mato Grosso” (State Law 9.096/2009) states that the fisheries management for both commercial-artisanal and recreational fisheries shall be designed around closed seasons (called “*defeso*”), restrictions on gear types, size limits and bag limits. The closed season runs from November to February, during the “*piracema*”, which is the fish reproductive migration. Gillnets are forbidden on all times and the fisheries are based on hook-and-line gear only. The size limit for the capture *piraputanga* is 30 cm. The bag limit is 150 kg per week for commercial-artisanal fishermen and 10 kg + 1 fish per trip for sport fishermen.

Life history of the species

The latin name of the *piraputanga Brycon microlepis* Perugia, 1894, was synonymized to *Brycon hilarii* (Valenciennes, 1850). This species belongs to the **Characidae** family and to the **Bryconninae** subfamily (Britski et al. 2007). This species occurs both in the Parana River and the Paraguay River Basins and inhabits lotic and lentic environments (Antunes et al. 2010). *Brycon hilarii* is considered K-strategist (Vazzoler 1996), it is omnivorous and feeds mainly on vegetable matter, seeds and fruits, with a high potential for seed dispersal of riparian vegetation (Reys et al. 2009). Its somatic growth varies seasonally, with the lowest growth rate during the dry season (Mateus and Estupiñán 2002). The average standard length on the landings of the species is 31.8 cm in our study area (Mateus and Estupiñán 2002). The length of first maturation (L_{50}) for females is on average 25.0 cm standard length (Okada et al. 2010). The species perform long-distance reproductive migrations and spawning occurs between November and December (Okada et al. 2010).

Data sources

The data were obtained during the projects “*Fisheries biology and fisheries in the area of influence of the APM Manso: biology, ecology and socioeconomy*” and “*Ichtyological studies around the area of influence of APM Manso*” both executed by the *Núcleo de Pesquisas em limnologia, Ictiologia e Aquicultura* (Nupélia) of the *Universidade Estadual de Maringá* (UEM) and funded by the hydropower company FURNAS, Centrais Elétricas S.A. The first project aimed to obtain fishery dependent data through monitoring of fish landings and fisheries biology studies, and run from the year 2000 to 2008. The second project aimed to obtain fishery independent data through surveys, and run from the year 2000 to 2004. The data and information from these two projects that were used to fit the population dynamics models are listed in Table 1.

The fishery landings data were separated into two series: the “legal fishery”, held between March and October, and the “illegal fishery”, which occurs during the spawning season (the closed season), between November and February every year. Landings were monitored monthly, even during the closed season, by 25 trained samplers from the fishing communities stationed at the main landing port of each fishing village. About 30 % of active fishermen had their landings monitored this way from 2000 to

Table 1 Source of data and information used for fitting the population dynamics model for the stock of *piraputanga* in the Northern Pantanal during the period 2000–2008

Type of the data	Years								
	2000	2001	2002	2003	2004	2005	2006	2007	2008
Fishery-dependent									
Legal landings	o	o	o	o	o	o	o	o	o
Ilegal landings	o	o	o	o	o	o	o	o	o
Legal effort	o	o	o	o	o	o	o	o	o
Ilegal effort	o	o	o	o	o	o	o	o	o
Legal length composition	o	o	o	o	o		o	o	o
Ilegal length composition	o	o	o	o	o		o	o	o
Legal age composition							o	o	o
Ilegal age composition							o	o	o
Fishery-independent									
Juvenile length composition	o	o	o	o	o				
Juvenile CPUE	o	o	o	o	o				
Adult length composition	o	o	o	o	o				
Adult CPUE	o	o	o	o	o				

o—indicates the year in which the data were collected

2008, and 34, 19 and 24 % of the catches for the first, second and third season were sampled (E.K. Okada, personal observation). The total catch by season was raised from the sampled accordingly. Samplings for fisheries biology studies were performed by technicians from UEM. They haphazardly selected fishermen whose activity was monitored for 24 h. All the landings of those fisherman during that time were fully identified to the specific level; all specimens were measured to the nearest mm (standard length), sexed and had their gonadal maturation stages estimated and, from 2006 to 2008, scales were also collected to determine the age composition. Also, for the estimation of age composition, seven local data collectors were trained and were stationed spread out in the study area. They collected samples of scales as well as length (total, furcal and standard). Once a month technicians from UEM visited them and collected the samples that were processed in the laboratory.

The fishery independent data were sampled through surveys using gillnets. Sampling within the reservoir and on the lentic stretches of tributaries were performed with three sets of gillnets operated on the margin both on the surface and on the bottom simultaneously. In the bays, the samples were done with two sets of gillnets operated simultaneously both on the shore and on the bottom. In rest of the sampling stations the set were composed by 17 gillnets, 14 of which were regular gillnets (mesh sizes: 2.4, 3, 4, 5, 6, 7, 8, 9, 10, 12, 14, 16, 18 and 30 cm between opposite knots) and 3 were trammel nets (mesh sizes: 20, 22 and 24 cm between opposite knots). Gillnets were set for 24 h and checked at early morning, dusk and late evening. The survey index for juveniles was computed using the catches from gillnets with meshes 6 cm or lower; the survey index for adults was computed using catches from gillnets with meshes larger than 6 cm. The catches from trammel nets were not used for producing relative abundance indices. Data were reported in number of individuals per 1,000 m^2. Specimens were identified to the species level, measured (standard length), sexed and had their gonadal maturation stages estimated.

Integrated modeling

The population dynamics model for stock assessment was implemented in Stock Synthesis 3.10 (Methot 2005). This model is divided into three components: population dynamics model, observation models and statistical models (q.v. electronic supplementary material and Fig. 1). The population dynamics model represents a population by the abundance, mortality, recruitment and growth functions. The observation models, from the population dynamics states, predict values that shall be seen on the data that are contrasted with the observed values. In the population dynamics model some auxiliary biological information are used, some of which are assumed to be known without errors, such as size at first maturation, length–weight relationship and natural mortality. Parameters estimation is performed by a statistical model, using a negative log-likelihood objective function that is minimized to obtain the best possible fit of the predicted values to the data (Fig. 1). The equations are available in the electronic supplementary material.

Initially we fit a Base-case model to be the "control" for comparison with other scenarios. For the Base-case model, the only forcing function explicitly considered to be acting on the system was the fisheries, both the legal and illegal fleets. For those two fleets the selectivity was assumed to be logistic functions and its parameters were estimated. The two survey indices were assumed to have double normal selectivity. Initially, the model was fitted to data using maximum likelihood, than the Bayesian procedure from Markov Chain Monte Carlo was used, which generated a sample from the posterior distribution for all estimable parameters conditioned on fixed parameter values and on the data, combining priors with likelihoods.

The Base-case was the starting point for the exploration of 86 scenarios that aimed to identify the main sensitivity to the model assumptions. The scenarios were grouped into seven sets: (1) the influence of the data, (2) the influence of natural mortality (M), (3) the ability of the model to estimate the growth coefficient (k), (4) the effect of asymptotic length (L_∞), (5) the effect of the steepness of the Beverton and Holt stock-recruitment curve (h), (6) the effect of the initial conditions; (7) the inclusion of hydrological attributes. Based on the results from the scenarios, two final models were established, which were called MF1 and MF2.

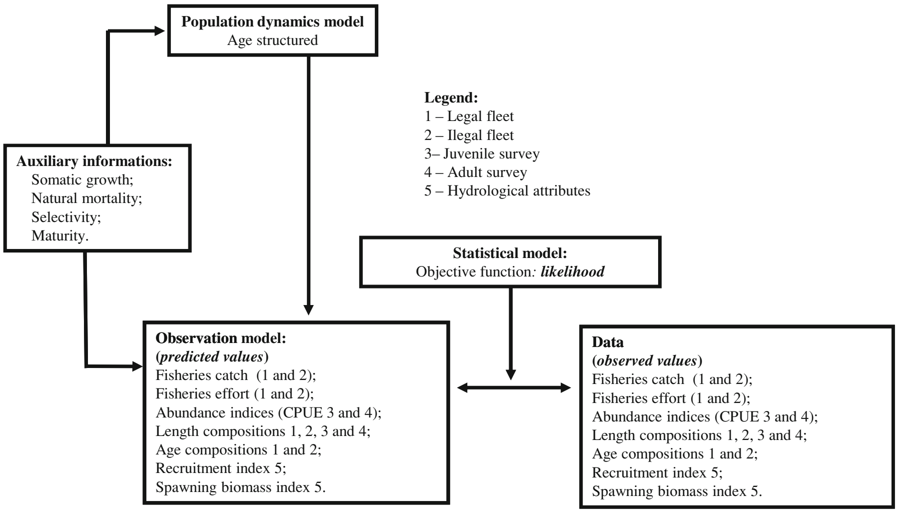
Structure of population dynamics model

The model structure was defined on the basis of available data, exploratory analyses and the literature. The population dynamics model is a one-sex model, structured by age and by season, started in 1980 and finished in 2008, the last year with records of catch data.

The population dynamics model considered both males and females as a single sex because the growth curves estimated by Balbi et al. (unpublished data) for both sexes were similar. The maximum age considered in the model (the plus group) was defined to be 10 years, which was the maximum age in the age-composition data.

Three seasons by years of 4 months each were assumed. The division of season replicated the conceptual model of Ferraz de Lima (1986) on the fishing of migratory fish in the Pantanal of Mato Grosso (Fig. 2). The first season starts in November and runs through February; it is related to the flooding of the river and the spawning season, when fish migrate to reproduce, a behavior named *piracema.* Fishing is prohibited, and by the end of the season the fish enter the flooded areas. The second season runs between March and June; in this period fish remain in the flooded areas to eat and at the end of the season they disperse, this behavior is called *lufada* (Ferraz de Lima 1986), when foraging fish concentrated in schools and

Fig. 1 Conceptual model of the integrated modeling analytical procedure used here



move out of the flooded areas of the river, being more vulnerable to fishing and to predation by piscivorous fishes, that also are attracted by this foraging behavior (*lufada*). At the beginning of the third season, which goes from July to October, the fish are still in the *lufada;* it is a dry and low flow period. Thus, the model assumes that the reproduction occurs in the first season, whereas recruitment occurs in the second one (Fig. 2). Although fishing is prohibited in the first season, illegal fishing occurs in this period. Therefore, two fleets were separated, one is the legal fleet that is active in seasons 2 and 3, and the other is the illegal fleet, which is active only in the first season.

The recruitments were estimated based on the mature biomass from the first season, however, recruits (age-zero individuals) were added into the population only during the second season (Fig. 2). The recruitments were estimated as lognormal deviates from the Beverton–Holt stock-recruitment curve, as reparameterized by Mace and Doonan (1988).

As initial condition we assumed that the initial fishing mortality was 0.5 for both fleets and that the stock was being fished at 257 t and 23.1 t for legal and illegal fleets, respectively, from 1980 to 1999.

Biological information's

The natural mortality (*M*) for Base-case was fixed at 0.58 according to Mateus and Estupiñán (2002). The somatic growth was modeled using the von Bertalanffy curve, reparametrized as length of a young age and asymptotic length (Methot 2005). The size increment happens each season accordingly. The coefficient of variation in length was assumed to increase linearly with age (Methot 2005). The values for growth parameters were fixed for the Base-case at 45.68 cm for asymptotic length (L_∞), 2.18 cm for length of a young age and 0.20 for the growth coefficient (k), as estimated by Balbi et al. (unpublished data). The maturity at length was assumed to follow a logistic function; the standard length in which 50 % of the females are mature (L_{50}) was fixed at 25 cm (Okada et al. 2010).

Hydrological cycles attributes

The attributes used to represent the hydrological cycle were derived from time-series data of daily water level of the Cuiabá River available online at Hydrological

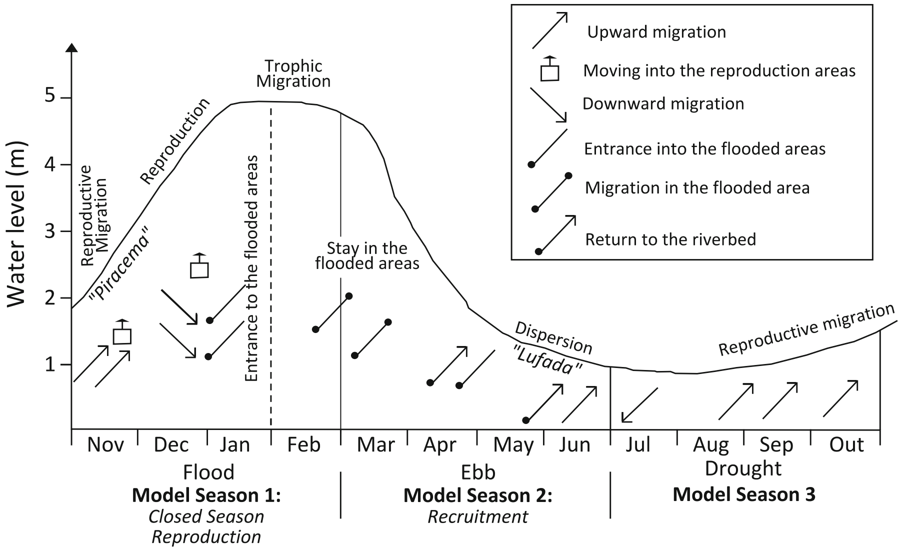


Fig. 2 Conceptual model of the inter-relationship between the variations in river water level and migratory movements of rheophilic fishes. *Piracema*: spawning migration. *Lufada*: displacement (dispersion) of fishes in schools from flooded areas to the riverbed. *Model Season 1* The first season for the model, assumed to be the spawning season and also the season in which the fleet operates illegally because is a closed season. *Model Season 2* Second season of the model, assumed to be the season when recruitment takes place. *Model Season 3* Third and last season of the model. (Redrawn and adapted from Ferraz de Lima 1981, 1986)

Information System (http://hidroweb.ana.gov.br/). These data were collected by *Empresa Furnas Centrais Elétricas S. A.* in the station 66260001 (located at coordinates 56°06'W and 15°36′S) and are under the responsibility of the National Water Agency. The attributes extracted from this series were *intensity*, *duration* and *delay of the floods* (Bailly et al. 2008; Suzuki et al. 2009). These attributes are related to the water level that is considered a flood, which is a threshold level when the floodplain starts to be inundated and connected to the main river. This threshold level was considered to be the average daily water levels from January 1933 to December 1998. Seven years (1946–1948, 1957–1959, and 1996) were excluded from the average because for those only incomplete time series were available. The year 1998 was chosen as the end period for calculation of the historical average because it was the last year before the closure of the Manso dam that altered the flood dynamics of the Cuiabá River.

We considered a period with 5 or more consecutive days with water level above threshold as being a "flood". The *intensity of the floods* for each year was defined as the highest water level for the "year" as defined above (beginning in November of the previous year and ending in October of the current year). The *duration of the floods* was computed for each year as the number of days in which the water level of the river remained above the threshold level, after the beginning of the flood. The end of the flood was considered to be the last day from which there were 5 or more consecutive days below the threshold.

The *delay of the floods* was calculated as the difference (in fortnights) between the day in which the flood began in each year and the first day of the first flood in the historical period. We ranked the attribute values in reverse order for the purpose of indexing it so that the shortest delay of flood received the highest value. Thus, in the year with the longest delay of the flood (in fortnights), this attribute was set to a value of 1; than the number of fortnights were counted so that the year with no delay had the largest value of this attribute.

The attributes of the floods were included in the model as recruitment index (Rec) whose expected value is (exp(recruitment deviations) and/or as a pre-recruit survey occurring before density-dependence (we called spawning biomass index, SPB)) to be fit to (Spawning biomass * exp(recruitment deviations). Those correspond to SS3 special surveys with selectivity patterns 31 and 32 respectively. In this way we modeled the variations in water level as affecting either the egg survival (before density-dependence) or the recruitment survival (after density-dependence) or both. These indices did not act directly as a forcing function (*strictu senso*) in the model, but were added as additional likelihood components so the model could obtain the best possible fit to these rates. This strategy seems to be advantageous because it remains unclear how to include the influence of water level, or any environmental effect, as a forcing function of the system.

Model fit

The parameters were initially estimated by minimizing the negative log-likelihood objective function. The likelihood assumed for the CPUE data of surveys and recruitment deviations was lognormal. Multinomial likelihood was assumed for the age- and length-compositions data (Methot 2005). For the Base-case and for the final models, the Bayesian approach was used to estimate parameters. The Bayesian approach uses the likelihood function in combination with the prior probability densities for the parameters. A join posterior density function was obtained using Markov Chain Monte Carlo algorithm (MCMC). The chains, for each model (Base-case, MF1 and MF2), were obtained with 10 million iteration steps, 100,000 burn-in, and 10,000 thinning interval. Normal prior distributions with high variance were used for the initial recruitment (R_0) and initial fishing mortalities. Symmetric Beta priors were used for the parameters of the selectivity of fleets; uniform prior distributions were assumed for the parameters related to the surveys (selectivities parameters and catchabilities). Three indicators to test for lack of convergence were used: Geweke test, Z-score and lag 1 autocorrelation (Cowles and Carlin 1996; Rohlf and Sokal 1995; Plummer et al. 2004); and all three chains of models for which a Bayesian posterior was obtained converged according to all three indicators.

Results

The total catch sampled from 2000 to 2008 was 664.5 and 59.3 tons for legal and illegal fleets, respectively. The total catch was extrapolated to 3,636 tons between the second season of 2000 and the third season of 2008

(Fig. 3). In this same period, the fishing effort for both fleets decreased and the CPUE showed a positive response (Fig. 3).

Base-case

The data set used and the configuration of the model for Base-case allowed us to recover information about the recruitment and biomass trends of earlier periods than the available length and age composition data. Figure 4 contains the posterior credibility intervals and median of the posteriors distribution functions of the annual recruitment (top) and fishing mortality (bottom) over the years. It was possible to estimate these parameters with a reasonable degree of uncertainty, since 1998. A reverse pattern was observed for the year 2008, the estimate of recruitment this year showed high uncertainty due to the lack of information about that year's cohort on the available data (Fig. 4, top). The recruitment for the year 1998 was strong, while recruitment in the years 1999, 2006 and 2007 were weak. Recruitments from 2000 (after the damming of the Manso river) to 2004 were similar, and, from 2004 to 2006 a downward trend in recruitment was estimated (Fig. 4, top).

The highest rate of fishing mortality occurred in 2000, which had a 50 % probability of belonging to the interval 0.62–0.69, followed by the years 2002 and 2008, with the last added to greater uncertainty (Fig. 4, bottom). After a decrease in the rate of fishing mortality in 2001, this rate suffers a brief increase and subsequent decline in the years 2002–2005 (Fig. 4, bottom).

The recruitment and mortality rates allowed us to estimate the absolute abundance of the stock in the Cuiabá River basin. Figure 5 shows the estimate of the spawning stock biomass and the harvested spawning stock biomass. From 1998 to 2008 it was estimated an extraction of stock caused by fishing of about 7,600 tons of mature females, an annual average of 690 tons, while an annual average of 689 tons remained in the stock (Fig. 5). The highest rate of fishing mortality occurred in 2000 (Fig. 4, bottom), however, this year was marked by less spawning biomass extraction, approximately 483 tons (Fig. 5), which may indicate over-exploitation of juveniles this year.

The model estimated an average of 1,473 tons of virgin spawning biomass. The spawning biomass that would provide the maximum sustainable yield ($\mathrm{SPB}_{\mathrm{MSY}}$) was estimated at 277 t (Fig. 6). Current abundance (2008) of mature females is about half of the virgin and three times higher than the $\mathrm{SPB}_{\mathrm{MSY}}$, indicating that the stock is not overfished (Fig. 6). As the posterior distributions are not overlapping, the uncertainties in these estimates did not affect the conclusion of the stock not being overfished. The fishing mortality rate that will maximize sustainable yield ($\mathrm{F}_{\mathrm{MSY}}$) would be about 0.48, while the current rate has a range of credibility of 95 % between the values 0.47 and 0.54. According to this scenario the probability that overfishing is occurring is 0.68, i.e. fishing mortality rate (2008) is greater than that which produces the maximum sustainable yield (Fig. 6). In 2000 overfishing has also occurred, and there was no overlap between the mortality rates this year with $\mathrm{F}_{\mathrm{MSY}}$. For 2002 and 2003, moreover, there is a

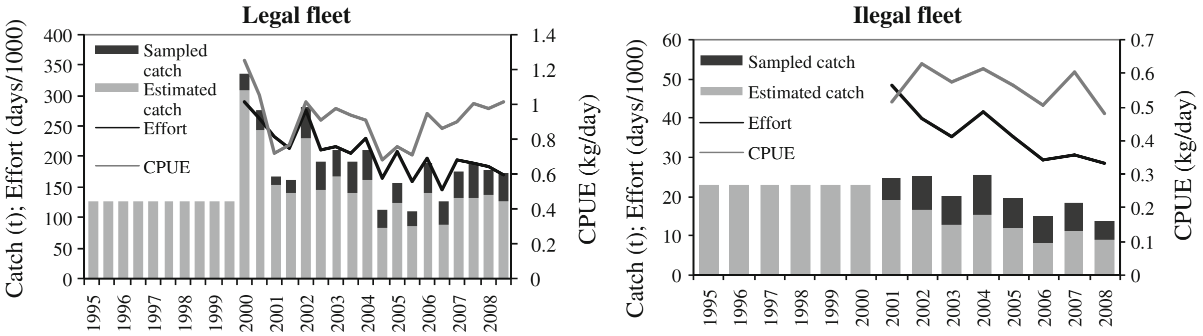


Fig. 3 *Columns* Total catch (sampled and estimated) (tons) per season; *Lines* Estimated total effort (number of fishing days/1,000) per season, and Catch per unit effort (CPUE—tons/fishing days) per season. *Left* Legal fleet (seasons 2 and 3). *Right* Illegal fleet (season 1)

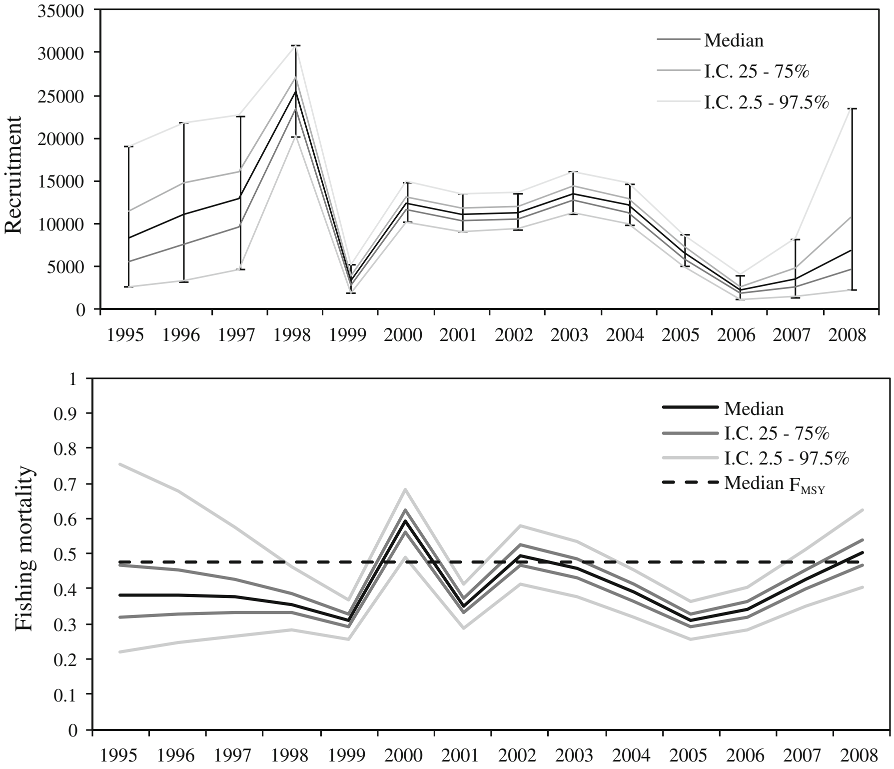


Fig. 4 Median and credibility intervals of 2.5–97.5 and 25–75 % of posterior distributions of recruitment (*top*), annual fishing mortality and median of F_{MSY} posterior distributions (*bottom*) for the Base-case

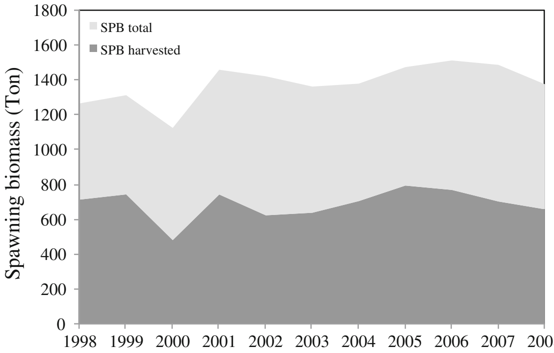


Fig. 5 Biomass of mature females (spawning stock): in the stock (*light gray*) and harvested (*dark gray*) over the years

probability of 0.66 and 0.31 of overfishing having occurred (Fig. 4, bottom; Fig. 6).

Scenarios

Figure 7 shows the effects of each type of data (Table 1) on the estimation of annual recruitment. Different estimates of recruitments have, in general, similar trends, regardless of which data set has been excluded. The largest difference was observed in the period before the year 1998, due to lack of information, and in 2000, soon after the damming of Manso River (Fig. 7).

Information about a strong recruitment in 1998 comes from the lengths compositions of the juvenile and adult surveys, because when these two data sets were excludes, we simultaneously observed that the estimates of recruitment of this year was lower (Fig. 7). The age composition of both fleets exerted strong influence on the general pattern of recruitment from 2002 to 2008. Whereas the first year of data available for this set was 2006, it is possible to observe that age-composition data were able to retrieve information from the previous 4 years.

Natural mortality was set at different values (0.20–0.75, values greater than 0.75 showed convergence problems). This parameter did not influence the trend of recruitment, but the scale, and further values of natural mortality led to estimates of higher recruitment (Fig. 8a).

When the natural mortality rate was fixed at different values and the growth coefficient (*k*) was set free to be estimated, a negative relationship between those two parameters became evident (values

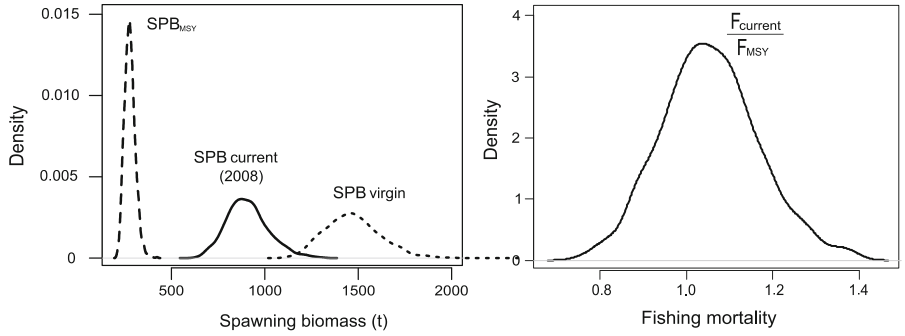


Fig. 6 *Left* marginal posterior distributions of the virgin spawning biomass (*dotted line*), current (2008—*solid line*) and that maximizes the sustainable yield (*dashed line*), *Right* marginal posterior distributions of the ratio between the current fishing mortality (2008) and fishing mortality that maximizes the maximum sustainable yield

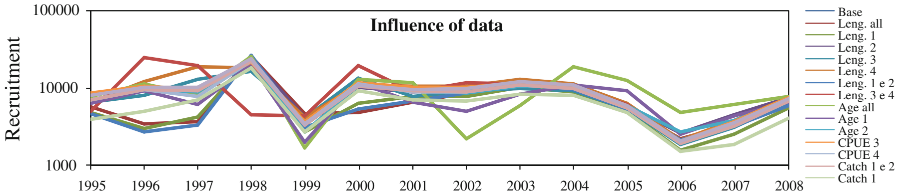


Fig. 7 Influence of different datasets on estimates of annual recruitment. *Each line* represents the estimate of recruitment without the use of the specified dataset. The numbers 1, 2, 3 e 4 on the legend represent the data source (*1* legal fleet; *2* illegal fleet; *3* juveniles survey; *4* adults survey). *Base* Base-case; *Leng.* Length data composition (all: refers to number 1, 2, 3 e 4, simultaneously); *Age* Age data composition (all: refers to number 1, 2, 3 e 4, simultaneously); *CPUE* Catch per unit effort, *Catch* Total catch of the fleet

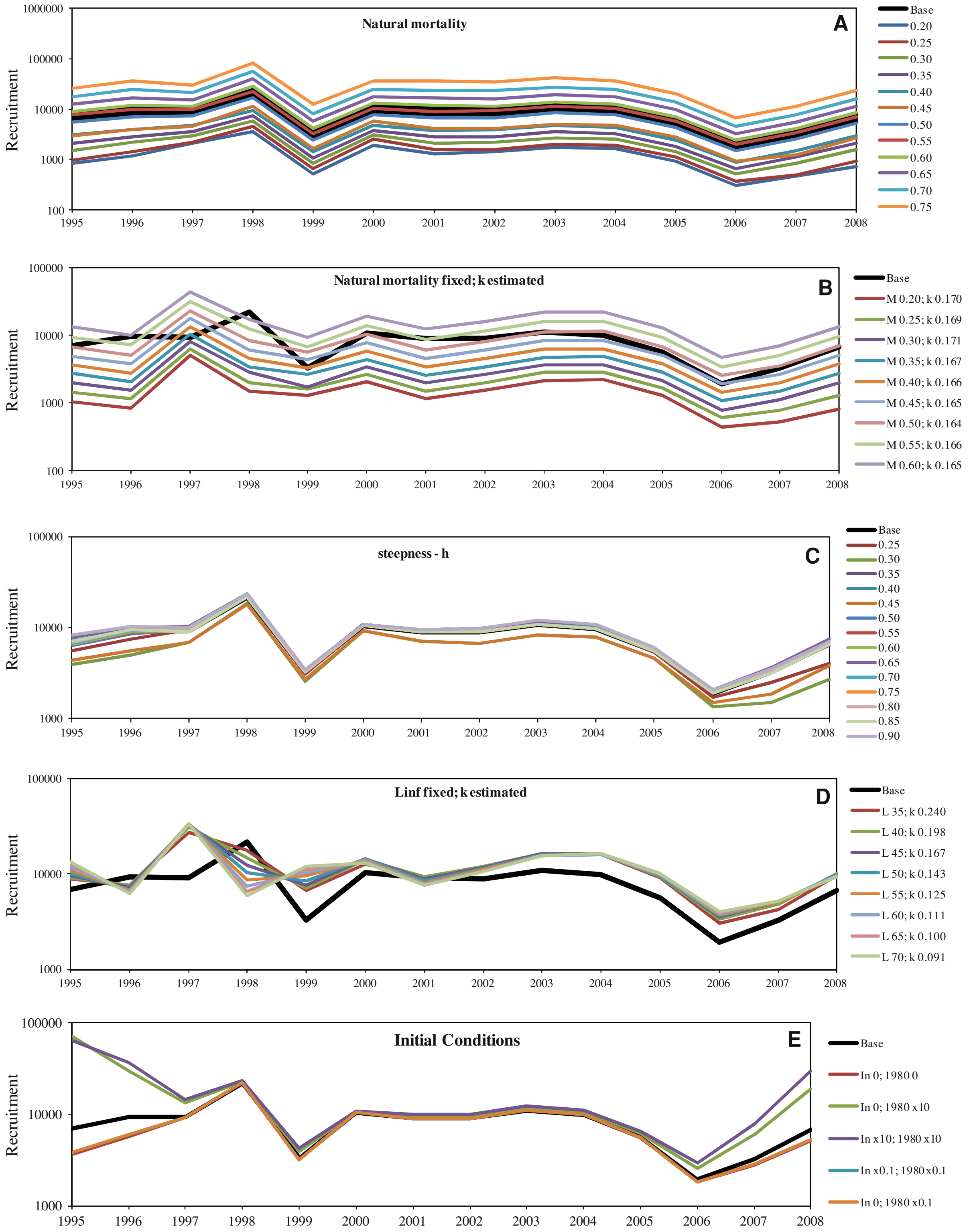
greater than 0.6 for M presented convergence problems) (Fig. 8b). When k was set free to be estimated, the models converged to lower values of k (from 0.164 to 0.171) than it was set in the Base-case (0.20). The overall effect was a shift in the peak of recruitment from 1998 to 1997 and an increase in the 1999 recruitment (Fig. 8b). The low variation in k was due to the asymptotic length (L_∞), fixed at 45.68 cm as in the Base-case. Changes in the L_∞ values while letting k to be estimated also showed a negative correlation of L_∞ (models with L_∞ set to values smaller than 45 cm showed convergence problems), however a wider range of k values were estimated, but always lower than the Base-case, which also influences the estimates of the recruitment from 1997 to 1998 (Fig. 8c).

The slope of the Beverton–Holt stock-recruitment curve (steepness h) was set at different values (0.25–0.90) but did not affect the estimates of annual recruitment (Fig. 8d).

The impacts of the assumptions about the initial conditions were also assessed. The initial fishing mortality and catches during the years 1980–1999 were set to ten times larger and ten times smaller than the Base-case. The model was able to estimate the same tendency for the recruitment from 1998 to 2006 regardless of initial conditions assumed (Fig. 8e). The lack of information on historical catches was not a problematical factor for estimation of recruitments, but may influence the estimates of the state of the stock, as they affect the estimate of virgin biomass, which ranged from 1110 to 6,079 t between the different scenarios assumed for the initial conditions.

Hydrological attributes

The average daily water level of the Cuiabá River was 242 cm, which was considered to be the threshold value for the flood. The first fortnight of November was period when the onset of flood was the earliest. The three attributes of the hydrologic cycle analyzed (intensity, duration and delay of the floods) showed similar trends, with a marked decrease for the years 2000 and 2001, shortly after the formation of the APM Manso dam (Fig. 9).


Natural mortality
A
Recruitment
1000000
100000
10000
1000
100
1995
1996
1997
1998
1999
2000
2001
2002
2003
2004
2005
2006
2007
2008
Base
0.20
0.25
0.30
0.35
0.40
0.45
0.50
0.55
0.60
0.65
0.70
0.75
Natural mortality fixed; k estimated
B
Recruitment
100000
10000
1000
100
Base
M 0.20; k 0.170
M 0.25; k 0.169
M 0.30; k 0.171
M 0.35; k 0.167
M 0.40; k 0.166
M 0.45; k 0.165
M 0.50; k 0.164
M 0.55; k 0.166
M 0.60; k 0.165
steepness - h
C
Recruitment
100000
10000
1000
Base
0.25
0.30
0.35
0.40
0.45
0.50
0.55
0.60
0.65
0.70
0.75
0.80
0.85
0.90
Linf fixed; k estimated
D
Recruitment
100000
10000
1000
Base
L 35; k 0.240
L 40; k 0.198
L 45; k 0.167
L 50; k 0.143
L 55; k 0.125
L 60; k 0.111
L 65; k 0.100
L 70; k 0.091
Initial Conditions
E
Recruitment
100000
10000
1000
Base
In 0; 1980 0
In 0; 1980 x10
In x10; 1980 x10
In x0.1; 1980 x0.1
In 0; 1980 x0.1

◀**Fig. 8** Estimates of annual recruitment with different sets of scenarios (**a**, **b**, **c**, **d**, **e**). **a** Different values set for natural mortality (*M*), with the growth coefficient (*k*) set at 0.2; **b** different values set for natural mortality, with *k* set free to be estimated by the model; **c** different values set for asymptotic growth (L_∞), with k set free to be estimated by the model; **d** different values for the slope of the Beverton–Holt stock-recruitment curve (steepness—*h*); **e** different initial conditions: In—initial equilibrium fishing mortality, 1980 × 10-years 1980–1999 were set to ten times larger than the Base-case and 1980 × 0.1-years 1980–1999 were set to ten times smaller than the Base-case

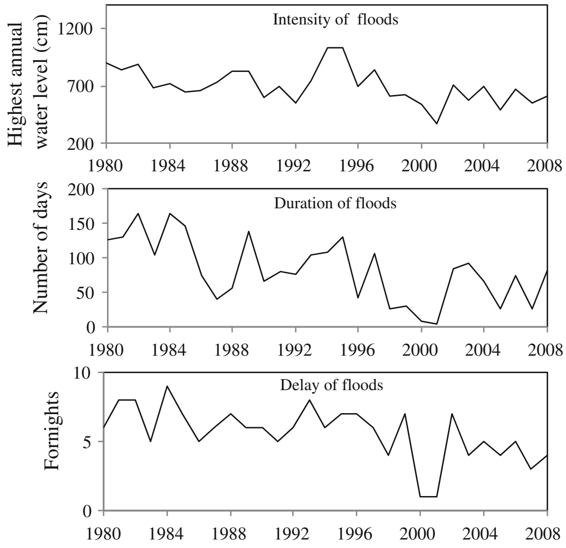


Fig. 9 Hydrological cycle attributes (intensity, duration and delay of the floods) for the period included in the models (from 1980 to 2008). The delay of the flood is ranked inversely, during the years with the longest delay of the flood (in fortnights) this attribute was set to a value of 1 (years 2000 and 2001); than the number of fortnights were counted so that the year with no delay had the largest value of this attribute

The inclusion of hydrological attributes as recruitment and spawning biomass indices increased the estimates of annual recruitment of the years 1997, 1999, 2006 and 2007, compared to the Base-case (Fig. 10). The trends in these recruitments were quite similar, as hydrological attributes had similar tendencies (Fig. 10). The delay of the floods, however, was the hydrological attribute that provided the best model partial fits to the data (considering only the components surveys, length- and age-compositions) either by including it as recruitment or as a spawning biomass index (Table 2). The inclusion of the hydrological attributes, both as a recruitment and as a spawning biomass index simultaneously, was done in two sets of scenarios, which are different in relation to growth parameters: in the first set of scenarios the asymptotic length L_∞ was fixed at 45.68 cm and the growth coefficient was fixed at 0.2; in the second set of scenarios those growth parameters were set free to be estimated (Fig. 10). The two sets of scenarios had better partial fit than the Base-case overall; the best partial fit was obtained when the attribute intensity of the floods was linked to the recruitment index (Table 2).

The different combinations of the recruitment and spawning biomass indices resulted in different estimates for the growth parameters, with an average value of 63.75 cm for the asymptotic length and 0.104 for the growth coefficient (Table 2). The mean value of the asymptotic length was used to perform a likelihood profile on natural mortality values (0.30, 0.40, 0.45, 0.50, 0.55 and 0.60). The lowest negative log-likelihood model was the one that included the natural mortality set at 0.45 and that estimated the growth rate of 0.104. However, this coefficient showed an average value of 0.103 when a range of values for the natural mortality were considered (Table 3).

Natural mortality had a positive relationship with the magnitude of annual recruitment estimates (Fig. 8a, b), so, this is a key parameter for conclusions about the state of the stock. Values from 0.45 to 0.60 for natural mortality indicate that the stock is not overfished and overfishing is not occurring, the opposite conclusion is obtained with values lower than 0.40 (Fig. 11).

Final models

The explorations of the different scenarios from the Base-case allowed for the definitions of two final models, which took into account the uncertainty around the natural mortality. Both final models were estimated using Bayesian MCMC procedure with the asymptotic length set as 63.75 cm and the growth coefficient set as 0.103, the average values for the sensitivity scenarios (Table 2). In the first final model (MF1), the natural mortality was set as a free parameter to be estimated, and in the second final model (MF2) the natural mortality was set at 0.45, the best estimate obtained in the sensitivity scenarios (Table 3). Figure 12 shows the marginal posterior distribution of the natural mortality (from MF1), the 50 % of credible range for M is 0.59–0.67 and the median in 0.63.

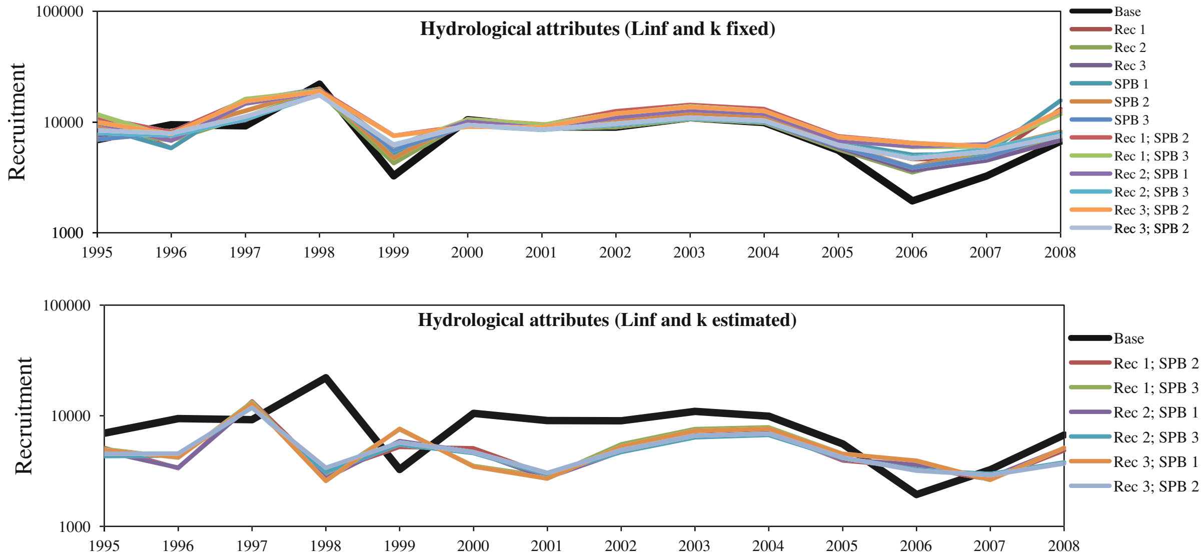


Fig. 10 Estimates of annual recruitment with different combinations of hydrological attributes (*1*: duration of the floods; *2*: Intensity of the floods; *3*: Delay of the floods) for recruitment (Rec) and spawning biomass (SPB) indices, with fixed (*Top*) and estimated (*Bottom*) growth parameters

Table 2 Total and partial [size and age compositions and CPUE of the surveys) negative log-likelihoods (−log (L)] for different combinations of recruitment and spawning biomass indices

Recruitment	Spawning biomass	Fixed Linf and k		Estimated Linf and k			
		−log(L) parcial	−log(L) total	−log(L) parcial	−log(L) total	Linf	k
Duration	–	879.19	889.43	–	–	–	–
Intensity	–	889.57	830.45	–	–	–	–
Delay	–	766.47	868.12	–	–	–	–
–	Duration	878.00	876.62	–	–	–	–
–	Intensity	872.17	810.25	–	–	–	–
–	Delay	758.25	860.76	–	–	–	–
Duration	Intensity	888.89	884.26	847.41	852.04	53.52	0.129
Duration	Delay	898.02	905.72	856.68	860.98	57.08	0.115
Intensity	Duration	888.21	862.12	836.25	802.28	71.93	0.088
Intensity	Delay	905.83	843.24	823.82	763.02	74.89	0.085
Delay	Duration	896.64	891.96	848.08	830.87	66.51	0.095
Delay	Intensity	905.02	851.43	844.73	794.69	58.58	0.115
					Mean	63.75	0.104

Fixed and estimated growth parameters were considered. For comparison, the Base-case had 824.23 of total negative log-likelihood and 860 of partial negative log-likelihood

The final models had a good fit for legal and illegal fleets, and both of them showed declines after 2004 (Fig. 12). This period coincides with the relative low abundance observed in the juvenile and adult survey indices. However, the model subtly fitted to these indices, showing a slight decline from 2000 to 2004 (Fig. 13). The fits of these models to the age- and length-compositions data can be seen in the electronic supplementary material.

In both final models, the intensity of the flood and the delay of the flood hydrological attributes and were well adjusted to the rates of recruitment and spawning

Table 3 Total likelihood profile [−log (L)] for fixed values of natural mortality (0.30, 0.40, 0.45, 0.50, 0.55 and 0.60) and the estimated growth coefficient (k) for the model with recruitment (intensity of the floods) and spawning biomass (delay of the floods) indices

Natural mortality	−log(L)	Estimated k
0.30	797.97	0.104
0.35	784.99	0.103
0.40	764.33	0.105
0.45	760.38	0.104
0.50	768.84	0.102
0.55	766.99	0.101
0.60	765.99	0.101
	Mean	0.103

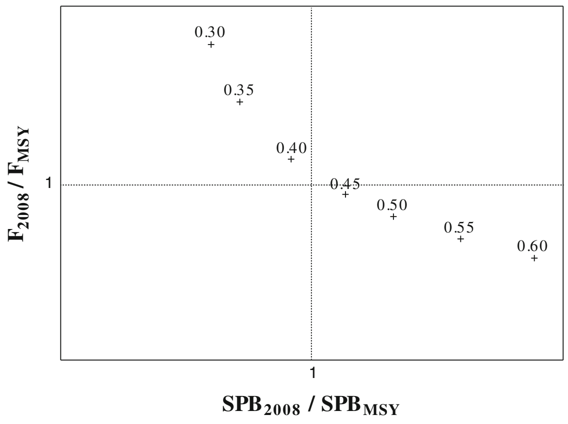


Fig. 11 Perception of the state of the stock based on the values of natural mortality. In the *horizontal axis*, the ratio between spawning biomass (SPB) of 2008 and SPB_{MSY} that provides maximum sustainable yield; values larger than 1 showed that the stock is not overfished. In the *vertical axis* the ratio between fishing mortality (F) in 2008 and that provides long-term sustainable yield (F_{MSY}); values greater than 1 show that the overfishing is occurring

biomass, respectively (Fig. 14). Both indices had similar trends and a decline in 2001, indicating a failure in recruitment this year (Fig. 14). The spawning biomass index was zero for 2000 and 2001, since there was no flood during these years due to the damming of the Manso river in December 1999. This seems to have negatively affected the spawning biomass and recruitment, but the model was not "forced" to fit exactly this trend of decrease of hydrological attributes, being flexible to consider all data sets in our setting (Fig. 14).

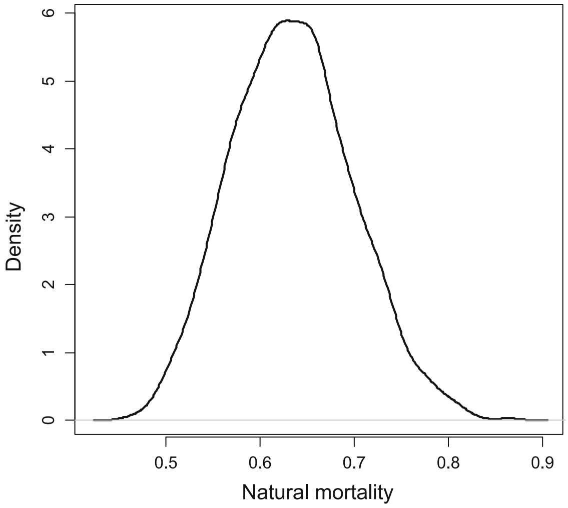


Fig. 12 Marginal posterior distribution for natural mortality estimated by the model with the growth parameters of 63.75 cm asymptotic length and growth coefficient of 0.103 and the intensity of the flood as recruitment index and the delay of the flood as the spawning biomass

The average of the mature biomass needed for maximizing the sustainable yield SPB_{MSY} was estimated to be 278 and 457 tons by MF1(free M) and MF2 (Fixed M), respectively (Fig. 15). From the results of MF1 is safe to say that the stock is not overfished, while in MF2 there is 0.011 of probability that the stock is overfished (Fig. 15). The main difference between these two results is related to fishing mortality because for MF1 there is a 0.68 probability that overfishing is occurring and for MF2 there is even stronger evidence exist that overfishing is occurring. The current level fishing mortality (of 2008) is up to three times higher than the level that maximizes the sustainable yield according to MF2 (Fig. 15). Thus, we can conclude that the stock of *Brycon hilarii* of the Northern Pantanal is still not overfished, however, the SPB must be rapidly declining due to the high rates of current fishing mortalities.

Discussion

This work possibly brings the first application of integrated analysis for tropical freshwater ecosystems documented in scientific literature and combines different fisheries dependent and independent data sets, as well as information on the hydrological regime, to which the fishing in tropical rivers is linked

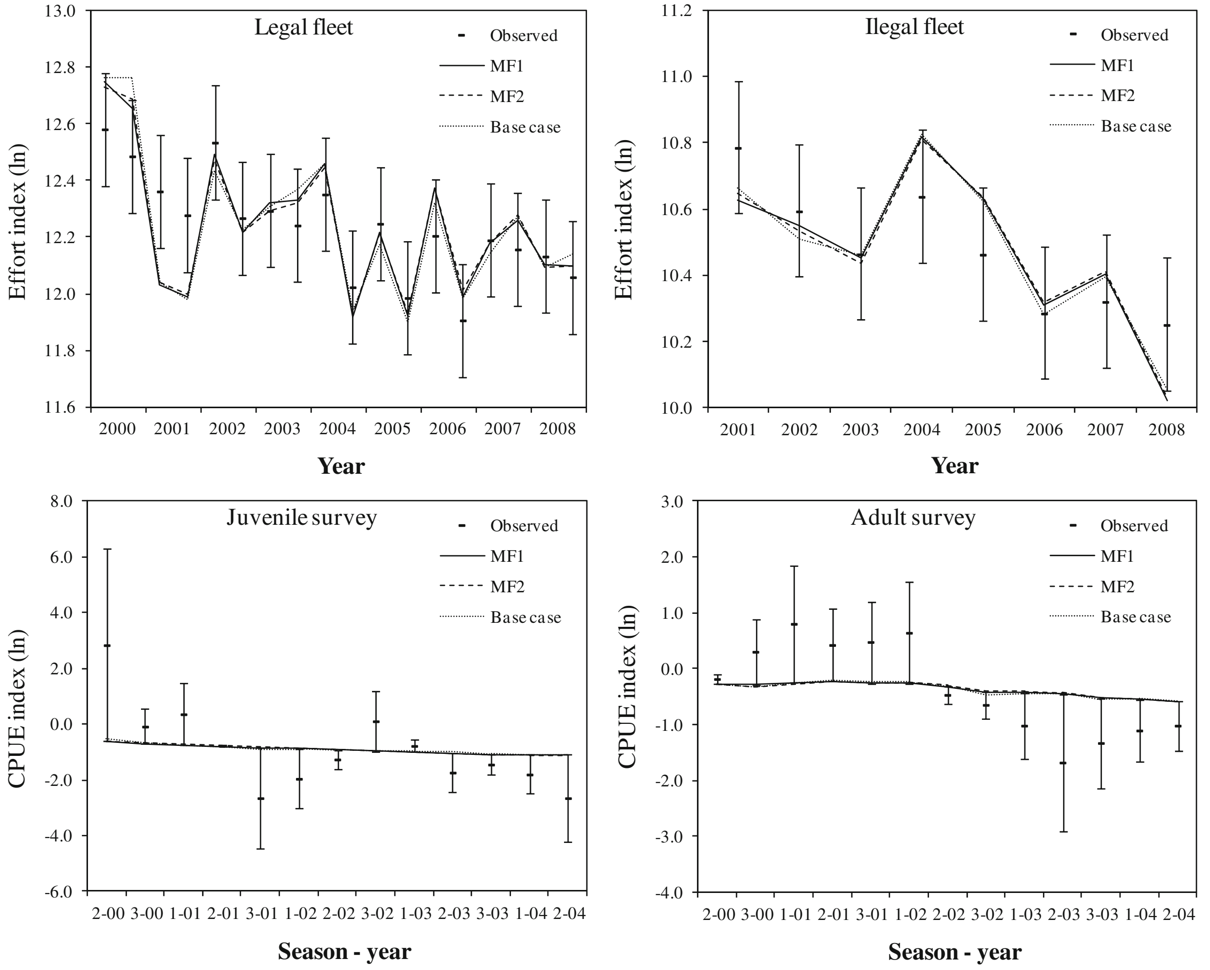


Fig. 13 *Top* legal and illegal fleets fishing effort indices observed and predicted by the models MF1 (*solid line*), MF2 (*dashed line*) and Base case (*dotted line*). *Bottom* juvenile and adult surveys CPUE indices observed and predicted by the models MF1, MF2 and Base case

(Welcomme and Halls 2003). Still, the final models were fit using the Bayesian approach that allowed representing the uncertainty through the posterior distributions (Kinas 1996; Maunder 2003).

Importance of data

In this study, both the length- and the age-composition data proved to be important in model fitting, regardless of their origin as commercial fishing data or survey data. The surveys, however, allow to access parts of the population not exploited by the commercial fishing (Hilborn and Walters 2001), such as the length composition of juveniles through the capture of individuals smaller than minimum length permissible for commercial fishing (30 cm total length). Thus, both surveys and age composition of commercial fishing had strong influence on estimates of recruitment. These data allowed both for the retrieval of information about the structure of the population and for inferences on periods prior to the data collection (Alves et al. 2011).

Data collection for the artisanal inland fisheries aggregates numerous hindrances (FAO 2003), among which we highlight the diversity of species landed, the range of fishing gear used, and the fact that landings are made in a diffuse manner (several landing points, with some located in remote locations) (Bayley and Petrere 1989; FAO 2003). Many studies have already criticized the quality and the quantity of these data for South America (Welcomme 1990; Petrere 1995;

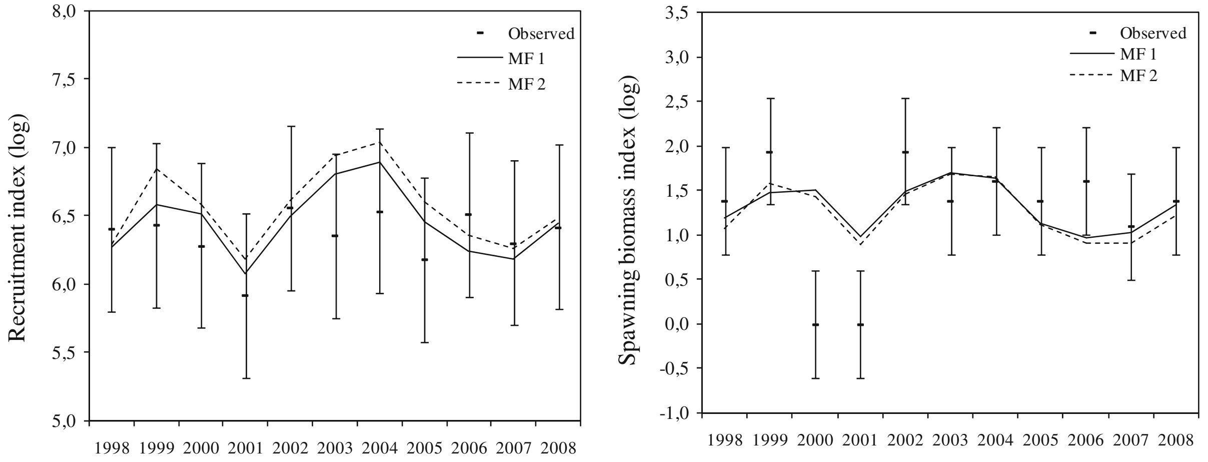


Fig. 14 *Left* intensity of the floods (observed) adjusted to the recruitment index by MF1 (free M) and MF2 (fixed M). *Right* Delay of the floods (observed) adjusted to the spawning biomass index for MF1 (free M) and MF2 (fixed M)

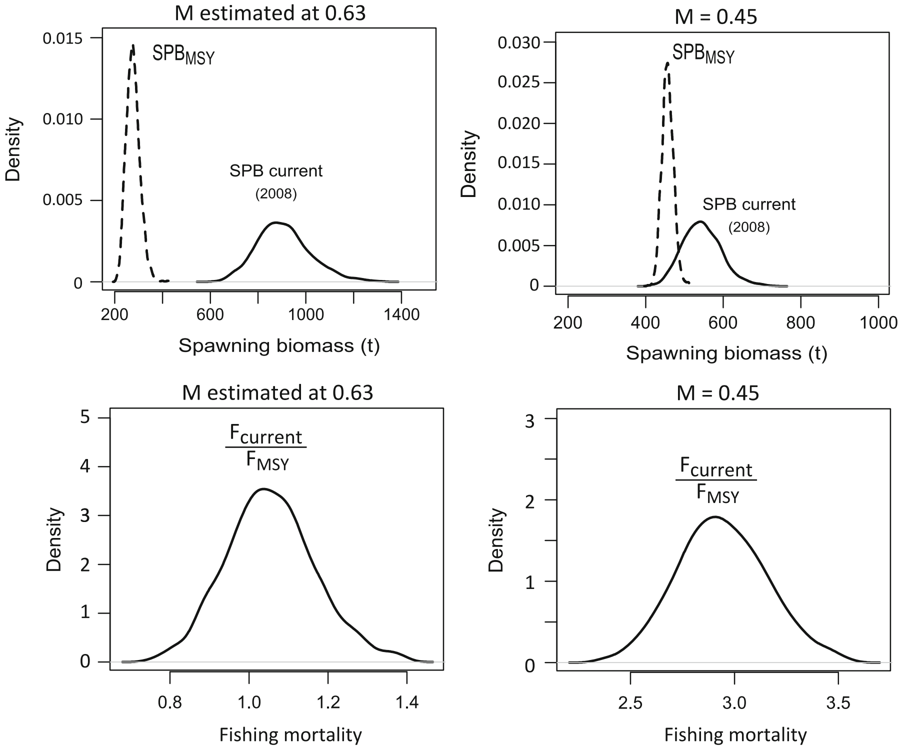


Fig. 15 Top: marginal posterior distributions of the current spawning biomass (2008—*solid line*) and the SPB that produces the long term sustainable yield (*dashed line*) for MF1 (Natural mortality estimated by the model at 0.67) and MF2 (natural mortality set at 0.45). *Bottom* marginal posterior distributions of current fishing mortality (2008—*solid line*) and the fishing that maximizes the long-term sustainable yield F_{MSY} (*dashed line*) for MF1 and MF2

Agostinho et al. 2007; Junk et al. 2007) and Africa (Ticheler et al. 1998; Béné and Neiland 2003; FAO 2003), however, there are some isolated but notable efforts for systematizing the fisheries monitoring programs that have great importance to "supplement" the lack of data from official statistics (Alves and Minte-Vera 2012). The direct involvement of fishermen through participatory monitoring in the collecting data process has been highlighted for both African (Ticheler et al. 1998; FAO 2003; Carvalho et al. 2009) and South American inland environments (Ruffino 1996; Catella et al. 2008). The integrated analysis enables to channel the heterogeneity of origins and of the data types to be fit to a single population dynamics model, which makes it a promising tool for management of artisanal fisheries.

In the assessment of stocks that are exploited by fishing, natural mortality (*M*) is one of the key parameters, and one of the most difficult to be determined (Zhang and Megrey 2006). In this work we use the empirical estimate derived by Mateus and Estupiñán (2002) from the empirical equation of Pauly (1980) as natural mortality of the Base-case. However, several scenarios were estimated with different values of natural mortality. High values of natural mortality implied more productive stock, and the conclusion that the stock is neither overfished nor overfishing is occurring, while lower values of M implied less productive stocks, and the conclusion that stock shall be overfished and overfishing is occurring. The final model adopted indicated that the probability density distribution of the natural mortality is between 0.5 and 0.8, implying that the stock is not overfished, but should be rapidly declining due to high fishing mortality rates. The different scenarios of *M*, however, did not affect the trends in recruitment, only its magnitude. Thus, this model can be used for the study of variation of recruitment.

In the models developed in this work we included the fishing effort, which was extrapolated to the entire sampling universe increasing the uncertainties. Hilborn and Walters (2001) point out that the fishing effort data have high noise and its inclusion in linear models for standardization of CPUE can make them a more reliable source of information. Thus, it is suggested that future studies include standardization of the series of capture in order to reduce the uncertainties of the model.

The use of fishing effort data to assist in obtaining the fishing mortality depends on the catchability coefficient (Arreguín-Sánchez 1996), which reflects the efficiency of fishing gear, variable with the environmental factors (Arreguín-Sánchez 1996). This was one of the reasons for the decision to separate fleets operating in the wet season (illegal) and outside it (legal fishing), because we assumed that in the flooding season, the capture efficiency is reduced due to the greater volume of water, which was observed in the fishery in the medium Amazon (Mota and Ruffino 1997). The catchability coefficient, in the parameterization adopted here, is not estimated, but calculated internally in the model as a scaling factor to relate the average fishery CPUE to the average abundance in the population.

Growth parameters found in the literature for *Brycon hilarii* of the Northern Pantanal varied from 40 cm standard length (SL) and 0.16 k, in the years 2006–2008 (Balbi et al. unpublished data), to 71 cm fork length and 0.29 k, for the years 1996 and 1997 (Mateus and Estupiñán 2002). This discrepancy can be attributed to the difference between the methodology used, or even the fact that Mateus and Estupiñán (2002) have used the market data of Cuiabá, in which about 33 % of the landing of *Brycon hilarii* comes from the Paraguay River where larger fish may occur (Mateus et al. 2004). However, it cannot be ruled out that in these 10 years, the fishing is exerting a selective pressure for phenotypic individuals of smaller size and slower growth (Parma and Deriso 1990; Berkeley et al. 2004). These parameters, when freed to be estimated by our model, showed intermediate values of asymptotic growth (63.75 cm SL) and lower growth coefficient (0.104) than those reported in the literature, and our data covered an interim period in these studies.

Hydrological attributes

The assessment of rheophilic fish stocks should be able to include in a flexible way the effect of hydrological regime as an additional forcing function of the system, and in this work, we tried to achieve this flexibility by making the recruitment annual rates and spawning biomass adjustable to the attributes of the hydrological cycle data. Our models showed that these attributes affected the population dynamics of *Brycon hilarii* of the Northern Pantanal, and their inclusion in the models allowed best partial fits (which considers only the components of the length and age compositions, and CPUE of juveniles and adults) to the data than to the Base-case (no hydrological attributes). The spawning biomass and annual recruitment indices allowed better partial fits when the attributes “delay of floods” and “intensity of floods” were respectively used, indicating that these characteristics of the population may be influenced by specific attributes of the water level, by for example interfering with the survival of eggs or recruits. Similar patterns to these were also observed for other species of the Northern Pantanal (Bailly et al. 2008), Upper Paraná River Basin (Suzuki et al. 2004) and Africa (Paugy 2002).

The use of integrated models made it possible to translate the conceptual model of Ferraz de Lima (1986) into a population dynamics model and to reconstruct the dynamics of a rheophilic fish population in the Northern Pantanal using fishing as a main forcing function and the attributes of the hydrologic cycle as additional forcing functions. These models can also be used to perform projections, which may contain a variety of future scenarios (hypotheses) about the changes in fishing levels and the hydrological attributes. Thus, it would be possible to predict the impact of each hypothesis on the dynamics of these populations, which could support both the process of decision making (Arthington et al. 2007), as studies of population ecology of migratory species and riverine ecology.

Recommendations

Historical records of the fishing production landed in the market of Cuiabá between the years 1980 and 1984 do not include the *piraputanga* among the main resources exploited because this species have a small volume in the market place (Ferraz de Lima 1986). In addition, Mateus and Estupiñán (2002), despite concluding that the *piraputanga* stock was not being overfishing by the year 1997, highlighted the trend in increased consumption of this species due to the growing appreciation of its flavor. Our work showed evidence that the stock of *Brycon hilarii* of the Northern Pantanal is not yet overfished but is overfishing is occurring. Thus, we can infer that the overfishing issue is relatively new to this stock, and so, it is not yet overfished. However, measures are needed to reduce the fishing effort exerted on this stock to prevent its further depletion.

The last stock assessment of Northern Pantanal occurred more than 10 years ago and used data from 2000 to 2001, and it was published in 2007 (Mateus and Penha 2007). Moreover, the delay between the data collection and the availability of stock assessments of the Pantanal North on scientific literature is on average greater than 6 years (Mateus and Estupiñán 2002; Mateus and Petrere 2004; Penha and Mateus 2007). This delay between the completion of data collection and publication of scientific papers of inland fisheries in Brazil is considered a concern for fisheries management, which should be guided by the use of current best scientific information available (Alves and Minte-Vera 2012).

It is recommended that the stocks of the Northern Pantanal were systematically monitored to enable periodic stock assessments, avoiding late discoveries and corrective decision-making. These studies in the Northern Pantanal region and in all country should be included as mandatory in the agendas of the relevant agencies with mandates for the management of fisheries resources, similarly as done elsewhere in countries like USA, Canada and New Zealand. However, unlike these countries, in Brazil, the inland fisheries management must also be integrated into the management of the water level aimed at developing an integrated watershed management, representing a genuine Brazilian system.

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